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Urban Breeding Gull Surveys: A Review of Methods and Options for Survey Design

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EXECUTIVE SUMMARY

1. This report has been commissioned by Natural England to inform the design and implementation of future census efforts for breeding urban gulls, and to make recommendations for the most cost-effective survey strategy for delivering urban gull population estimates for the UK and Republic of Ireland, as well as any specified key sites.

2. Within this report we review existing and potential urban gull survey methods (section 2); review the existing knowledge of breeding gull distribution within the UK and Ireland (section 3); and develop a bespoke survey design to deliver an urban gull census (section 4).

3. The review of existing methods covers land-based methods (section 2.2), including: counts from vantage points, sample quadrat counts; transect counts (and distance sampling) and flush-counts of adults; aerial methods (section 2.3), including: digital aerial survey (DAS), visual aerial survey, unmanned aerial vehicles (UAVs), microlight, cameras on kites, aerial thermal imagery and satellite; and survey methods for Kittiwake (section 2.4). The benefits and risks of each method are considered (including privacy issues for aerial survey methods and health and safety) as well as costs. Techniques to account for detectability (double observer, distance sampling) are also discussed.

4. Generally, land-based methods create more disturbance to birds than aerial methods and are likely to be more labour- and time-intensive than aerial methods. However, land-based methods may better enable differentiation between species and between breeding and non-breeding birds. The most suitable remote methods from a purely technological viewpoint are likely to be digital aerial stills, video and UAV. These methods also provide repeatability, permanent data record and adequate resolution for differentiating gull species. However, flight restrictions for UAVs may render their use impractical in urban settings for anything other than targeted surveys of some known colonies.

5. In section 3 we provide a review of the current breeding distribution of gulls in relation to urban areas. Using data from Bird Atlas 2007-2011 (Balmer *et al.* 2013) and urban land cover data from Land Cover Map 2007 (Morton *et al.* 2011), we show that there is a clear association between breeding gull occurrence and urban areas at inland sites, but also that gulls may nest inland even at sites with very low urban cover. The 'urban' habitat in which gulls may nest (i.e. man-made structures, and particularly flat rooftops) occurs virtually everywhere in the UK and Ireland, both in areas of high urbanisation and in landscapes that would otherwise be defined as 'rural' or not urban, by any habitat classification scheme. Given this, a truly complete census of urban gull populations in the UK and Ireland is unlikely to be feasible.

6. As an alternative, in section 4 we thus propose a broader survey using a paired key site and stratified sampling approach, the latter covering the entire spectrum of urbanisation. The proposed stratification would be based on gull abundance, region, % urban cover, and whether the site is coastal or inland. We suggest, in the first instance that the survey would best be achieved by digital aerial survey, given the practicalities of using cherry pickers or vantage point surveys on a broad scale.

7. Costs are thus provided separately for coverage by digital aerial survey of potential key sites and for covering any 10 km square within defined regions and at the country level. Nevertheless, it should be noted that the key sites identified in this report represent a potential suite of sites that might be selected, and it is likely that a final choice of key sites

will depend on casework needs and statutory monitoring priorities. Once a final selection of key sites has been determined, consideration should be given as to whether it may be possible to save on survey costs at some of these sites by using alternative methods, such as visual aerial survey, cherry pickers or vantage point surveys, especially where these have proven successful before, utilising volunteer or public involvement where appropriate.

1. INTRODUCTION

1.1. Background

Natural England is engaged with the other Statutory Nature Conservation Bodies (SNCBs), BTO, JNCC, RSPB and other project partners to organise and deliver the periodic census of breeding seabirds for the UK, its constituent countries and dependencies and the Republic of Ireland (subsequently referred to as the UK and Ireland), currently under the working title 'Seabirds Count'. As part of this census, robust estimates of the number of gulls nesting in urban environments are required. This report has been commissioned by Natural England to inform the design and implementation of future census efforts, and to make recommendations for the most cost-effective survey strategy for delivering urban gull population estimates for the UK and Ireland, as well as any specified key sites.

In the context of this project, 'urban' is taken to mean all man made (non-natural) habitats, including but not limited to buildings and other structures found in villages, towns, cities and industrial land. 'Urban gull' thus refers to any member of the gull family that is found in and around built-up areas of human habitation (Calladine *et al.* 2006). In the UK and Ireland these are in particular Herring Gull, *Larus argentatus* and Lesser Black-backed Gull*, Larus fuscus*; but also include Black-headed Gull, *Chroicocephalus ridibundus*; Common Gull, *Larus canus*; Great Black-backed Gull, *Larus marinus* and Kittiwake *Rissa tridactyla.*

1.2. Objectives of this report

The overall project objective is to develop a plan for surveying urban nesting gulls in the UK and Ireland, with a full breakdown of likely cost. Within this, the main objectives of this report are:

- To review existing and potential urban gull survey methods with reference to supporting literature, providing details of work needed to provide or test proposed methods (section 2);
- To review the existing knowledge of gull distribution, referring to available evidence (e.g. the latest Britain and Ireland Bird Atlas, (Balmer *et al.* 2013) in as much detail as possible (section 3);
- To formulate a bespoke survey design to deliver an urban gull census, drawing on information from section 2 and section 3 (section 4); to also consider a sampling design approach, as an addition option to a full census design.
- To provide costings for the census, according to the suggested methodological approach (including costs per unit effort and the estimated number of units). These are to be provided to Natural England as a separate confidential document.

1.3. Challenges associated with surveying urban breeding gulls

Counting breeding gulls is challenging for a number of reasons – species nest in colonies of widely varying sizes and at varying densities. Their extended breeding season means that single counts made early in the season might exclude a large number of late breeding attempts. In addition, the nests of similarly sized species (e.g. Herring Gull and Lesser Black-backed Gull) are difficult to differentiate unless occupied by an adult, meaning remote or vantage point methods that do not displace the adults from the nests have an advantage.

One of the key challenges associated with surveying urban gull populations in particular is the visibility of nests, and accessibility of nesting sites to surveyors. Gulls build their nests on a variety of substrates, and visibility can be reduced by nearby vegetation. Furthermore, as nests commonly occur atop buildings and other tall structures, they are often not visible from ground level. However, as urban structures may be complex (e.g. overhangs), birds may be well concealed not only from the ground, but also from vantage points or remote platforms, so any counts are likely to underestimate the true numbers. Buildings with a series of pitched roofs are particularly difficult to survey (Sellers and Shackleton 2011). Access to rooftops may be restricted or unsafe, limiting the feasibility of direct sampling methods. Additional challenges in estimating the size of the breeding population arise from the fact that residents may illegally remove nests from their property – up to 15% of nests are removed from urban areas by residents by the end of May (Calladine *et al.* 2006).

1.4. Previous surveys

Coastal breeding gulls and other seabirds in Britain and Ireland have previously been surveyed three times: Operation Seafarer in 1969–1970 (Cramp *et al.* 1974); Seabird Colony Register in 1985–1988 (Lloyd *et al.* 1991); and Seabird 2000 in 1998–2002 (Mitchell *et al.* 2004). Of these, only Seabird 2000 attempted a high level of coverage of urban areas, which were patchily covered in the previous surveys, and likely underestimated numbers (Coulson and Coulson 2015; Rock 2005). The counts of gulls in urban areas in Scotland during 1998-2002 for Seabird 2000 were made predominately from vantage points (78% of counts) and thus were likely to be underestimates; seventeen percent of counts were conducted by aerial survey, some with ground-truthing documented; and three percent of counts were conducted from the ground (Calladine *et al.* 2006). Two additional surveys of urban gulls were conducted in 1976 (Monaghan and Coulson 1977) and 1994 (Raven and Coulson 1997). A survey of large breeding gulls in Cumbria was conducted in 2009, including coastal, non-urban inland and urban colonies (Sellers and Shackleton 2011).

Surveys of winter gull roosts have been run each decade since 1953 and are of relevance to this report in terms of survey design. The most recent winter gull roost survey (WinGS) ran over three years from 2003/04-2005/06 (Banks *et al.* 2007; Burton *et al.* 2013). In the first year, the survey targeted birds at 484 key large roost sites (known from past surveys and county bird reports to hold >1000 roosting gulls), and in subsequent years a stratified sampling approach was used to estimate numbers at smaller sites. In the latter two winters, 701 inland 2x2 km tetrads and 933 stretches of coast were randomly selected. Inland sites were stratified according to three factors:

1) Winter gull density based on The Atlas of Wintering Birds in Britain and Ireland (Lack 1986) – 10 km grid squares were classified as low (0 -500 gulls), medium (501-3000) or high (>3000);
2) Freshwater cover data derived from CEH Landclass 2000 (Fuller *et al.* 2002) – classified as no water (0%), low water (>0%), <=5% or high water (>5%);
3) Proximity to the coast – any tetrads clipping a 1 km buffer around the coastline were classified as coastal; all others were classified as inland.

Fieldwork for WinGS was conducted primarily by volunteer surveyors, coordinated by regional organisers with extensive local knowledge of the birds and the landscape.

2. REVIEW OF METHODS FOR SURVEYING URBAN GULLS

2.1. Introduction

This chapter reviews the two main classes of survey: land-based methods and remote/aerial methods. Additional consideration is also given specifically to the cliff- or ledge-nesting Kittiwake.

There are five potential methods for surveying populations of breeding gulls listed in the Seabird Monitoring Programme (SMP) Seabird monitoring handbook for Britain and Ireland – vantage point counts, quadrat counts, transect counts, flush counts and aerial counts (Walsh *et al.* 1995). These methods are intended for general monitoring use, so are feasible to conduct on a large scale. We consider each of these in the context of the urban environment, plus the addition of distance sampling to the basic transect method (following Barbraud *et al.* 2014). Aerial methods are covered in greater detail than in the monitoring handbook, as the technology has developed substantially since the mid-90s. Seven techniques are reviewed: digital aerial surveys (DAS) and visual aerial surveys, unmanned aerial vehicles (UAVs), microlights, cameras on kites, aerial thermal imagery and satellite imagery. While many of these techniques are currently in use for wildlife surveying, satellite imagery is not currently available commercially at high enough resolution to identify birds, but is included as a potential future method. Due to their tendency to nest on the vertical sides of structures, methods for surveying Kittiwakes are considered separately.

For each method, we summarise the technique and technology involved, consider the benefits and problems, in terms of accuracy, efficacy, speed and practicalities (e.g. additional validation work required). We consider the repeatability of each method, which is important for ensuring data collected from future surveys are comparable and can thus identify trends. We also outline cost considerations.

The following census units are recommended by Walsh *et al*. (1995) and Gilbert *et al*. (1998):

- Apparently Occupied Nests (AONs): defined as a well-built nest capable of containing eggs, with at least one adult present. These include nests which are obscured but where sitting birds are visible (although in an urban setting some nests may be poorly constructed with just enough material to hold eggs in place (P. Rock pers. comm.);
- Poorly built ‘trace’ nests with adults in attendance, are likely to involve non-breeding birds. Trace nests may indicate a late breeding season, a decrease in the proportion of adults breeding, or more likely in an urban context, a failed first breeding attempt followed by a second (P. Rock pers. comm.);
- Assumed incubating birds;
- Apparently Occupied Territories (AOTs): based on the spacing of birds or pairs viewed from a vantage point, if actual nests or incubation cannot be discerned;
- Counts of individual birds of breeding age are recommended;
- If adults are not present, ‘active nests’ that contain eggs or show other signs of use may also be recorded (although it may be difficult to attribute these to a particular species).

The most robust census unit for estimating number of breeding pairs in an urban setting may be a combination of AON and AOTs, as there will always be a large proportion of AONs that are not visible from ground-based or aerial survey (P. Rock pers. comm.). In some urban areas, such as Bath, the complexity of the roofscape means that many nests are missed, even using multiple vantage points. Counting the birds of breeding age on rooftops allows unseen nests to be inferred.

The ability to differentiate between AONs, AOTs and poorly built 'trace' nests is taken into account when reviewing the suitability of the methods. Note that in urban environments, a small proportion of second year birds and most third year birds will breed, meaning that not all breeding birds will have full adult plumage.

The methods covered in the following section are:

1) Land-based methods (section 2.2), including: counts from vantage points, sample quadrat counts; transect counts (and distance sampling) and flush-counts of adults;
2) Aerial methods (section 2.3), including: digital aerial survey (DAS), visual aerial survey, unmanned aerial vehicles (UAVs), microlight, cameras on kites, aerial thermal imagery and satellite;
3) Survey methods for Kittiwake (section 2.4).

2.2. Land-based methods for surveying gulls

2.2.1. Vantage point counts

This method involves birds being observed from one or more vantage points, such as hilltops or buildings. The recommended census unit is apparently occupied nests (AONs), apparently active nests or apparently occupied territories (AOTs). If several counts are made, the highest is used as the population estimate but all counts are reported. If parts of the colony are obscured from view, minimum and maximum estimates for the missing parts are added to the total count. Where only part of a roof is visible, reasonable estimates can be achieved by dividing the observed AONs by the proportion of the roof that was visible (Sellers and Shackleton 2011), although this method assumes homogeneity in spacing of nests. Estimating the number of obscured nests based on the number of birds in attendance at the site has also used in previous surveys (Mudge and Ferns 1980).

Ideally it should be possible to observe individual nests for a period of time to differentiate between AONs, AOTs and trace nests; however, in practice time constraints may limit this process. When multiple counts are made from separate vantage points, there is also a risk of counting the same area twice. Careful mapping of the areas visible from each vantage point using a 1:5000 map, noting key landmarks or landscape features minimises this problem. There is a risk of double counting both members of a nesting pair sitting in close vicinity, (although this risk may be less than for other methods if extended observation time is possible). Another problem is that nests may be obscured by vegetation, especially later in the season.

This method is known to slightly underestimate numbers of large nesting gulls – with maximum detection rates estimated at 78% (Coulson and Coulson 2015); however, correction factors can be developed to account for detection bias. The detection rate is lower at highly industrial or commercial sites (Coulson and Coulson 2015). Using two observers or distance sampling can give an indication of nest detection probability (Koneff *et al.* 2008; Barbraud *et al.* 2014). As with other methods where rooftop access is necessary for surveying, health and safety is a major issue, and access permissions may be difficult to obtain from property owners. Vantage points used can be recorded using GPS and by taking digital photographs of the observable area to ensure counts are repeatable.

The main costs for this method are the cost of hiring fieldworkers (although there is potential for using volunteers) and the cost of cherry pickers to view/access rooftops (includes a driver, fieldworker and sometimes police to assist). The precise costs for a given urban area will depend on the availability and accessibility of suitable vantage points and the landscape topography and thus the time required. As a rough guide, one of the largest urban colonies in the UK and Ireland, Cardiff (which is ca. 7 km^2), was surveyed in six days in 2011, using a 13 m cherry picker for two days and 26 m cherry picker for one day (Rock 2011).

2.2.2. Sample quadrat counts

This method (based on Tasker *et al.* 1991) essentially involves taking a number of point counts, within a defined area around the point. The recommended census unit is an AON, i.e. a fully constructed nest containing eggs or chicks with signs of frequent use. The colony is mapped and overlaid with a grid. Quadrat points are randomly selected (suggested minimum sample size is 30; suggested quadrat size 300 m^2 for smaller gull species and 5,000 m^2 for larger gull species). Circular quadrats are recommended, as tethering a rope to the central point and walking around it provides a convenient method for determining the outer boundary of the quadrat. Quadrats are marked out, counting the number of active nests and clutch size. Nests are marked with a stake, visited several times during the laying period and the population for that quadrat is the maximum number of nests on any one date. Total colony size (total number of active nests) = (mean number of active nests per quadrat) x (total area of colony/area of quadrat).

The benefit of the quadrat method is that reasonably accurate nest counts are obtained within the sampled area, and that it is quicker/cheaper than full transects. In addition, reasonable estimates of the proportion of nests not visible on a rooftop can be achieved by dividing the number of AONs by the proportion of roof visible (Sellers and Shackleton 2011). However, this method is best applied to colonies that can be safely accessed by foot, so feasibility is limited in a large scale urban setting where the vast majority of birds will be roof nesting. Permissions would need to be gained from a large number of building owners, which would be both time-consuming and difficult. Also, there are logistical considerations in urban settings if the quadrat is larger than a single roof top. In addition, it involves considerable disturbance to the colony. The error associated with the total population estimate can be considerable if sampled quadrats are not representative of the whole colony – thus there may be a need for stratification. As with other methods where rooftops need to be accessed health and safety and gaining access permissions are major issues. Resurveying the same quadrats in future years allows direct comparison of counts within the sampled area; however, the method does not take into account changes in colony area, for which the colony needs to be re-mapped and new random quadrats sampled.

The costs associated with this method are the cost of hiring fieldworkers (potential for using volunteers) and the cost of cherry pickers to view/access rooftops (includes a driver, fieldworker and sometimes police to assist). Additional administrative costs involved with gaining permissions from building owners to access rooftops.

2.2.3. Transect counts

In this method (based on Wanless and Harris (1984), colonies are divided into areas based on landscape features. Areas are divided into strips no more than 10 m wide. Observers walk along strips, covering the entire area, counting and marking nests (e.g. with paint or bamboo canes). A second observer should recount a sample of the area by walking at right angles to the original counter. This **double observer method** enables nest detection probability to be determined (e.g. Koneff *et al.* 2008). The number of active nests = (number of active nests marked) x (total number of active nests on recount/number of marked nests on recount). The recommended census unit is active nests (defined slightly differently to AONs) – i.e. a fully constructed nest containing eggs and/or chicks (in or near the nest), or empty but judged capable of holding a clutch (Walsh *et al.* 1995). The basic transect method can be improved upon using **distance sampling** (e.g. Barbraud *et al.* 2014). This involves measuring the perpendicular distance of any detected nest to the transect line to correct for visibility bias, allowing detection probability to be estimated and nest densities to be corrected accordingly. This provides an alternative method for estimating nest detection probability.

This method is reasonably thorough and accurate – nest detection probability for large gulls on island breeding colonies estimated as 76% for a single observer – comparable to that achievable by vantage point counts – but is accuracy increases to 94% using the double observer method (Barbraud and Gélinaud 2005). More details on expected accuracy are provided in section 2.2.5. This method is best applied to colonies that can be safely accessed by foot, so applicability is limited in an urban setting where the vast majority of birds will be roof nesting. It is labour- and time-intensive. Several observers may be required to perform counts, and individual counters may have varied levels of skill or experience (Barbraud *et al.* 2014), and the number of observers can affect the counting efficiency (Harris and Calladine 1994) – this must be taken into account in the analysis. Considerable time may be required to count large colonies and this may create disturbance to birds. As with previously discussed methods, health and safety is a major issue for rooftop surveying, and access permissions may be difficult to obtain from property owners. The use of GPS and digital photos ensures the same areas/routes are covered on repeat survey. Resurveying the same transects in future years allows direct comparison of counts within the sampled area; however, the method does not take into account changes in colony area.

Costs for this method include hire of fieldworkers (potential for using volunteers) – use of distance sampling has been shown to reduce overall costs up to 87% by decreasing the number of observers required and the total time taken (Barbraud *et al.* 2014); also, the cost of cherry pickers to view/access rooftops (includes a driver, fieldworker and sometimes police to assist). Additional administrative costs involved with gaining permissions from building owners to access rooftops.

2.2.4. Flush counts of adults

This method (from Walsh *et al.* 1995) involves flushing adults using a horn or loud noise, standing in a prominent position overlooking the colony (e.g. from a cherry picker). Those gulls visible on the ground and in the air are counted. The count is repeated several times and the mean number is recorded. The census unit for this method is individual adult birds.
This method is good for rapid assessment of approximate colony size. However, count error is higher than for other land-based methods as quantifying large numbers of birds in flights is more difficult than methodically counting undisturbed nesting birds. In addition it may be difficult to relate the number of adult birds to the number of nests and to differentiate between species. In particular, large colonies may be very difficult to count in the air. This method creates unnecessary disturbance to the colony and there are significant health and safety considerations with observers on cherry pickers disturbing colonies. This method is the least repeatable land-based method, as non-breeding gulls may be present, and counts at large colonies are thus likely to be highly inaccurate.

Costs include hire of fieldworkers (potential for using volunteers) and cherry pickers to view/access rooftops (includes a driver, fieldworker and sometimes police to assist).

2.2.5. Accuracy of land-based counts

Few studies have compared accuracy of methods for nest detection in an urban setting. Coulson and Coulson (2015) compared the accuracy of vantage point counts with ‘street surveys’ (i.e. surveying rooftops from ground level). They showed that maximum nest detection rates were 78% from vantage point surveys, 48% from street surveys, and 88% when the two approaches were combined. However, this work was based on a relatively small number of colonies (seven colonies within six different urban areas), and detection probabilities may vary considerably depending on the character of the urban area (e.g. proportion of industrial area, hilliness or number of tall buildings providing suitable vantage points, *etc*.). The authors recommend the use of high vantage points or aerial survey to overcome the issues of poor detection from ground-based methods in an urban setting.

Surveys of the urban gull population of Gloucester were conducted independently by ground-based and aerial survey in 2002 (Rock 2002; Durham 2003). In this case, the aerial survey reported 3.4% fewer pairs (1,299 vs 1,345 pairs) than the ground-based survey, which was conducted using vantage points and cherry pickers (P. Rock pers. comm.).

Studies conducted in natural colonies give some indication of relative effectiveness of land-based-methods; however, detectability of nests is likely to differ between urban and natural colonies. The urban landscape is generally less open so vantage point counts may be obscured by buildings and other man-made structures, and nesting occurs on rooftops may not be visible from the ground. On the other hand, less natural vegetation in urban sites compared with natural sites may increase detection probabilities. Wanless and Harris (1984) estimated that the accuracy of a single transect count of nesting gulls in a natural colony is ±20%, whereas (Coulson *et al.* 1982) reported an accuracy of ±2% can be achieved by repeated counts. So the number of observers can affect the counting accuracy in transect counts. Counting accuracy of a single observer has been estimated at 86% of that achieved by a team of people systematically searching, recording and marking nests of all Herring and Lesser Black-backed Gulls on the Isle of May (Harris and Calladine 1994). Similarly, Barbraud and Gélinaud (2005) estimated that nest detection probability for a single observer following a transect methodology was 76%, whereas for two observers detection probability was 94%. The double observer method provides a means to estimate detectability so counts can be corrected to the value relating to 100% detectability. In summary, transects can deliver variable detection probability (within the range of 60–95%). Detection probability for vantage point surveys are within this range, so on average are no better or worse than transect in terms of accuracy, but create less disturbance to the colony.

The variability in detection rates arises due to factors such as breeding density, vegetation cover, topography and observer skill. Whereas surveys of nests on bare ground have a detection probability approaching 1, Barbraud and Gélinaud (2005) found that the greatest variation in detection probability was due to the observer, rather than vegetation height. Barbraud *et al.* (2014) found that the transect method underestimated numbers to a greater extent in colonies with higher breeding densities.

2.2.6. Correction for detectability

Given the huge variation that can account from detectability, as demonstrated in Table 1, including a method to account for detection probability in the survey design is advisable. Digital aerial surveys – as discussed in the following section – may not need any such correction if the gulls are nesting on open habitat with a clear sight line to the birds and their nests. Examples of methods for accounting for detectability include:

1) Using two observers (e.g. Koneff *et al.* 2008) – involves using a second observer to independently search for nests;
2) Distance sampling (e.g. Buckland 2001) – involves correcting for bias in visibility by measuring the perpendicular distance of each detected nest to the transect line. Thus this method creates less disturbance than comprehensive transect counts (Barbraud *et al.* 2014). It provides meaningful confidence intervals around the mean abundance estimate (Barbraud

et al. 2014). Distance sampling requires fewer observers per colony and reduces the amount of time required – e.g. Barbraud *et al.* (2014) estimated 87% less field effort compared with traditional transect, reducing the average number of observers required per colony from 6.5 to 1.4 and the average time taken per colony from 3 hours to 1.7 hours. This method has been used successfully to census breeding seabirds (e.g. Lawton *et al.* 2006; Robertson *et al.* 2008; Kirkwood *et al.* 2007);

3) Removal models (e.g. Farnsworth *et al.* 2002) – point counts are divided into at least three intervals of variable length, counts from the different time periods are used to estimate total detectability for the whole period.

Table 1. Gull nest detection probability estimated in previous studies.

Method	**Species**	**Location**	**Number of colonies surveyed**	**Nest detection probability (%)**	**Study**
Vantage point counts vs 'street surveys'	Herring Gulls, Lesser Black-backed Gulls	Urban areas in NE England and Scotland, including South Shields, Jarrow, Sunderland, Berwick-upon-Tweed, Durham and Dumfries	7	78 for vantage point, 48 for street surveys	Coulson and Coulson (2015)
Transect, marking nests, double-observer	Herring Gulls, Lesser Black-backed Gulls	Flat Holm island, south Wales	1	83.1 ± 3.3	Smith *et al.* (1981)
Transect, marking nests, double-observer	Herring Gulls, Lesser Black-backed Gulls, Great Black-backed	Offshore islands, Brittany, France	9	76.1 ±1.6 (single observer) 94.3 ±0.8 (two observers)	Barbraud and Gélinaud (2005)
Transect with team of counters marking nests	Herring Gulls, Lesser Black-backed Gulls	Isle of May, Scotland	1 colony (divided into 6 study plots)	80–95	Wanless and Harris (1984)
Strip transect vs distance sampling	Herring Gulls, Lesser Black-backed Gulls, Great Black-backed	Offshore islands, Brittany, France	10	61.4±1.5 (range 51.9 ± 6.4 to 70.6 ± 1.5) for transect method; 9–31% higher for distance sampling	Barbraud *et al.* (2014)
Marking nests, (transect?)	Terns and gulls	Massachusetts, USA	?	88±2.4 (range 78–96)	Erwin (1980); quoted in Barbraud and Gélinaud (2005)

Grey shading indicates study was conducted in an urban setting.

2.3. Aerial methods

2.3.1. Digital Aerial Survey (DAS)

Digital aerial surveys (DAS) have been used for monitoring seabird distribution and abundance at breeding colonies (e.g. Orford Ness Lesser Black-backed Gull colony (Figure 1), APEM pers. comm.) and offshore developments. At present, there are two main DAS techniques: High Resolution (HR) digital still imagery and High Definition (HD) digital video imagery. In both instances the imagery tends to be gathered from specially modified aircraft.

Figure 1. Example images of nesting Lesser Black-backed Gulls at the Orford Ness breeding colony. Copyright 2016 APEM Ltd.

DAS altitude is flexible and will always be made to adhere to the 'Standard European Rules of the Air' as laid down in regulation (EU) 923/2012 in section SEPA.5005(f), which states that: "Except when necessary for take-off or landing, or except by permission from the competent authority, a VFR (Visual Flight Rules) flight shall not be flown:

1) over the congested areas of cities, towns or settlements or over an open-air assembly of persons at a height less than 300 m (1,000 ft) above the highest obstacle within a radius of 600 m from the aircraft;
2) elsewhere than as specified in (1), at a height less than 150 m (500 ft) above the ground or water, or 150 m (500 ft) above the highest obstacle within a radius of 150 m (500 ft) from the aircraft."

Thus to comply with regulations for urban areas, surveys should always be flown at least 1,000 ft higher than the highest obstacle in the urban area being surveyed. The Civil Aviation Authority (CAA) requires that a minimum safe altitude of 500 ft (150 m) separation above hazards, including man-made structures, is attained in non-urban areas.

In this section, 'resolution' is referred to as 'resolution' for still imagery and 'definition' for video imagery. The resolution (the size of the individual pixels that make up the image) at which an image is gathered is affected by a range of factors but key are the focal length of the objective (lens) and distance of the camera to the object being photographed. A camera with a larger focal length objective allows an aircraft to fly higher to gather images at a certain resolution than the same camera with a smaller focal length objective.

The resolution at which digital imagery is acquired is very important in the context of urban gull and any other wildlife surveys as it affects the ability of an image analyst to identify the species recorded in the images gathered. For example, for a Common Gull measuring 40 cm in length and 10 cm in width, on average 16 pixels would cover the bird using 5 cm GSD (Ground Sample Distance) resolution imagery, whereas about 52 and 100 pixels would cover the same bird if the imagery were to be take and 3 cm or 2 cm GSD resolution, respectively. It is clearly much easier to get the detail required to identify a bird to species level from 52 or more pixels than 16. However, this is slightly simplistic. Key to accurate species identification is the gathering of images of a suitable resolution that are crisp and free of blurring; similar resolution images but with blurring may not be suitable for accurate species identification. In comparison with bespoke DAS of birds, DAS by Ordnance Survey are undertaken at a height of 8,000 ft, equating to a resolution of 15 cm – far too coarse to identify species or even to detect colonies (T. Dunn pers. comm.).

In the early days of DAS, aircraft vibration was problematic, affecting image quality, especially when compounded with the image blurring inherent in the 'Rolling Shutter' used by video. Normandeau Associates Inc. (2012) recorded motion blur created by the combination of the rolling shutter repeatedly scanning the sensor line by line and the constant movement of the aircraft.

Gyro-stabilisation and in-flight camera angle adjustability are thus important considerations if a HR or HD camera in a survey aircraft is to delivery imagery with little or no blurring (Normandeau Associates Inc. 2012). Gyro-stabilisation or an appropriately dampened camera mount significantly reduces vibrational effects, considerably improving image clarity and quality thus making it possible to deliver robust aerial HR wildlife imaging surveys. In-flight camera angle adjustability enables in-flight sun glare mitigation, where adjustable angle mounts roughly double the amount of low-glare daylight survey time compared to any fixed angle mount.

HR still images are further improved by helping to compensate for ground speed by using a combination of carefully selected shutter speed, ISO and aperture settings and Forward Motion compensation (FMC) technology (Coppack and Weidauer 2014).

Both HR still and HD video digital imagery can gather survey data very quickly, normally flying at between 1000–2000 ft (300–600 m). Following a transect approach an aerial survey by either method carried out at 120–190 knots (60–100 ms^{-1}) as recommended by Thaxter and Burton (2009) would take under three minutes to cover the whole of a 2 km x 2 km tetrad excluding turn time.

DAS design allows for survey repeatability as the exact location of each image captured along flight transects is known from pre-planned specific flight lines at a known altitude. The data collected from the DAS would also be comparable to various other methods such as vantage point counts as exact locations are known. Every bird and its precise location captured in an image are recorded. There may be occasions when a nest is partly hidden on a ledge or completely under an incubating adult gull and therefore not captured by the imagery. Field validation of a representative section of the survey area would ensure that all information is captured and any error in misidentified/unaccounted for nests can be estimated; however, even direct ground counts are prone to human error (Gilbert *et al.* 1998). If ground validation surveys are carried out it is important to minimise disturbance to the nesting birds (Walsh *et al.* 1995; Gilbert *et al.* 1998; Bakó *et al.* 2014), but as well-designed aerial surveys would gather very precise GPS coordinates of each nest it should be possible to plan these counts to minimise disturbance. Ideally, several validation surveys should be conducted during the breeding period as not all birds are necessarily in the colony at the same time (Bakó *et al.* 2014).

An advantage of DAS is that the image or video collected is a permanent record and can be revisited as required. Also, single birds can be given spatial co-ordinates within a GIS framework that allows further analysis beyond estimating numbers. Such analysis could include determining with precision fine-scale environmental factors that affect distribution and abundance over time.

High Resolution (HR) digital still imagery can be based on either a grid sampling design, whereby a series of independent images with a randomised starting point are collected throughout the study area, or imagery can be collected in complete transects of abutting images running parallel across the survey region. High Definition (HD) video imagery tends to be used for complete transects. Care must be taken to allow for spatial autocorrelation when using survey data to generate population estimates and associated confidence intervals (CIs). Spatial autocorrelation occurs when a sample in time and space is so similar to an adjacent sample that both samples are effectively one sample of the population being sampled. If two such samples are used to generate population estimates and CIs the intervals will tend to be incorrectly tight (precise) due to the same value having effectively been used twice – a phenomenon known as pseudo-replication. If a grid node or transect is taken as a single sample and there is no evidence of auto-correlation between the nodes or transects used to generate the population estimates and CIs these estimates should be accurate. However, especially with transect-based surveys where the number of transects is much smaller than the number of nodes in a grid survey, if transects are split into abutting sections, they should be tested for evidence of spatial autocorrelation to ensure independence of data points. In addition, it is possible to use a modelling approach, such as generalized estimating equations (GEE) to account for spatial autocorrelation.

The relatively small data files generated by HR stills allow the operator to view and review the imagery in flight at the moment of capture. This makes it possible for the operator to compensate in real time for any under or over exposure that could result from highly reflective surfaces such as roof tops and glass. Incorrect exposure leads to poorer quality imagery that makes species identification more difficult. Real time assessment of the imagery also allows preliminary in flight quality assurance and makes it possible for any survey line imagery that falls below a minimum standard to be flown again immediately.

Consideration should be given to the species identification rates of different methodologies. Gathering HR or HD imagery is particularly important for birds which are relatively small and for which there are closely allied species with similar plumages. Thaxter and Burton (2009) recommended the then widely used resolution of 5 cm for digital aerial imagery as a minimum for bird survey. Although HR still imagery can be gathered at almost any resolution depending on the focal length of the lens used, a resolution of 2 to 3 cm offers the best compromise in terms of species level identification, ground coverage and encounter rate (APEM pers. comm.). At a resolution of 2 cm GSD, HR stills would allow over 99% of the target gull species to be definitely[1] identified to species as sufficient detail is captured in the image to discriminate between confusion species. At a lower resolution of 3 cm GSD, HR stills should allow over 95% of large gull adults (Herring, Lesser-black Backed and Great Black-backed Gull), and over 80% of the two most difficult species, Black-headed Gull and Kittiwake to be definitely[1] identified to species (APEM pers. comm.). At 5 cm GSD, HR stills should allow over 80% of large gull adults to be identified to species. These percentage identification rates would not be affected by a study colony being of single or mixed species. Using 3 cm GSD stills over 90% of wintering Red-throated Divers were differentiated from

[1] Some aerial survey providers define identification rates as (definite + probable + possible) identification

the similar Black-throated Divers (Goodship *et al.* 2015), two species that are more difficult to separate than breeding gulls. The authors of this report do not have access to HD video percentage gull identification rates.

With 2 cm and 3 cm GSD HR stills it should be definitely and probably possible, respectively, to distinguish between an AON and a trace nest if the nests are not obscured by the sitting adult. Similarly, it should be possible to identify AOTs based on the spacing of individuals, although repeated survey would help confirm the assessment.

HR still pixel resolutions down to 0.5 cm GSD or lower can be achieved if required but such imagery tends to be quite expensive to gather if a large area needs to be covered.

Still imagery can be used to deliver oblique surveys that consist of the systematic gathering of high resolution digital photography of the exterior of objects, such as buildings or steep cliff faces / ledges that would not be visible from the vertical surveys. The imagery is collected using straight flight lines or by orbiting the area in question, and in a manner that the imagery can be viewed singularly or stitched together to form a montage. Such imagery is usually collected at an altitude of some 300 m and at a distance of 300 m from the area to be surveyed. Such imagery has been used successfully to survey cliff-nesting seabirds off Hawaii (Normandeau Associates Inc. and APEM Ltd, 2015) and could be useful to survey urban Kittiwake colonies on man-made cliff-like structures.

Digital aerial video survey involves capturing moving images along transects. Current methodology comprises positioning the cameras at an angle to the ground with objects visible for >0.5 seconds, and can be conducted at definitions between 0.5 cm and 5 cm; however, 2 cm has often been used to identify seabird species (Thaxter and Burton, 2009). The video imagery is manually reviewed offline and only birds that cross the horizontal image centreline are counted.

The advantages of digital aerial video survey include surveys being conducted along pre-defined transects which can be re-flown for repeated surveys, and providing permanent records of the exact locations of the surveyed species on imagery that can be revisited. The method also allows rapid survey of large areas.

2.3.2. Visual aerial survey

Visual aerial survey is a well-established technique where observers in a small aircraft record target wildlife, often flying along transects at some 250 ft (80 m) altitude (Camphuysen *et al.* 2004). While visual aerial surveys are useful for surveying large, remote areas in a short period of time, the low altitude generates large-scale disturbance amongst birds. Camphuysen *et al.* (2004) and Buckland *et al.* (2001) identified several disadvantages of using this method for capturing data, including:

- Safety concerns associated with the use of low-flying aircraft.
- Observer bias, particularly when observers are ‘swamped’ by large numbers of birds and unable to accurately record numbers.
- The possible disturbing effect of low-flying aircraft on the distribution and double counting of birds.
- The lack of a permanent observation record. Although records are recorded on dictaphone and transferred to databases following a survey, unlike DAS or satellite imagery no exact location or time can be applied to a particular bird.
- Visual methods cannot be validated after the event to assess reliability of counts and species identity.

Aerial visual is not a suitable technique for the proposed surveys of breeding urban gulls as it is a Civil Aviation Authority requirement that aircraft fly at least 500 ft (150m) above possible hazards. This invalidates the use of aerial visual surveys over towns.

2.3.3. Unmanned aerial vehicles (UAVs)

Capabilities of UAVs vary from small joystick controlled units within a range of a few hundred metres to high-altitude UAVs used for military applications that have ranges of 1,000s of km and can fly at 50,000 ft (15,000 m) above sea level (Koski *et al.* 2010). Considerations of the terrain in which the UAV will be used should be made; multi-rotor UAVs can be launched from a platform due to their vertical landing and take-off ability whereas fixed wing UAVs need open space for landing and take-off. Ratcliffe *et al.* (2015) showed that multi-rotor UAVs can be used for surveys over short durations and range, while larger fixed-wing UAVs have potential to survey large areas in a single mission (Hodgson *et al.* 2013). UAV imaging systems provide an immediate snapshot of sightings and accurate GPS location of each image (Hodgson *et al.* 2013). The geo-referenced images allowed precise mapping of species distribution. UAV technology can and has been used to map distributions of nesting birds (Ratcliffe *et al.* 2015; Chabot *et al.* 2015). The survey equipment is cheaper to use than a low flying manned aircraft, and being quiet it has been suggested that it is probably less disturbing to wildlife. Vas and Lescroël (2015) studied the effect of disturbance of fixed wing UAVs on three species of bird (Mallard *Anas platyrhynchos*, Greater Flamingo *Phoenicopterus roseus* and Common Greenshank *Tringa nebularia*). They demonstrated that in 80% of all cases one specific drone type could fly to within 4 m of the birds without visibly modifying their behaviour. Approach speed, drone colour and repeated approaches did not appear to have any significant impact on bird reaction; however, approach angles had marked impacts across all three species. A fixed wing Phantom drone approaching a bird vertically was usually more disturbing, maybe because it was associated with a predator attack. However there is much visual evidence available on the web that rotary drones cause sufficient disturbance to lead to them being attacked by hawks, geese, ravens, gulls and various unidentified birds[2].

UAV technology can be used for repeated surveys at a small and large scale to map species such as seabirds. Chabot *et al.* (2015) used a UAV to conduct a census of Common Terns *Sterna hirundo* at a colony. Imagery of approximately 3 cm resolution was obtained; however, tern counts ranged from 91–98% of nest counts performed on the ground. Not all terns were detected in the aerial imagery due to difficulties in detecting terns against poor quality background (e.g. dead vegetation), poor lighting and in blurrier proportions of the imagery. Ground counts also had challenges as not all nests on the ground were assumed to be active, and some may have been left unattended.

Issues with aerial UAV imagery such as poor contrast between birds and their background, visibility due to lighting conditions and blurred images may affect the detection and identification rates of seabirds. Imagery of gulls on an estuary using a modern off-the-shelf fixed wing UAV was at best of moderate quality (APEM pers. comm.). Higher resolutions could be achieved as UAVs have the ability to fly lower than manned aircraft. The automated flight control technology of such UAVs also makes them easy to fly.

A major limitation with the use of UAVs in the UK is due to the restrictions in operation enforced by the Civil Aviation Authority. UAVs cannot be operated more the 500 m from where the operator is

[2] E.g. gull/skua: www.youtube.com/watch?v=DzfiLmbhvqg;
golden eagle: http://www.mirror.co.uk/news/weird-news/golden-eagle-vs-drone-incredible-5050720
hawk: www.youtube.com/watch?v=smv7cBzg-Ok

positioned and must remain within 'line of sight' of the operator at all times. This can make the use of UAVs inefficient when trying to survey over large areas. They should also only fly within 50 m of people and vehicles with the permission of the people, the vehicle owners and the owners of the land. The regulations for UAVs are changing very fast and it is difficult to predict what future relevant regulations will exist for them.

2.3.4. Microlight

Microlight aircraft offer low survey speeds for wildlife surveys to increase identification rates (e.g. Murn *et al.* 2002); however, they can create higher levels of noise in comparison to UAVs and the combination of low speed and noise can be highly disruptive (Rehfisch and Michel, in press). There is also a considerable health and safety concern with flying microlights in the vicinity of disturbed large birds. In 2014, a microlight craft was badly damaged by a bird strike (thought to be a gull) and forced to make an emergency landing (Air Accidents Investigation Branch 2014). While microlights may provide a cost-effective alternative to fixed-wing aerial survey, there is a considerable safety risk to flying over an urban area where birds are likely to be disturbed, so increasing the risk of collision, and where an emergency landing would be difficult.

2.3.5. Cameras on kites

Rehfisch and Michel (in press) report that Fraser *et al.* (1999) conducted a census of Adélie Penguin *Pygoscelis adeliae* colonies using a kite equipped with a remote-controlled camera. Two personnel are required to control and launch the kite. The trials suggested that when using a kite as a census tool wind patterns must be well understood. The kite remained stable and the camera produced relatively sharp photographs (GSD not provided) in wind speeds up to 50 km/h. Reactions from penguins to the presence of a kite overhead varied with the height of the kite, but the kite was generally ignored at altitudes in excess of 50 m. A kite camera has also been used to successfully survey common hippopotamus *Hippopotamus amphibius* in small wetlands (Rehfisch and Michel in press). A disadvantage to using a camera mounted on a kite is that the camera direction will be wind-dependent rather than operator controlled. In most circumstances this method is unlikely to be able to obtain accurate and repeatable counts of nesting gulls. The method would take a long time to cover large areas and is more suited to localised colonies such as those studied by Fraser *et al*. (1999).

2.3.6. Aerial thermal imagery

Thermal imagery has been used to distinguish between seal species (APEM pers. comm.). Kinzel *et al.* (2006) used aerial thermal imaging to evaluate annual variation in roost locations and spatial densities of Sandhill Cranes *Grus canadensis*. Surveys were conducted at night using manned aircraft. Infrared video images were displayed to the camera operator in real-time so the operator could optimize the image contrast, whilst simultaneously being archived. Information on time and exact location were also recorded. Even when flying at relatively low altitudes it is difficult to distinguish from thermal video imaging between bird species based solely on their thermal signatures. Detection of birds may only be possible if they are nesting on highly emissive backgrounds (which would include most surfaces found in urban environments); however, species identification may still prove difficult.

Gillette and Coates (2013) used cooled mid-wave infrared cameras to successfully survey Sage Grouse *Centrocercus urophasianus* leks; however in the study sites there were no confusion species and thus the images did not have to be of a very high resolution. Garner *et al.* (1995) trialled the use of a thermal infrared scanner for wildlife censuses or estimation of wildlife in urban areas and parks. A thermal infrared system uses a detector, a thermal imager and a real-time recording device and requires low airspeeds. Garner *et al*. (1995) provided recommendations for using thermal infrared methods for studying wildlife, these include:

- Scanning should be limited to times that provide the highest thermal contrast and lowest thermal loading e.g. flying in overcast days, early morning or late at night.
- Several images of the object of interest should be analysed prior to and during the survey to generate an approximate range of delta-*ts* (temperatures).
- Ground-truthing should be conducted to verify the accuracy of the images.
- Completely concealed objects behind foliage, etc. will not be detected.

Costs of flying a digital aerial thermal survey would be broadly similar to those of an digital aerial survey.

The urban environment with its wide range of emissive surfaces is unlikely to be suitable for this technique. The technique can be almost completely dismissed for use to survey breeding urban gulls due to the 500 foot flight height restriction of the Civil Aviation Authority.

2.3.7. Satellite

At the time of writing, commercially available civilian satellites are not capable of collecting imagery at high enough resolution to identify individual nesting gulls.

As technology advances, finer spatial resolutions of optical, multispectral data observing in the Red, Green, Blue and Near Infrared spectrum (RGBN) are becoming commercially available such as Digital Globe's WorldView-3 which should become available at 25 cm resolution in 2016. The military have had sub-centimetre space-borne sensors for some time; due to its sensitive nature this information is not publicly available with future restraints on spatial resolutions subject to political influences. At the time of writing, the RGB data with the finest spatial resolution commercially available is Digital Globe's WorldView-3 that launched imagery at a resolution of up to 31 cm in August 2014. The highest resolution currently available from optical imagery satellites is given Table 2. In due course it should become possible to identify birds to species in urban areas from satellite imagery but it is difficult to determine when. Though commercially available space-borne imagery remains too coarse to reach down to the species level, or count birds at the time of writing, it can be useful in alternative ways as described below.

Habitat classification

Many studies have used remote sensing technologies for conservation and habitat classification. Methods classifying biophysical parameters as a proxy to estimate spatio-temporal changes in the distribution of abiotic conditions, and the distribution, structure and composition of functioning ecosystems on an absence/presence basis are widely utilised (Pettorelli *et al.* 2014). However, this is a qualitative approach, rather than a direct measure (Kuenzer *et al.* 2014) due to the limitations associated with coarse imagery. In isolation, using satellite data do not provide a viable way to estimate populations, but combined with traditional ground surveys can be an invaluable tool to help target land based surveys. Satellite derived data can be used to identify areas in which urban gulls are most likely to be nesting, and could potentially identify colonies or large groups using object based classification methods (documented to have up to 82.8% accuracy) (Hamedianfara and Shafriab 2015) and by visual identification respectively. Urban environments remain extremely difficult to map. Traditional habitat classifications performed over relatively homogenous areas normally work well as an occupation indicator, but when faced with scenes of a high heterogeneity these become complex to the point of redundancy. Issues of shadow cover, high buildings, a vast range of surfaces that exhibit different spectral characteristics, and high levels of suspended atmospheric particulates all serve to create a chronic problem of mixed pixels, and unrepresentative classifications.

Classification of probable nesting sites using 3D data

High resolution 3D models of the urban area from aerial surveys, or an aerial LIDAR survey derived 3D models could be used to classify areas most suitable for nesting habitats, utilising roof slope as a proxy. It is known that flatter roofs are preferred nesting environments (Raven and Coulson 1997), and obstacles such as chimney stacks would provide shelter and therefore be classed as a hot spot. The literature also documents using algorithms to sharpen satellite derived thermal data to a suitable resolution over urban areas (Feng *et al.* 2015) so coupled together with 3D models and RGB data it is feasible that in the future satellite derived thermal imagery may also prove useful to bird surveys aimed at identifying colonies or flocks over urban areas. When coupled with land based surveys, this could serve to increase the correct identification of different species, the ground data also validating and building on what has been derived from satellite imagery.

Phenological studies as a proxy for migratory patterns

The temporal resolution of satellite imagery makes it invaluable in terms of quantifiable phenological metrics as an indicator of times when certain species of gulls may be more likely to settle in urban areas. Should the resolution become sufficiently fine, this can aid continuous studies over time – helping to monitor nesting, migratory and feeding habits during the daytime. In the case that the resolution becomes sufficiently fine to capture the presence of these birds, each scene can cover ≥12 km in one image, on a daily basis. The cost benefit compared to land and aerial surveys would be much improved.

The regular temporal resolution of satellite imagery would be ideal for utilising habitat surveys as a proxy for nesting habitats. Most of the highest resolution sun-synchronous satellites can complete a full orbit on a daily basis, operating throughout the year, producing at least one cloud free image per month. Sun-synchronous orbits have similar overpass footprints at the same time each day so the data are directly comparable should a different method be used. It is also possible for customers to task the satellite directly, allowing for flexible data capture.

Future directions

It would be possible to identify birds down to the species level using satellite derived data over a complex environment, such as within an urban context, by using hyperspectral data if it were available in sufficient resolution. Hyperspectral technologies are limited to aerial platforms, requiring large budgets, and in depth end user processing knowledge, with future missions plagued by uncertainty. Future potential missions are ASI's PRISMA (Kramer 2015) planned for 2017, DLR's 18 m resolution EnMAP planned for to be operational by about 2020 (EnMAP 2015) and HyspIRI (NASA JPL 2015) also incorporating a thermal band better than any available at the moment - which is looking at a possible, yet unconfirmed launch in 2022. In the future, this may provide the solution to bird surveys over the UK and further afield.

2.4. Privacy issues

For any survey method capturing photographs or videos within residential areas, there are potential issues with privacy. The Information Commissioner's Office, the UK's independent authority responsible for the enforcement of the Data Protection Act 1998 and Freedom of Information, has published guidance relating to image captures for surveillance purposes and includes a section on unmanned aerial systems (UAS) or any other aerial vehicle with an attached camera (Information Commissioner's Office 2015). This report states that use of UAS for non-domestic or commercial purposes needs to comply with data protection obligations. This may involve performing a 'privacy impact assessment' – a tool for identifying the most effective way to comply with their data protection obligations and meet expectations of privacy (Information Commissioner's Office 2014). One major issue is that it is likely that individuals may be recorded without their knowledge or consent, and information regarding the surveys should thus be provided. For example, this might involve placing signage in the area in which the survey is occurring explaining the purpose of the survey with some form of privacy notice and a link to a website where further information can be obtained.

Table 2. Table to compare specifications of the highest resolution satellite derived imagery at the time of writing.

Sensor	Spatial Resolution (m at nadir)	Temporal Resolution (days by metres GSD)	Radiometric Resolution (comparability)	Accuracy (without GCPs)	Swath width (km)	*Approximate* price per km^2
Worldview-3	Pan = 0.31 MS = 1.24 SWIR = 3.7 CAVIS = 30 m	1.24 = <1 0.31 = 4.5	8 band MS + pan 30 m CAVIS for atmospheric correction and detection.	3 m	13.1	£10 - £20
Worldview-2	Pan = 0.46 MS = 1.84	0.52 = 3.7 1 = 1.1	8 band MS + pan	3.5 m	16.4	£8 - £10
Geoeye-1	Pan = 0.46 MS = 1.84	1	4 band MS+ pan	3 m	15.2	£8 - £10
Worldview – 4 (2016)	Pan = 0.30 MS = 1	3 (10.30am)	4 band + pan	3-4 m	-	£8 - £10
Pleiades– 1B	Pan = 0.50 MS = 2	1	4 band MS + pan	3 m	20	£6 - £8
KOMPSAT-3A	Pan = 0.55 MS = 2.2 IR = 5.5	1	5 band MS + pan	?	12	£4 - £6

*MS = Multi-spectral, Pan = Panchromatic, SWIR = Short Wave Infrared, IR = Infrared, CAVIS = Clouds, aerosols, vapours, ice and Snow. Grey shading indicates a future mission.

2.5. Survey methods for Kittiwake

While Kittiwakes generally breed on coastal cliffs, they are also known to nest in urban areas, for example in Newcastle (Turner 2010). As they tend to nest on the vertical sides of structures, ground-based surveys, vantage point surveys, oblique imagery surveys or specially adapted remote survey equipment offer the best opportunities for survey.

2.5.1. Vantage point method for Kittiwake

The vantage point method for Kittiwake (based on Heubeck *et al.* 1986) differs slightly to that for other gull species. The recommended census unit is AONs. Separate counts of empty nests, unattended nests with eggs or dead chicks, occupied trace nests and adults are also recommended. The boundaries of a colony are defined and subdivide based on natural features using a 1:10,000 Ordnance Survey map. Any Kittiwake roosts on cliffs/buildings are mapped, as these can develop into colonies. On densely populated individual cliffs/buildings, the count is subdivided using obvious features to avoid missing or double counting birds (photographs or rough sketches may be useful).

Each individual cliff/building is counted twice to ensure accuracy. The maximum and minimum number of hidden AONs in concealed sections is estimated. The highest reliable count of all AONs in the colony should be reported, rather than sum of peak counts from individual subsections.

The benefits of this method are that it is possible to observe individual nests for a period of time, it is generally possible to differentiate between AONs, AOTs and trace nests. However, high densities of nests, often positioned haphazardly over large areas increases the risk of double counting or overlooking nests. In addition, colonies may shift locations, so changes in the population of a sub-colony may not reflect the wider population status. Provided the vantage points used are recorded and the surveyed area is carefully mapped, counts are repeatable. As with previously reported methods, using two observers can give indication of nest detection probability (Koneff *et al.* 2008).

The costs involved with this method include hiring fieldworkers (potential for using volunteers) and hiring cherry pickers to view/access high vantage points. The precise costs for a given urban area will depend on the availability of suitable vantage points and the landscape topography and thus the time required.

2.5.2. Remote survey methods for Kittiwake

The remote survey methods listed above generally apply to Kittiwakes also, but as Kittiwakes tend to nest on man-made structures resembling cliff faces, partly or fully hidden from above, the great control affordable by UAVs could prove beneficial for this species.

2.6. Species-specific considerations

Differentiating the number of active nests of Herring and Lesser Black-backed Gull, and to some extent Black-headed and Common Gull, may be difficult. For methods that involve counting the number of active nests, estimating the numbers of nests of each species is generally undertaken by determining the ratio of adult gulls of each species and assuming that the proportion of nests is equal to the proportion of adults. However, this makes the assumption that the nest production rate is equal between species. The validity of this assumption for gulls is often unknown. It may also be problematic to obtain accurate counts of the species proportions of adult birds in mixed-species colonies where the species are spatially clumped. Incomplete coverage may lead to inaccurate species ratios. In addition, slight differences in the laying periods of different species can lead to inaccuracies (Wanless and Harris 1984).

While incubating gulls should be easily discernible by direct or remote methods, other occupied nests may not be. Herring and Lesser Black-backed Gull chicks cannot be reliably separated in the field, although well-feathered chicks are identifiable in the hand, and fledged young are identifiable in the field with experience (Walsh *et al.* 1995). Great Black-backed Gull chicks are generally discernible due to their size and heavier bills (Walsh *et al.* 1995).

2.7. Timing of counts

One issue with counting gulls is that they breed over an extended period, so single counts may miss a large proportion of breeding attempts. However, the bulk of the population will be in incubation late May to early June, although earlier breeding may occur in some urban locations (Walsh *et al.* 1995, Rock 2005). Table 3 shows the timing of breeding for our six species of interest. Recommended count timing for gulls is May – late June (i.e. early incubation to early fledging) (Walsh *et al.* 1995). Kittiwake is late May to mid-June, or a single count in mid-June. Note, for Kittiwakes, if the breeding season is late, i.e. a large number of trace nests occur in June, a further count in late June is recommended (Walsh *et al.* 1995).

A study of nesting Herring and Lesser Black-backed Gulls on the Isle of May suggests that counts should be made as late in the season as possible (Wanless and Harris 1984). Counts conducted prior to the end of May significantly underestimated population size, and in the 20 days prior to this the number of clutches increased by 12% every three days (Wanless and Harris 1984). Conversely, counting too late in the season may also lead to underestimates of population size because failed nests might be missed, and chicks over a week old may be difficult to see as they leave the nest and hide. One further complication is that late in the season, Lesser Black-backed Gulls can lay eggs in apparently 'incomplete' nests – i.e. shallow, unlined depressions, as opposed to well-formed cups lined with material (Wanless and Harris 1984); however, including incomplete nests in the count leads to population estimates greater than the true numbers (Wanless and Harris 1984). However, it is thought that timing of counts might be less crucial within an urban context compared with colonies in natural habitats (Calladine *et al.* 2006). As breeding success is generally higher at urban sites, the bias due to failed nests will be less than at natural sites (Calladine *et al.* 2006). Counting nests later in the season also increases the likelihood that some nests may be removed or destroyed, therefore counts should take place from May to late June.

Table 3. Timing of breeding for gulls (Cramp and Simmons 1983; Gilbert *et al.* 1998; Furness 2015). *BDMPS=biologically defined minimum population scales (Furness 2015).

Species	BDMPS* migration-free breeding season	Laying date	Incubation period (days)	Fledging period (days)
Herring Gull	May–July	From late April	26–32 (usually 28–30)	35–40
Lesser Black-backed Gull	May–July	From mid-May	24 – 27	30–40
Common Gull	–	Late May and June	22–28 (usually 24–27)	~35
Black-headed Gull	–		23–26	~35
Great Black-backed Gull	May–July	From May	27–28	49–56
Kittiwake	May–July	Late May	25–32	43

2.8. Validating of and correcting for vantage-point or remote counts

In colonies that are accessible by foot, the most comprehensive methods for assessing the number of AONs are walking transects across the entire area, or performing counts of randomly sampled quadrats. However, in an urban setting, where the majority of nests occur on rooftops, these methods are not feasible, so vantage point counts are the most viable option for ground-based methods. Coulson and Coulson (2015) compared vantage point and street survey methods in six urban conurbations to compare detection efficiencies for Herring and Lesser Black-backed Gulls. Large gull numbers were underestimated from these surveys and so there was a need for a correction factor to be applied, established from cherry-pickers or aerial surveillance methods. Walsh *et al.* (1995) provide a method for correcting vantage-point counts, which has been used on Skomer and could potentially be amended for use in an urban setting:

- Count incubating birds and pairs within a sub-colony (colony can be divided into areas of different habitat e.g. grass, flat roofs, etc.) from a suitable vantage point;
- Count the same sub-colony using walking transect and marking nests*;

- Use the ratio of these counts;
- Repeat over several sub-colonies to produce an average count ratio that can be used to estimate the proportion of nests missed by vantage point counting.

*In an urban setting this is constrained by choosing an area where roofs can be accessed. And care must be taken to ensure that the sub-colonies counted are representative of the wider colony. In addition, it will be extremely time consuming and access to sites may be limited on health and safety grounds.

2.9. Ranking of methods and conclusions

Based on the information presented in the preceding sections, we have scored each method according to the standardised rankings provided in Table 4. The various methods are scored according to the criteria presented for large gulls (Herring, Lesser Black-backed and Great Black-back) in Table 5, small gulls (Common Gull, Black-headed Gull) in Table 6 and Kittiwake in Table 7.

Table 4. Rating definitions applied to each survey method.

Rating	Disturbance	Resolution	Detectability	Ability to differentiate between species*	Ability to differentiate between breeding and non-breeding birds*	Efficacy (time taken and cost per tetrad)	Repeatability
1	None	0–2 cm	80–100%	80–100%	80–100%	Least expensive / most efficient	Yes/No
2	Possibility that bird is aware of survey.	2–5 cm	60–80%	60–80%	60–80%		
3	Bird is aware of survey.	5–25 cm	40–60%	40–60%	40–60%		
4	Bird likely to react to survey.	25–100 cm	20–40%	20–40%	20–40%		
5	Bird reacts to survey with risk to eggs and young.	>100 cm	0–20%	0–20%	0–20%	Most expensive / least efficient	

* Note, these are estimated range categories

Generally, land-based methods create more disturbance to birds than aerial methods and are likely to be more labour- and time-intensive than aerial methods. However, land-based methods may better enable differentiation between species and between breeding and non-breeding birds.

The most suitable remote methods from a purely technological viewpoint are likely to be digital aerial stills, video and UAV. These methods are all repeatable for future surveys and can be used in comparison to other methods such as vantage point surveys for validation. Data are captured at the exact time of survey and can be easily stored and revisited in the future. Lesser Black-backed gulls measure approximately 48–56 cm, Herring Gulls 54–60 cm, Great Black-blacked Gulls 61–74 cm, Common Gulls 40–46 cm, Black-headed Gulls 35–39 cm, and Kittiwakes 37–42 cm (Svensson 2010) – all large enough to be identified using the resolution that these methods can provide.

HR still and HD video digital aerial surveys have not been shown to cause any detectable disturbance to birds. Drones cause disturbance and there are many examples of them being attacked by hawks, geese and gulls. The two aircraft-based survey methods can cover large areas very quickly and the flight restrictions over towns do not affect their delivery. Drones must only fly within 50 m of people or vehicles with permission from the people and/or land owner and can only be flown up to 500 m in line of sight of the operator.

Where urban Kittiwakes nest on man-made structures similar to cliffs (i.e. nests partly or fully hidden from above), UAVs could be particularly beneficial, due to the manoeuvrability and capability to move up and down next to a vertical surface. This would only be feasible on a relatively small scale due to CAA restrictions (e.g. UAV must remain in the line of sight of the operator), but could be used for targeted survey of known colonies. Oblique digital still imagery can also be used to survey cliff-nesting seabirds (Normandeau and APEM 2015).

Table 5. Method ratings for the identification of large gulls (see Table 4 for a definition of ratings).

Large gulls	**Vantage point**	**Sample quadrat**	**Transect**	**Flush counts**	**Digital aerial survey**	**Aerial visual**	**Aerial thermal**	**UAV**	**Satellite**
Disturbance	1	5	5	5	1	4	1	1–5*	1
Resolution	NA	NA	NA	NA	1–2	–	3–4	2	3–4
Detectability	?	?	1	?	1–2	2	3	1–2	–
Ability to differentiate between species	1	1	1	1	1	3	5	1–2	4
Ability to differentiate between breeding and non-breeding birds	1–2	1–2	1–2	5	1–2	2	5	1–2	5
Efficacy	3	3	5	3	1	3	4	3	2
Repeatability of methods	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes
Suitability for breeding urban gull survey	Yes	?	Yes?	?	Yes	No	No	?	No

* Dependent upon the type of drone. For rotocopter drones, values of 3–5 is more likely at a realistic survey height; for fixed-wing drones, 1–2 is probable (APEM, pers. comm).

Table 6. Method ratings for the identification of small gulls (see Table 4 for a definition of ratings).

Small gulls	**Vantage point**	**Sample quadrat**	**Transect**	**Flush counts**	**Digital aerial survey**	**Aerial visual**	**Aerial thermal**	**UAV**	**Satellite**
Disturbance	1	5	5	5	1	4	1	1–5*	1
Resolution	NA	NA	NA	NA	1–2	–	3–4	2	3–4
Detectability	?	?	1	?	1	3	3	1–2	-
Ability to differentiate between species	1	1	1	1	1–2	4	5	2–3	5
Ability to differentiate between breeding and non-breeding birds	1–2	1–2	1–2	5	2	3	5	2–3	5
Efficacy	3	3	5	3	1	3	4	3	2
Repeatability of methods	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes
Suitability for breeding urban gull survey	Yes	?	?	?	Yes	No	No	?	No

* Dependent upon the type of drone. For rotocopter drones, values of 3–5 is more likely at a realistic survey height; for fixed-wing drones, 1–2 is probable (APEM pers. comm).

Table 7. Method ratings for the identification of Kittiwakes (see Table 4 for a definition of ratings).

Kittiwake	**Vantage point**	**Sample quadrat**	**Transect**	**Flush counts**	**Digital aerial survey**	**Aerial visual**	**Aerial thermal**	**UAV**	**Satellite**
Disturbance	1	5	5	5	1	4	1	1–5*	1
Resolution	NA	NA	NA	NA	1–2	–	3–4	2	3–4
Detectability	?	?	1	?	1–3†	3	3	1–2	–
Ability to differentiate between species	1	1	1	1	1–2	4	5	2–3	5
Ability to differentiate between breeding and non-breeding birds	1–2	1–2	1–2	5	2	3	5	2–3	5
Efficacy	3	3	5	3	1	3	4	3	2
Repeatability of methods	Yes	Yes	Yes	No	Yes	Yes	Yes	Yes	Yes
Suitability for breeding urban gull survey	Yes	?	?	?	Yes	No	No	Perhaps	No

* Dependent upon the type of drone. For rotocopter drones, values of 3–5 is more likely at a realistic survey height; for fixed-wing drones, 1–2 is probable (APEM pers. comm).

† Probably only applicable to data collection using oblique survey techniques and HR digital stills

3. CURRENT STATE OF KNOWLEDGE OF GULL DISTRIBUTION IN BRITAIN AND IRELAND

3.1. Introduction

In order to develop a survey design and an estimate of the survey effort required to deliver a census of urban gull populations in the UK and Ireland, we first need to fully review the current breeding distribution of gulls. In this chapter, we provide a detailed description of gull breeding distributions within the UK and Ireland, and where they occur in relation to urban areas, using a number of data sources and tools.

3.2 Methods

3.2.1 Gull distribution and abundance

Here we provide information on the distribution and abundance of breeding gulls within Britain and Ireland, focussing on current data. Gulls are known to associate with urban areas and there has been an acknowledged expansion of urban populations in recent years (Nager and O'Hanlon 2016, Rock 2005). Recent work has shown Herring Gull population growth rates are higher in urban areas, but a similar trend was not identified for Lesser Black-backed Gulls (Nager pers. comm. 2016). While historical surveys can provide a useful baseline for assessing changes in gull abundance, here we focus on data collected during and since Seabird 2000 (Mitchell *et al.* 2004).

To describe current breeding gull distributions we use data from Bird Atlas 2007–11 (Balmer *et al.* 2013). Data on the breeding distributions of gulls from Seabird 2000 (Mitchell *et al.* 2004) were also obtained from the JNCC Seabird Monitoring Programme database (these data differentiate substrates on which gulls are nesting, enabling urban roof-nesting birds to be identified).

Additional information was gleaned from BirdTrack (http://www.birdtrack.net; organised by BTO in partnership with Royal Society for the Protection of Birds, BirdWatch Ireland, Scottish Ornithologists' Club and Welsh Ornithological Society) records for the period between 2012 and 2015, for which there is an option for observers to enter breeding status against each individual species record. The 'pinpoint sighting' feature within BirdTrack also allows observers to enter 6-figure grid references identifying precise locations of nests. BirdTrack data provides a more up-to-date record of urban gull breeding sites than the Bird Atlas, aiding identification of sites that have been colonised by breeding gulls since the period of the Bird Atlas. Only 'confirmed' or 'probable' breeding records from BirdTrack were included. We excluded 'probable' breeding records where criteria used were 'permanent territory' or 'pair in suitable habitat', as these were not considered reliable enough evidence for breeding in gulls.

3.2.2 Urban land cover in the UK

In order to identify 'urban areas' that would require coverage in a census of urban gull populations we used data from Land Cover Map (LCM) 2007 (http://www.ceh.ac.uk/services/land-cover-map-2007; Morton *et al.* 2011), which provides data for the UK on classes of land cover at 1 km^2 resolution. Urban cover was summarised for 14 regions of Britain and Northern Ireland (these following those used for the 2003/04-2005/06 Winter Gulls Roost Survey (Burton *et al.* 2013), with Wales being subdivided into north and south).

3.2.3 Concurrence of breeding gulls and urban areas

The relationship between % urban cover and the likelihood of confirmed, probable or possible breeding gulls being present during Bird Atlas 2007–11 was examined in R using logistic regression. For this analysis, 'coastal' sites were excluded, by excluding any 10 km square that occurred within a 1 km buffer of the coastline, including estuaries. This was to allow us to better understand the relationship between urban habitat and gull numbers, removing the confounding effects of the coast, as many coastal gulls occur in non-urban colonies.

3.3 Results

3.3.1 Gull distribution data from Bird Atlas 2007-11 and from BirdTrack for 2012-15

Figure 2 shows the distribution of breeding gulls within the UK and Ireland at 10 km resolution, according to data collected for Bird Atlas 2007–11 (Balmer *et al.* 2013). Although records in Bird Atlas 2007–11 were presented at the 10-km scale, there are considerable underlying data available at the tetrad (2-km square) level. The Bird Atlas also includes breeding evidence at the levels of confirmed (e.g. nest containing eggs, adults carrying food for young), probable (e.g. a pair observed in suitable nesting habitat or courtship displays observed) or possible (e.g. species observed in breeding season in suitable nesting habitat) for each 10 km square, and also at the tetrad level. The maps show squares in which breeding was confirmed, probable or possible. Additional observations of breeding gulls recorded in BirdTrack between 2012 and 2015 have been included to show areas where breeding gulls have colonised since the Bird Atlas data were collected. The number and percentage of 10 km squares within the UK and Ireland with breeding gulls present are given in Table 8. Comparisons with the data collected for Seabird 2000 (Mitchell *et al.* 2004) are included within the text, and the maps are reproduced, with permission, in Appendix 1.

Table 8. Number and percentage of 10 km squares within the UK and Ireland with breeding gulls present based on data from Bird Atlas 2007-11 and from BirdTrack for 2012-15.

Species	No. of 10 km squares with breeding gulls present			% of 10 km squares with breeding gull records from Bird Atlas 2007-11 and/or BirdTrack
	Bird Atlas 2007-11	BirdTrack	Bird Atlas 2007-11 and/or BirdTrack	
Black-headed Gull	1036	276	1066	26.3
Common Gull	807	123	824	20.3
Lesser Black-backed Gull	956	164	988	24.4
Herring Gull	1236	306	1264	31.2
Great Black-backed Gull	729	119	752	18.5
Kittiwake	238	107	254	6.3
All species	2183	687	2204	54.3

Black-headed Gulls have a wide breeding distribution, with the highest breeding densities in Britain occurring in Orkney, northern England, East Anglia, the Thames Estuary and the Solent. In Ireland, breeding is less ubiquitous, with agglomerations around Lough Neagh, Strangford Lough and wetland habitats in the west and northwest of Ireland (Balmer *et al.* 2013). Between 2012 and 2015, breeding has been recorded in an increasing number of squares in southwest England, where the species was absent during Bird Atlas 2007-11. Comparing with the distribution of breeding Black-headed Gulls reported in Seabird 2000 (Mitchell *et al.* 2004), the breeding range has expanded within the south and Midlands regions of England and also within the Republic of Ireland and to a few previously unoccupied sites along the northwest coast of Scotland.

Common Gulls breed mainly in the north and west of Scotland and Ireland. Within Scotland, the highest densities occur in the east, from Angus to Moray Firth, Caithness, the Northern Isles and many straths and glens in the Highlands. Within Ireland, the breeding distribution is largely coastal, with the exception of counties Mayo and Galway (Balmer *et al.* 2013). Common Gulls were scarce breeders within England prior to 2012, with a few isolated colonies in northern England, and coastal colonies in East Anglia, Kent and Hampshire. However, since 2012, breeding has been increasingly recorded throughout England, including some inland sites. The Seabird 2000 data show most of the breeding colonies in Scotland from 1998-2002 occurred at natural sites, with only 14 roof-nesting colonies identified, comprised of 621 AONs (Mitchell *et al.* 2004).

Lesser Black-Backed Gull breed throughout most of the British coastline, but are absent from much of the southeast coast of Ireland (Balmer *et al.* 2013). The species' breeding range has expanded, with a notable increase in inland sites occupied since 2012. A particularly high abundance of roof-nesting birds were recorded by Seabird 2000 in and around Glasgow, Bristol and Gloucester (Mitchell *et al.* 2004).

Herring Gull breeding distribution is predominately coastal, although the species' tendency to nest on buildings has led to colonisation of many inland urban areas (Balmer *et al.* 2013). At the time of Seabird 2000 (Mitchell *et al.* 2004), at least 15% of nesting Herring Gulls occurred in urban areas. The proportion of nesting gulls occurring in urban areas is likely to be higher now (Coulson 2015). There has been an increase in occupancy of inland sites throughout England since 2012.

Great Black-backed Gull breeding distribution is mainly coastal within Britain and Ireland. The species was absent from the eastern coast of Britain between Lothian and Kent prior to 2012 (Balmer *et al.* 2013); however, a few individuals have bred more recently (e.g. the north Norfolk coast, Yorkshire and Northumbria). Highest abundances of Great Black-backed Gulls during the breeding season occur in the Northern Isles, northwest Scotland and western Ireland.

Kittiwake breeding colonies generally occur on coasts with rocky cliffs; however, man-made structures such as buildings, bridges and offshore oil rigs provide additional nesting habitats. The species' breeding abundance is greatest along the eastern coast of Britain between Flamborough Head and Orkney. Breeding range expansion since 2012 has been minimal, with 13 additional 10 km squares occupied, most of which occur along the northwest coast of England and Scotland. Apart from a few additional breeding sites along the southwest coast of Ireland, the breeding range does not appear to have altered much since Seabird 2000.

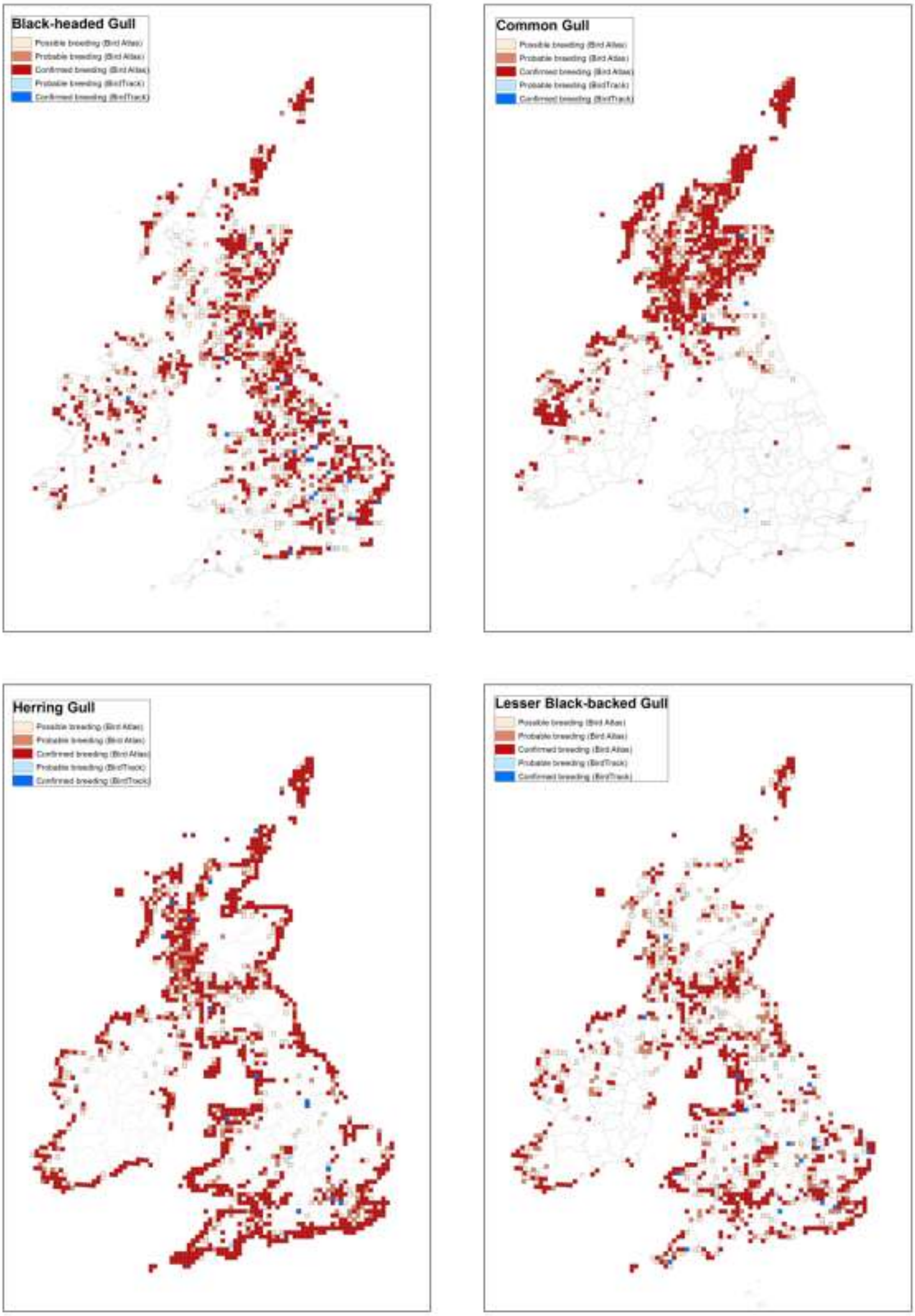
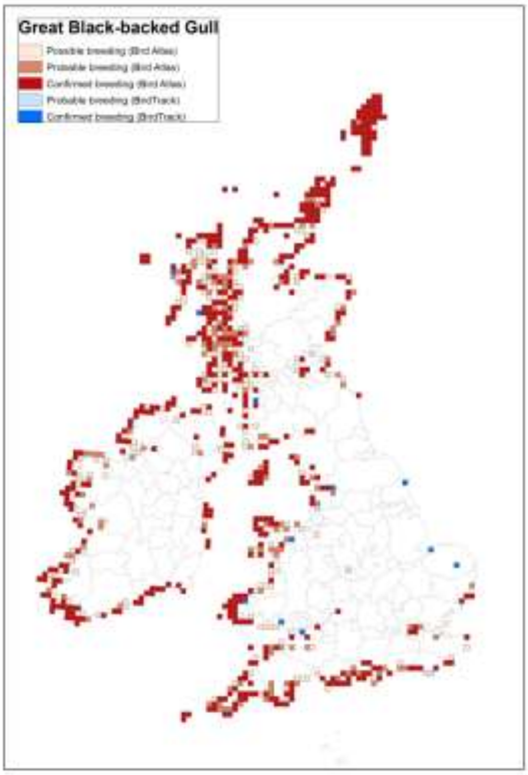


Figure 2. The distribution of breeding gulls within the UK and Ireland at 10 km square resolution (data source: Bird Atlas 2007–11 and BirdTrack 2012–15) [continued on next page]



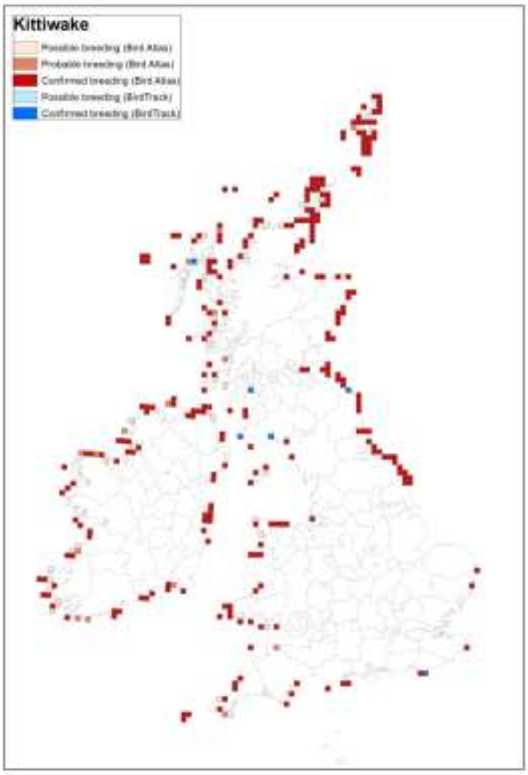


Figure 2. The distribution of breeding gulls within the UK and Ireland at 10 km square resolution (data source: Bird Atlas 2007–11 and BirdTrack 2012–15) [continued from previous page]

3.3.2 Other recent survey data

A 2009 survey of large breeding gulls in Cumbria found 43 colonies of Lesser Black-backed Gull, 40 colonies of Herring Gull and seven colonies of Great Black-backed Gull, with total number of AON recorded for each species: 15,489, 4,747 and 85, respectively (Sellers and Shackleton 2011). Most of the gulls surveyed occurred in large natural coastal colonies, such as South Walney and Rockcliffe Marsh, but a substantial proportion occurred in large coastal towns, including Carlisle, Barrow, Sellafield and Maryport, with many smaller colonies in smaller towns, and inland lakes or quarries. Since the Seabird 2000 census (Mitchell *et al.* 2004), increases in numbers of Lesser Black-backed and Herring Gulls were observed within the urban colonies; however, there were declines in overall numbers, driven by large decreases in numbers at coastal colonies. Great Black-backed Gulls were largely restricted to coastal colonies with less than 10 pairs in urban sites either during the Seabird 2000 census or in the 2009 survey.

This survey showed that the preferred urban nesting locations varied between species. Lesser Black-Backed Gulls tended to nest on open roof tops that were flat or gently sloping (64% of all nests) and preferred roofs covered in moss or grass. Herring Gulls also commonly occurred on open roof tops (37% of all nests), but often chose less open locations such as chimney stacks (27%) or behind vents (13%). Twenty-four percent of Lesser Black-backed Gull nests occurred on the ground, compared with only 6% of Herring Gull nests (Sellers and Shackleton 2011). In addition, a recent study of Common Gulls nesting in built-up areas in the Scottish Highlands showed a preference for open, pitched roofs, and older, weathered roofs with moss, lichen or other vegetation present (Sellers 2015).

3.3.3 Gull abundance

While understanding the distribution of breeding gulls is useful in itself, data on abundance can provide further information that is of use in determining strata for potential sampling. Using Bird Atlas 2007–11 data, we examined the abundance of breeding gulls within the UK and Ireland, including only records with possible, probable or confirmed evidence of breeding. Although data were collected at the tetrad level, not all tetrads were covered, so average tetrad abundance for each 10 km square within Britain and Ireland was calculated. Histograms of abundance obtained for each species are plotted below on a natural log scale.

3.3.4 Urban land cover in the UK

There are various conceivable methods for defining what is meant by urban. However, what we are interested in for the purposes of counting ‘urban’ gulls is the presence of any man-made structures, particularly flat-roofed buildings, which provide suitable nesting habitat for gulls. Such structures may occur in high abundance within urbanised areas, but similarly, one or two solitary structures may occur in a landscape that would otherwise be defined as ’rural’ or not urban, by any habitat classification scheme.

Given this, a truly complete census of urban gull populations in the UK and Ireland is unlikely to be feasible. A census might thus consider only areas that include above a certain threshold coverage of ‘urban’ habitat. Alternatively, a broader survey might be undertaken using a paired key site and stratified sampling approach, the latter covering the entire spectrum of urbanisation. Further details of proposed survey design are discussed in Section 4.

The percentage urban cover data for the UK at 1 km square resolution, from the LCM 2007 dataset are presented in Figure 3. From this figure, it is clear that large proportion of the UK is covered by urban areas. With the exception of the Scottish Highlands, parts of northern England and north Wales, most 1 km squares had some degree of urbanisation or occurred close to areas of urbanisation.

We identified potential key urban sites within the UK, using the ESRI ‘mjurban’ shapefile, and selecting all urban areas large enough to have their own metropolitan district or unitary authority administrative boundary. These potential key sites are mapped in Figure 4, with the largest sites highlighted in dark red (London, Manchester, Birmingham, Glasgow, Edinburgh, Dublin and Liverpool). The approximate areas of these sites are provided in Table 9. These sites were identified based solely on geographical extent as indicated by the ‘mjurban’ shapefile (excluding sites that did not have their own metropolitan district or unitary authority administrative boundary), and are intended only as an indicative guide in developing survey methodology and costings. It is likely that the actual site selection for the survey will be guided by other practical reasons such as statutory monitoring priorities for individual countries and casework requirements. So while the current selection of sites presented here displays some bias towards urban sites in England, it is expected that actual site selection would differ.

The regions used in Figure 4 are based on those used for the Winter Gull Roost Survey (Burton *et al.* 2013), with Wales being subdivided into north and south. As with the potential key sites, these regions act as a guide in generating a survey design, and could be altered in the final survey design.

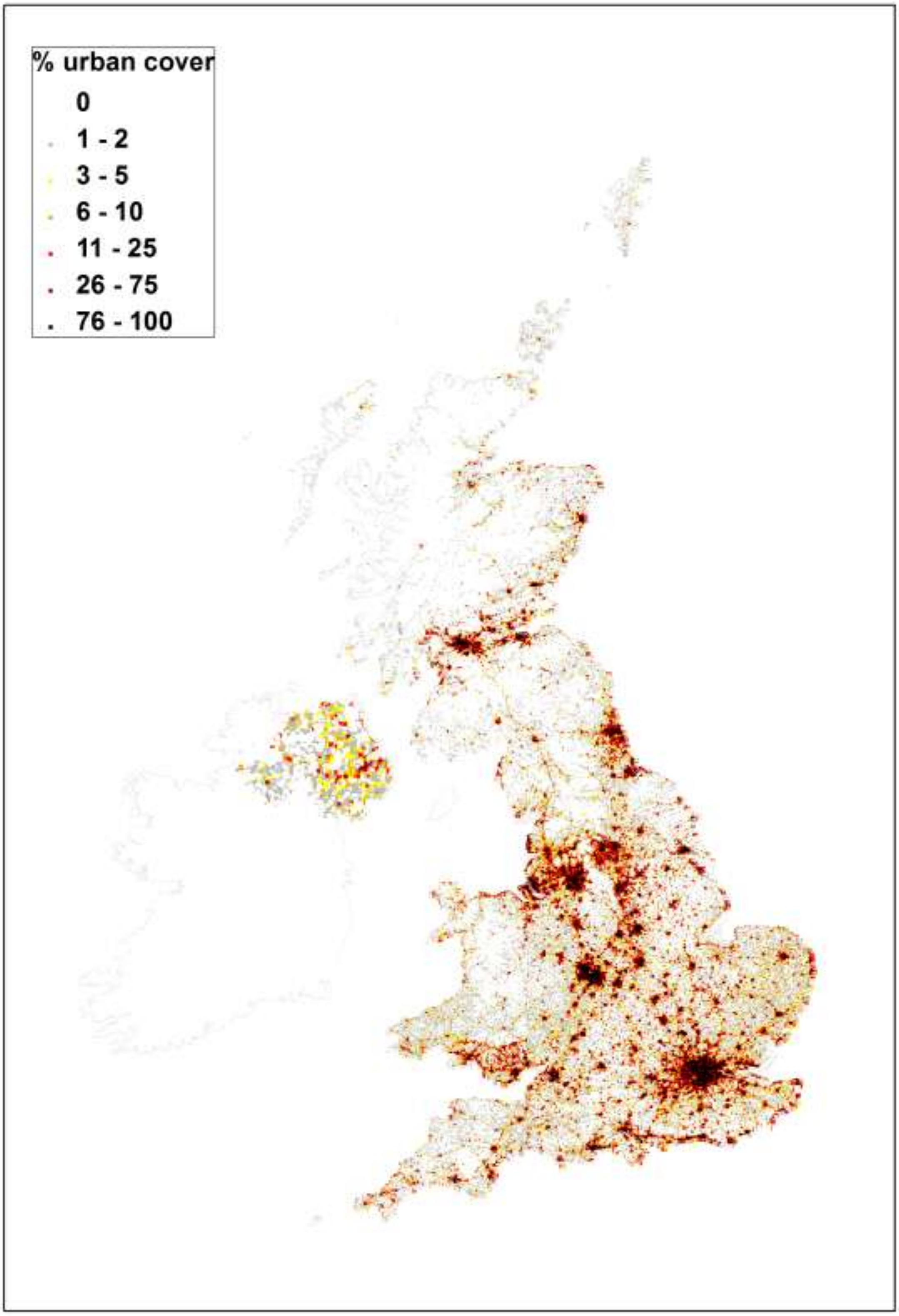


Figure 3. Percentage urban cover in the UK at 1 km square resolution. Based on Land Cover Map 2007 data© NERC (CEH) 2011. Contains Ordnance Survey data © Crown Copyright 2007, 2009. © third-party licensors.

Table 9. Potential key urban sites within the UK and Ireland and their approximate area.

Urban Areas	**Area (km^2)**
Birmingham	621
Dublin	926
Edinburgh	261
Glasgow	175
Liverpool	105
London	1574
Manchester	935

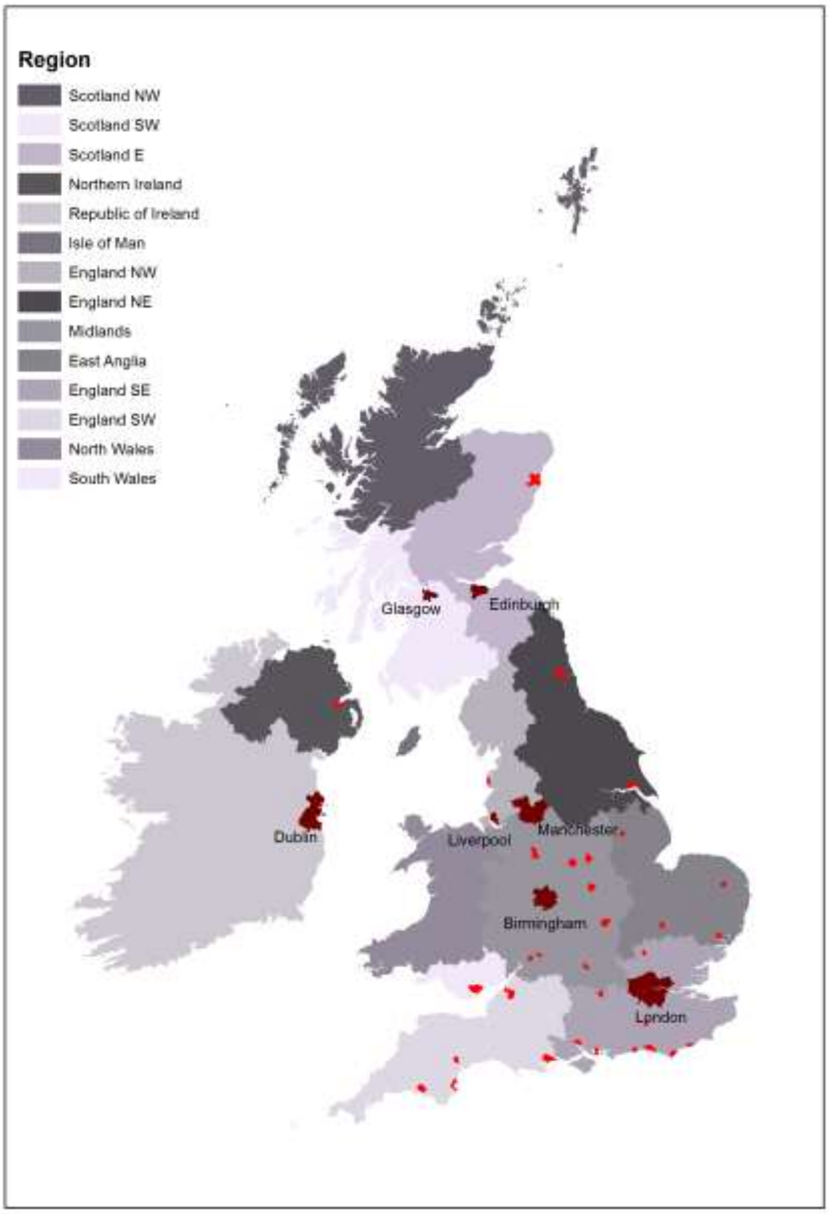


Figure 4. Map showing potential key urban sites in dark red with other urban areas shown in red. Sites selected based on size as indicated by ESRI 'mjurban' shapefile. Sites that did not have their own metropolitan district or unitary authority administrative boundary were not included.

The proportion of 1 km squares within each of seven categories of urban cover (0%, >0–2%, >2–5%, >5–10%, >10–25% and >25–75% and >75%) for each region are shown in Figure 5. The three Scottish regions had the highest proportion of squares with 0% urban coverage – 77% in Scotland E, 81% in Scotland SW and 92% in Scotland NW. By contrast, in England SE only 34% of squares had 0% urban cover, 20% of squares had >25% urban cover and 8% had >75% urban cover.

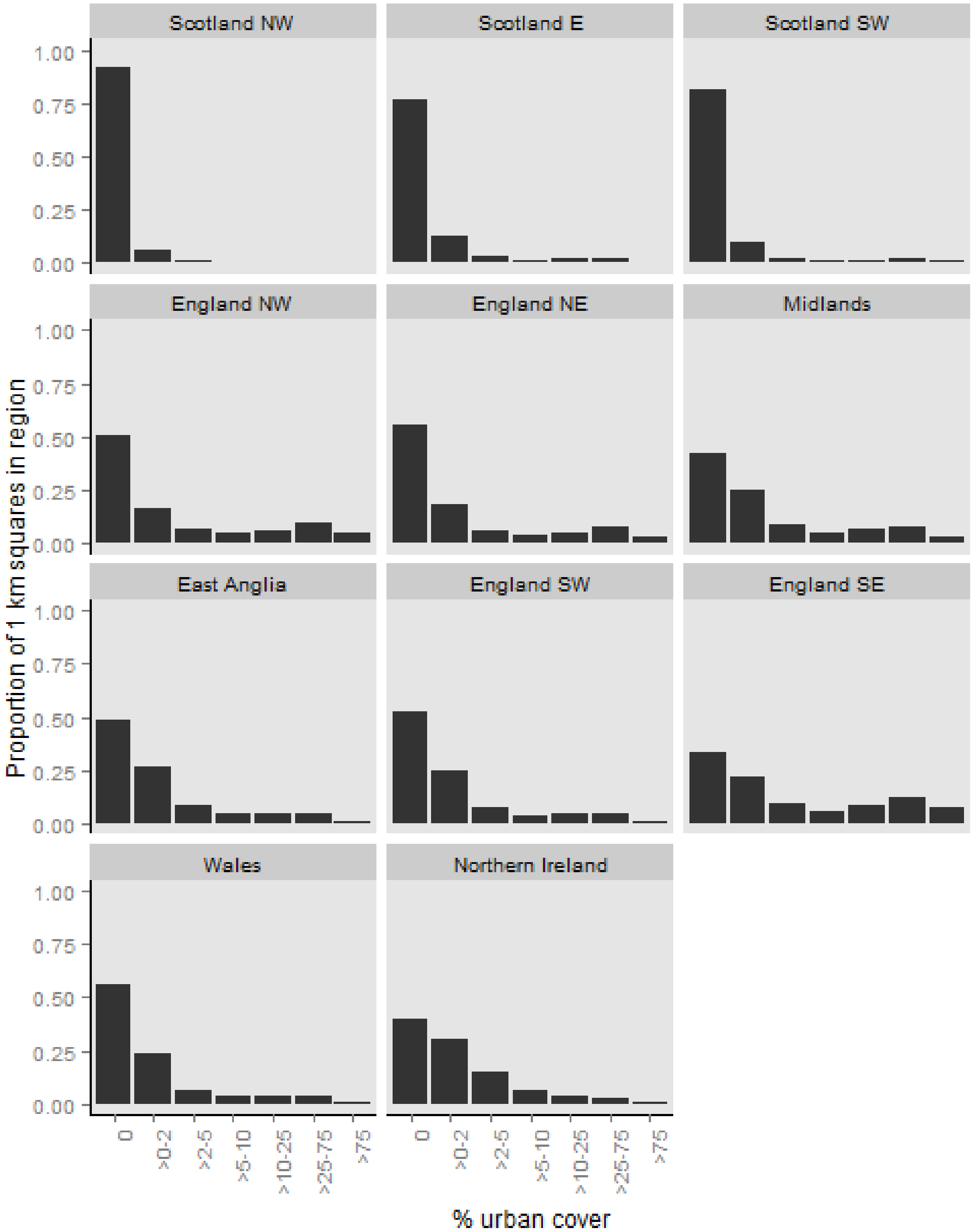


Figure 5. The proportion of 1 km squares within defined regions of Britain and Ireland within each category of urban cover (0%, >0–2%, >2–5%, >5–10%, >10–25%, >25–75% and >75%).

3.3.5 The concurrence of breeding gulls and urban areas

Based on data from Bird Atlas 2007–2011 and from BirdTrack for 2012–15 at the 10 km square level, breeding Herring and Lesser Black-backed Gulls – the most common urban breeding gull species in Britain and Ireland – were present in all but three of the urban areas identified in Figure 4 (Nottingham, Stevenage and Cambridge). The concurrence of urban sites and Lesser Black-backed and Herring Gull distribution is shown in Figure 6.

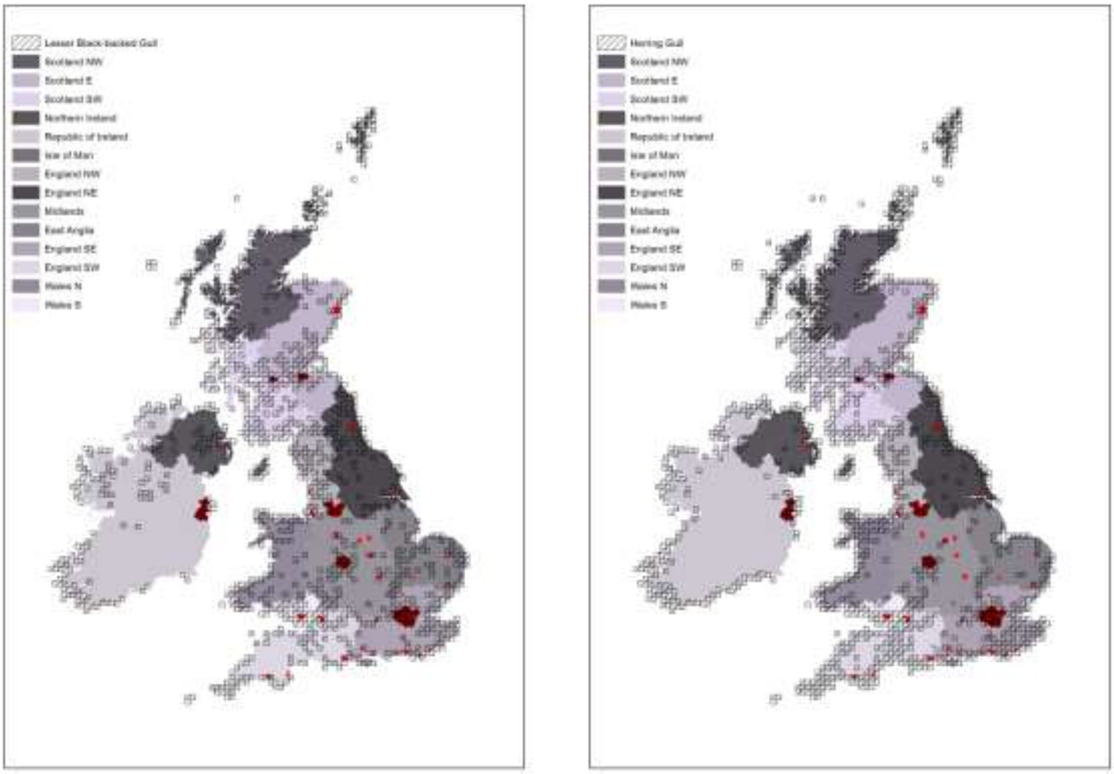


Figure 6. The concurrence of breeding Lesser Black-backed Gull (left) and Herring Gull (right) at the 10 km squares level (data source: Bird Atlas 2007–11 and BirdTrack 2012–15) with potential key urban sites. Includes possible, probable and confirmed breeding records from Atlas and probable and confirmed breeding records from BirdTrack.

The probability of occurrence of breeding Herring Gulls or Lesser Black-backed Gulls within inland 10 km squares in the UK based on data from Bird Atlas 2007–2011, according to the % urban cover within the square according to LCM 2007 is presented in Figure 7. This analysis indicates that there is a clear association between breeding gull occurrence and urban areas at inland sites. The probability of breeding gulls occurring increases rapidly in squares with >25% urban cover and in squares with 100% urban cover the probability of gulls occurring is 95% for Herring Gulls and 94% for Lesser Black-backed Gulls. The probability of breeding gulls occurring increases markedly between 25 and 75% urban cover; however, even at very low urban cover breeding gulls may be observed. The probability of observing breeding gulls within squares with 1% urban cover is 10% for Herring Gulls and 16% for Lesser Black-backed Gulls.

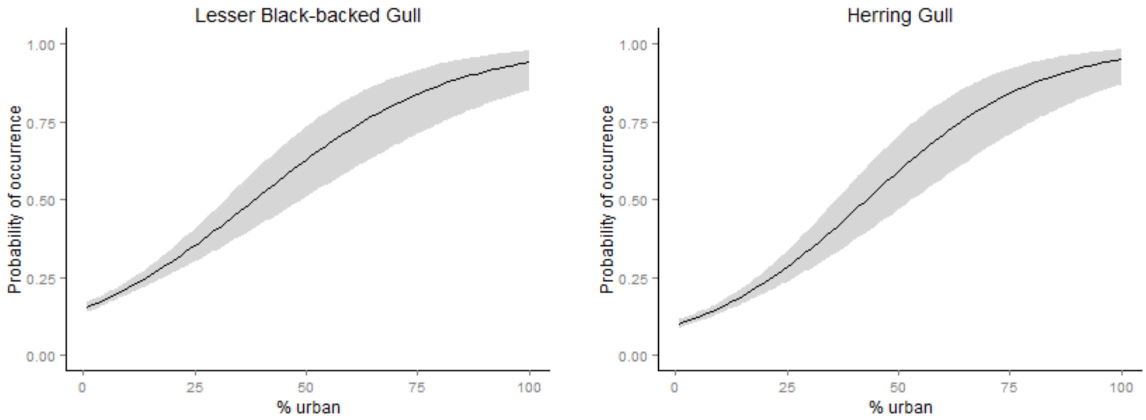


Figure 7. Relationship between probability of occurrence of breeding Lesser Black-backed Gull (left) and Herring Gull (right) and % urban cover within a 10 km square for inland sites in the UK, based on data from Bird Atlas 2007–2011 and Land Cover Map 2007. Grey shading indicates 95% confidence limits.

The relative frequency of tetrads with low, medium and high gull abundance, within squares of different levels of urban cover is presented in Figure 8. The abundance classes were defined according to natural breaks in the abundance frequency (see Figure 9). For Black-headed Gull, Great Black-backed, Herring Gull and Lesser Black-backed Gull the classes were defined as low: <=10 gulls per tetrad (on average per 10 km square), medium: 11–50 and high: 51+. For Common Gull, the classes were low: <=3, medium: 4-10 and high: 11+, for Great Black-backed Gull, low: <=2, medium: 3–10, and high: 11+; for Kittiwake, low: <=10, medium: 11–400 and high: 401+. For all species combined the groupings were low: <=10, medium: 11-100, and high: 101+. Note the critical feature of interest for this figure is not the actual numbers birds occurring at each level of urban cover, but the relative proportions of birds in the L, M and H categories (so note different scales on the y axis for each level of urban cover).

In Figure 8, for the all species group, the proportion of low gull abundance squares decreases and the proportion of medium gull abundance squares increases in more urbanised areas. This appears to be driven largely by Lesser Black-backed Gull abundance – in areas of little or no urbanisation, Lesser Black-backed Gulls occur mainly at low abundance; however, as the extent of urbanisation increases, the proportion of medium abundance squares increases dramatically. At >75% urban cover, the proportion of medium abundance squares exceeds that of low abundance. In addition, there appears to be an initial jump in the proportion of medium abundance Lesser Black-backed Gull squares between the 0–2% urban and >2–5% urban categories, suggesting that 2% might be considered an appropriate threshold coverage of 'urban' habitat. By contrast, for Herring Gull the frequency of squares of low, medium and high abundance all increase as the urban cover increases (suggesting a positive association of all abundance classes with level of urbanisation), but there are proportionally more birds in the low and medium abundance categories as the level of urbanisation increases. Common Gull appears to be strongly negatively associated with urban areas, with the frequency of squares of all abundance categories declining as the extent of urbanisation increases.

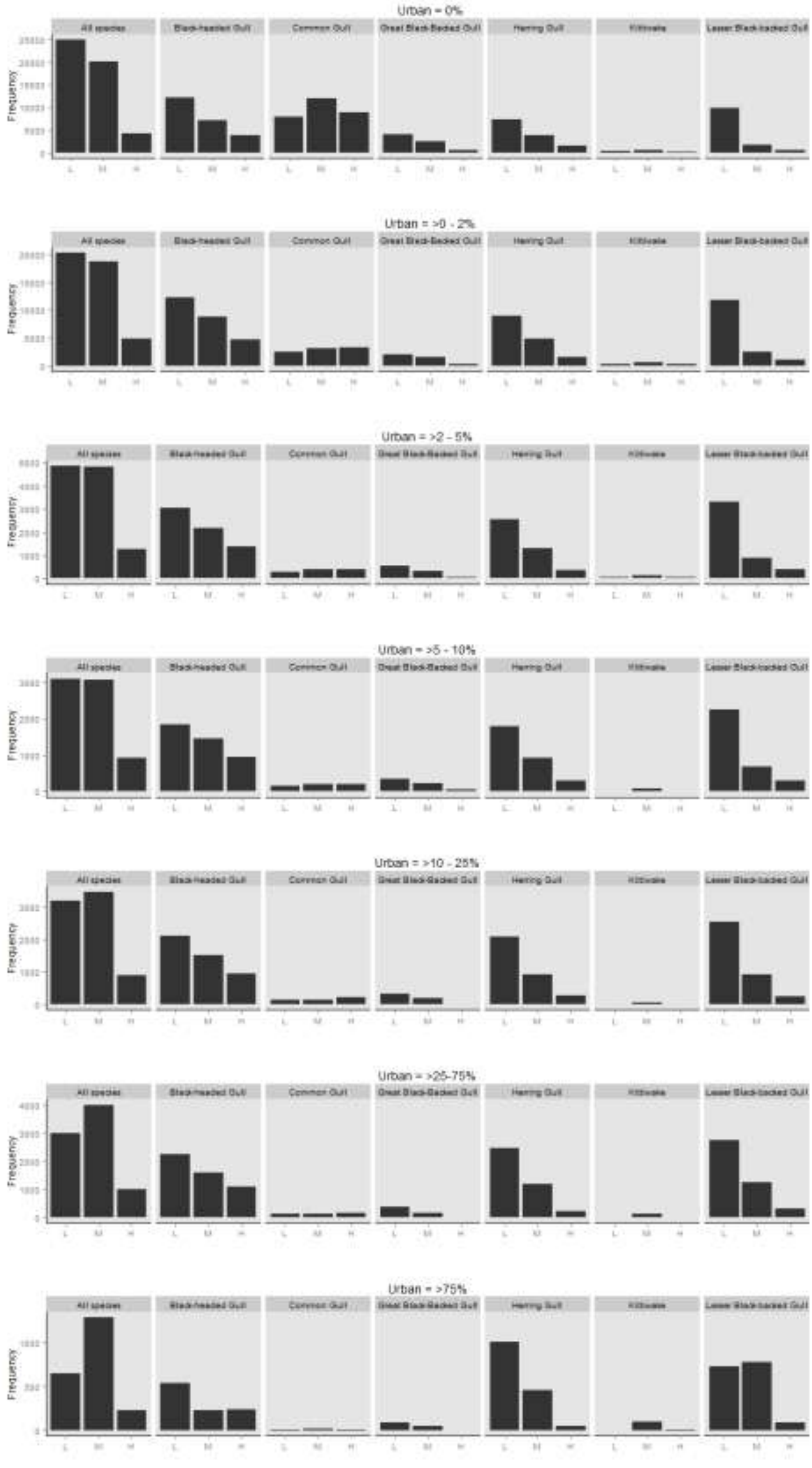


Figure 8. The number of squares with low, medium and high abundance of gulls split by level of urban cover, based on data from Bird Atlas 2007–2011 and Land Cover Map 2007. Note the differing y axes for each level of urban cover.

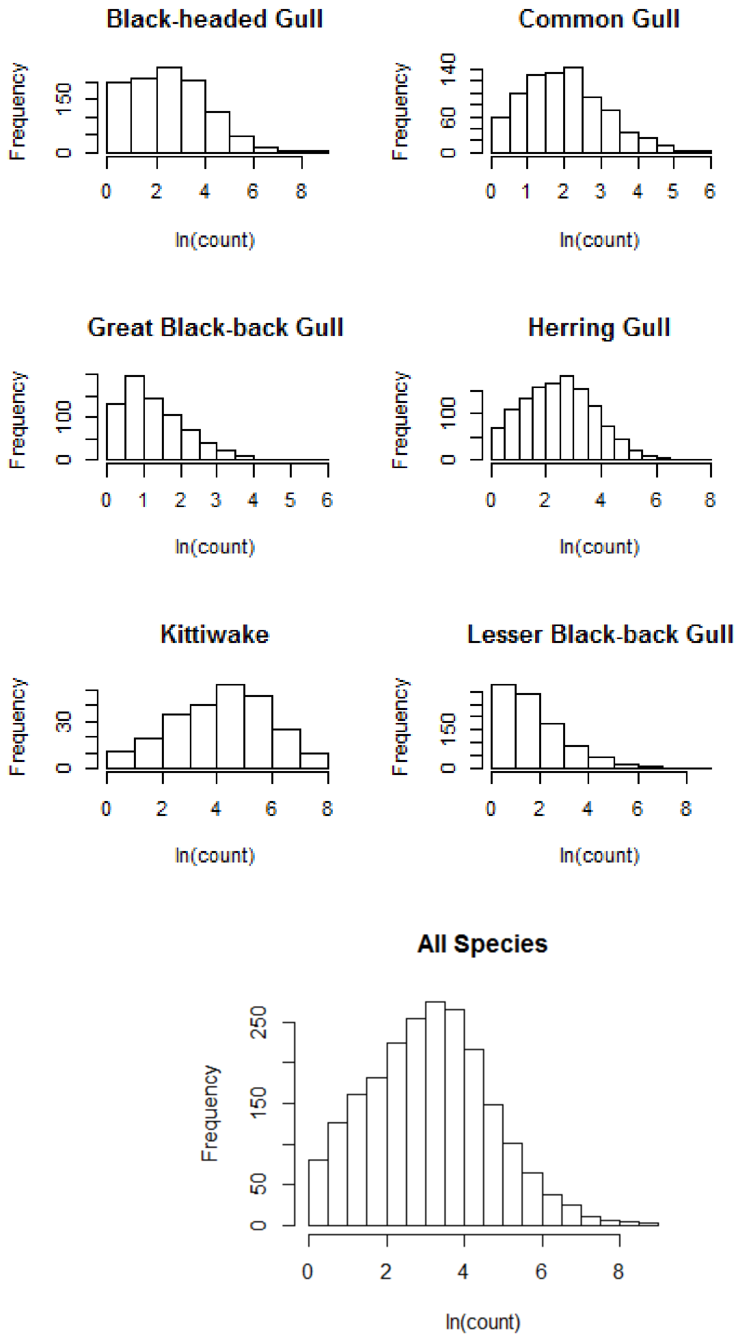


Figure 9. Histogram of average tetrad breeding gull abundance for each 10 km square in Britain and Ireland, for individual species and all species, based on data from Bird Atlas 2007–2011.

4. DEVELOPING A CENSUS DESIGN

4.1 Introduction

This report has been commissioned by Natural England to inform the design and implementation of future census efforts, and to make recommendations for the most cost-effective survey strategy for delivering urban gull population estimates for the UK and Ireland, as well as any specified key sites.

As stated in section 3 above, the 'urban' habitat in which gulls may nest (i.e. man-made structures, and particularly flat rooftops) occurs virtually everywhere in the UK and Ireland, both in areas of high urbanisation and in landscapes that would otherwise be defined as 'rural' or not urban, by any habitat classification scheme. Approximately 65% of 1 km squares in the UK have 0% urban land cover according to the LCM 2007 dataset; however, even squares with 0% urban according to this dataset may contain man-made structures suitable for nesting gulls, as the minimal mappable unit for all habitats was 0.5 ha (Morton *et al*., 2011). Furthermore, gulls occur at sites even with very low urban cover (see Figure 7).

Given this a truly complete census of urban gull populations in the UK and Ireland is unlikely to be feasible. A census might consider only areas that include above a certain threshold coverage of 'urban' habitat, although the difficulties of obtaining complete coverage even on this basis would be considerable and likely impracticable. In addition, Scotland in particular would likely be underrepresented using this method.

As an alternative, we thus propose a broader survey using a paired key site and stratified sampling approach, the latter covering the entire spectrum of urbanisation. Potential large key urban sites (as defined in Figure 4) including London, Manchester, Birmingham, Edinburgh, Glasgow, Liverpool and Dublin are covered; for the remainder of each region, a sample of 10 km squares are selected. These 10 km squares would be randomly selected according to the stratification approach outline below.

Note: it is assumed that many non-urban coastal sites, large inland colonies and protected sites would be covered by the national seabird census and thus may effectively be treated as key sites and excluded from the areas to be covered by the proposed sampling. A large proportion of Great Black-backed Gulls (which occur mainly at large coastal colonies), would already be covered, for example. In addition, it would be practicable for the more widely distributed non-urban inland colonies of Common and Black-headed Gulls – which would not be well covered by a key sites approach – to be covered through a volunteer-based survey, itself potentially using a key site and sampling approach, rather than by remote survey.

4.2 Stratification

The proposed stratification would be based on gull abundance (low, medium high), region (according to regions defined in Figure 4), % urban cover, and whether the site is coastal or inland. Table 10 shows the number of 10 km squares within each region in which gulls occur (for Herring and Lesser Black-Backed Gulls combined, and all gull species combined). The first two columns show the total number of squares in which these gulls occur, and subsequent columns show the number of squares in which gulls occur that have at least >0% urban cover and >2% urban cover (see section 3.3.5 above).

Table 10. The number of 10-km squares in which Lesser Black-backed Gulls (LB) and Herring Gulls (HG) occur in each defined region of the UK and Ireland, based on data from Bird Atlas 2007-11 and from BirdTrack for 2012-15.

Region	All squares HG & LB only	All squares All species	>0%urban HG & LB only	>0%urban all species	>2%urban HG & LB only	>2%urban all species
Channel Islands	12	12	0	0	0	0
East Anglia	86	121	86	121	74	95
Isle of Man	105	178	105	178	60	98
England NE	81	112	81	112	48	66
England NW	14	14	0	0	0	0
England SE	85	165	85	165	77	140
England SW	55	65	51	61	29	37
Midlands	205	276	4	7	2	3
Northern Ireland	321	430	206	279	16	17
Republic of Ireland	213	254	167	208	35	39
Scotland NW	138	264	133	251	73	81
Scotland SW	152	165	152	165	143	159
Scotland E	169	168	160	159	114	118
Wales	150	160	144	154	85	86

4.3 Costings

Costings of aerial surveying (encompassing costs for coverage of sites, transit and analysis) were provided for the purposes of this project from two commercial providers (APEM Ltd and HiDef Aerial Surveying Ltd). While specific costings are commercially sensitive and thus strictly confidential, we are able to provide average costings from the two providers for coverage of the potential key sites proposed and the average cost for covering any 10 km square within each region. These costs are provided separately to Natural England in a confidential spreadsheet and with a supporting document.

We have considered costs for data obtained at 2 cm, 3 cm and 5 cm resolution (HR) / definition (HD). Obtaining data at 5 cm resolution or definition would lead to an increase in uncertainty for species identification, especially for smaller gull species, but, ca. 80% of adult Lesser Black-backed Gull and Herring Gull could be identified to species level (based on HR imagery – see section 2.3).

4.4 Comparison with cherry picker costs

A quote for hiring cherry pickers was obtained from HSS Hire, which operate throughout the UK. Hire costs range between £250 and £400 per week, depending on the specification of the machine. Transport costs to anywhere within the UK are between £70 and £100, except for the Scottish Highlands and southwest England which are priced on a case-by-case basis. Additional costs include obtaining an operator licence, which involves a 1-day training course, costing between £175 and £195 per person.

A previous survey of Cardiff using cherry pickers was completed in six days (Rock 2011). Based on the above, assuming the operator is already licenced, cherry picker rental for this length of time would cost between £320 and £500 per week. However, there are many logistical issues to consider such as obtaining access permissions, police assistance, etc., plus the additional costs of drivers and fieldworkers that would increase the time taken and make scaling this method up to a national scale impractical. However, it is proposed that for the purpose of ground-truthing and validation of aerial counts, this method can be implemented at reasonable cost for small areas.

4.5 Conclusions

Due to the extensive occurrence of 'urban' habitat and the widespread distribution and association of gulls with this habitat through Britain and Ireland, a truly complete census of urban gull populations in the UK and Ireland is considered to be impracticable, even considering only areas that include above a certain threshold coverage of 'urban' habitat.

We have thus proposed a broader survey methodology using a paired key site and stratified sampling approach. In the first instance, it is suggested that this would best be achieved by digital aerial survey, given the practicalities of using cherry pickers on a broad scale. Costs are thus provided separately for coverage by digital aerial survey of potential key sites and for covering any 10 km square within defined regions and at the country level.

Nevertheless, it should be noted that the key sites identified in this report only represent a potential suite of sites that might be selected, and it is likely that a final choice of key sites will depend on casework needs and statutory monitoring priorities. Once a final selection of key sites has been determined, consideration should be given as to whether it may be possible to save on survey costs at some of these sites by using alternative methods, such as visual aerial survey, cherry pickers or vantage point surveys, especially where these have proven successful before, utilising volunteer or public involvement where appropriate.

Without additional simulation, it is not possible to firmly conclude on an appropriate level of coverage to delivering robust urban gull population estimates for the UK and Ireland. Such a simulation would also need to take into account other potential considerations not available at this time, including which urban areas might need to be covered as key sites (e.g. due to statutory country priorities or casework needs), an understanding of coverage of non-urban populations by the national seabird census as a whole, as well as the potential funding available which will trade-off with the accuracy of estimates obtained.

Confidence limits around estimates will principally be a product of the level of species' identification obtained by aerial surveys (this reflecting resolution of images), the accuracy of aerial surveys in determining whether individuals are nesting or not and the level of coverage obtained. The latter will reflect the balance between covering key sites or sampling, and the coverage obtained within survey sites (whether key sites or sample areas) and across the country as a whole.

In the absence of additional simulation, an approximate figure of 100 10 km squares (equating to 2,500 2-km tetrads) in addition to the potential key sites, is proposed, reflecting coverage by other previous surveys (e.g. the 2003/04-2005/06 Winter Gull Roost Survey: Banks *et al.* 2007; Burton *et al.* 2013; and the Dispersed Waterbird Survey: Jackson *et al.* 2006). Examples of the costs that might be required to achieve this level of coverage, depending on coverage within the sample 10 km squares and potential key sites and assuming, at this point, that sample 10 km squares are evenly distributed across regions, are provided with the confidential spreadsheets. Note that additional costs would be required to manage the project as a whole and to subsequently produce the population estimates required.

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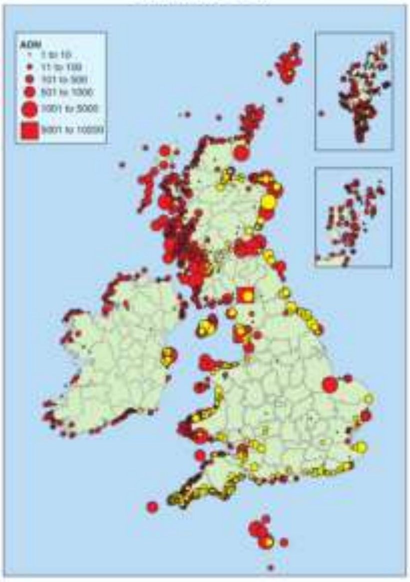
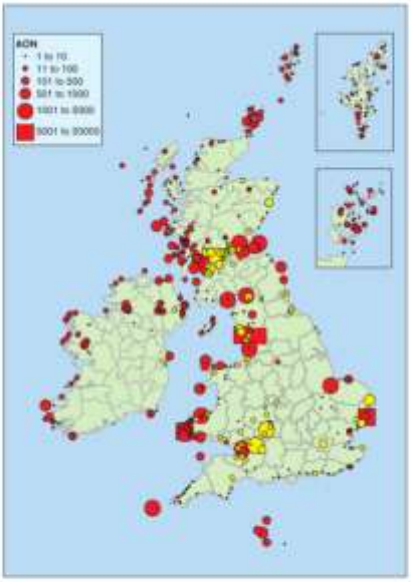
APPENDIX 1. Abundance and distribution of breeding Gulls in Britain and Ireland 1998–2002.

Natural sites are shown in red and man-made sites (e.g. rooftops) are in yellow (the scale is the same for both types of sites). Reproduced with permission from Seabird 2000 (Mitchell *et al.* 2004).

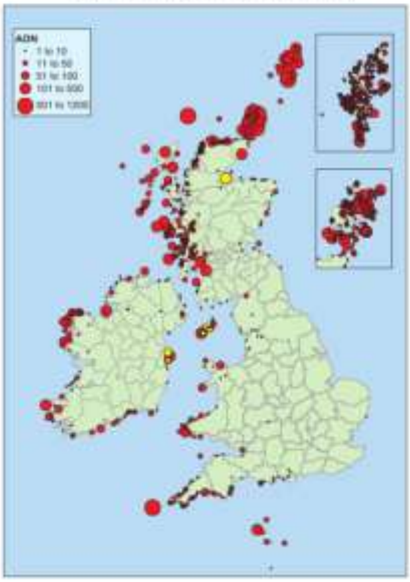
Herring Gull



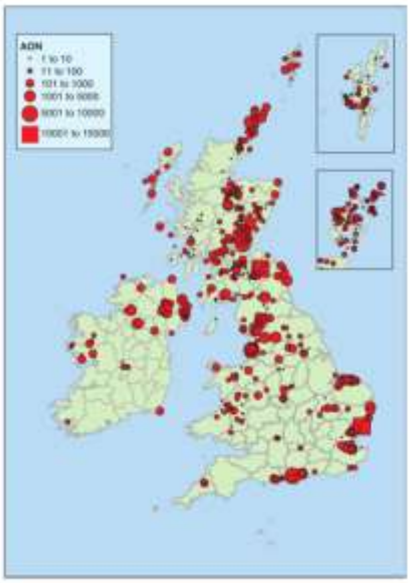
Lesser Black-backed Gull



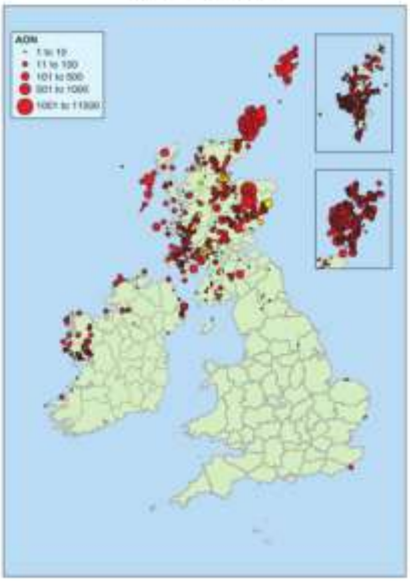
Great Black-backed Gull



Black-headed Gull



Common Gull



Black-legged Kittiwake