Population estimates for urban and natural nesting Herring Gull *Larus argentatus* and Lesser Black-backed Gull *Larus fuscus* in England.

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EQA:
This report is compliant with JNCC's Evidence Quality Assurance Policy
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Executive Summary

The fourth Britain and Ireland breeding seabird census, *Seabirds Count* (2015-present) was scheduled to complete in 2020 but the COVID-19 pandemic suspended voluntary fieldwork that season and the census was therefore extended into 2021. *Seabird 2000* (1998-2002) was the first breeding seabird census in Britain and Ireland to produce separate inland, coastal, natural-nesting and roof nesting population estimates of Herring Gull (HG) and Lesser Black-backed Gull (LB); *Seabirds Count* will be the second census to achieve this.

In 2019 and 2020, all GB Statutory Nature Conservation Bodies (SNCBs) engaged in substantive reforms of their licensing regulatory functions, which affected a number of species that included HG and LB. Reform was driven by conservation status concerns, legal challenge and by improved understanding about interrelationships and impacts by human activities relating to these two gull species. In England, to ensure that policy decisions and licensing reform were based on robust evidence, the urban-nesting gull survey programme was accelerated with the aid of Defra funding. Contracted surveyors delivered ground-based surveys (GBS) and corresponding digital aerial surveys (DAS). The Joint Nature Conservation Committee (JNCC) was commissioned by Natural England (NE) to produce two reports – one that describes methods of obtaining correction models based on comparative analysis of DAS and GBS of HG and LB to account for detectability issues posed by the urban environment[1]; and this report, which presents gull breeding population estimates and the extrapolation analysis used to derive these estimates for urban nesting gulls in association with correction factor models.

This report presents up-to-date breeding population estimates for HG and LB in England and of that, estimates of urban (roof-nesting) and rural (natural-nesting) populations, including their corresponding coastal (<5 km from coast) and inland (>5 km) populations. Urban-nesting refers to HG and LB on elevated urban fabric, but excludes ground-nesting within urban environments. Natural-nesting HG and LB are those nesting anywhere else and includes within urban landscapes at ground level. Natural-nesting population estimates presented in this report that coincide with and use *Seabird 2000* data have been adjusted from those previously published from that census to reflect corrections subsequently made to clerical errors in the original dataset, and to accommodate additional colony counts. These results therefore allow direct and reliable comparisons between *Seabird 2000* and *Seabirds Count*.

Natural nesting breeding abundance data for HG and LB continues to be collected as part of the *Seabirds Count* census using standardised methods (Walsh et al., 1995). An 'urban gull sub-group' of the *Seabirds Count* steering group was set up in 2015 to produce a sample-based approach and methodology for GBS of urban nesting populations of HG and LB in England; this was launched in 2019. The GBS methodology collects abundance counts in three ways – i) apparently occupied nests (AON), apparently occupied territories (AOT) and breeding adults (IND), all from a stratified random sample of 1 km urban squares. Urban gull population estimates were then derived from two methods, each were two-step processes – Method 1): firstly, this corrected GBS counts using species-specific correction model, then extrapolated estimates through a random replacement survey sample exercise of species-specific 'groupings' ('bootstrapping') to the England level. Method 2): firstly, this extrapolated through modelled distribution and abundance of GBS IND, then corrected these estimates using species-specific correction model.

[1] Burnell, D. 2021. Urban nesting Herring Gull Larus argentatus and Lesser Black-backed Gulls Larus fuscus population estimates: devising species-specific correction models for ground-based survey data. Natural England publication ref: JNCC21_01

The associated report, 'Urban nesting Herring Gull *Larus argentatus* and Lesser Black-backed Gull *Larus fuscus* population estimates: devising species-specific correction models for ground-based survey data. Natural England publication ref: JNCC21_01 (Burnell, 2021), describes the correction factor models produced to account for known detectability issues present in urban GBS data (Coulson and Coulson, 2015).

There is statistical confidence in declines shown in England's natural-nesting populations of HG (-38%) and LB (-45%) over the past 20 years. Direct comparisons between roof-nesting gull population estimates in this report with hitherto published estimates derived from *Seabird 2000* are unreliable due to under-estimations in that census. Furthermore, conclusions drawn from results presented here about urban gull population estimates should be made cautiously and caveated with regard to the particularly wide confidence limits (CLs). There remains some uncertainty about the robustness of the correction model and the inherent detectability issue with the survey method itself.

Propagating error through the two-step processes, or in other words accounting for the error present in the first step when conducting the second, has been a challenge. To ensure that margins of error are as transparent as possible, results from both methods each have three estimate ranges produced using the - i) estimated mean, ii) lower (LCL) and iii) upper (UCL) of AONs, derived from the correction factor model and that each carry their associated errors from the extrapolation step. This is an atypical way to present estimates and determining a more effective way to account for error through this type of analysis should be a priority for future work.

The two extrapolation methods used to produce urban nesting population estimates give markedly different results for both species. For HG in England, produced from the estimated mean was 93,703 AON (33,440 - 302,001[2] AON) from Method 1, compared to 53,006 AON (21,197 - 153,159[2] AON) from Method 2. Results consistently show greater proportions of breeding HG are supported by urban environments with 84.2% (70.9 - 92.6% CL) from Method 1 and 75.1% (60.8 - 86.5% CL) from Method 2. Natural sites now only support an estimated 17,573 AON (13,693 - 23,959 AON). Results also show the vast majority (97.1%) of natural-nesting HG are coastal (within 5 km of the coast); and approximately 74 - 79% (Method 1) or 77 - 81% (Method 2) of roof-nesting HG are coastal.

For LG in England, produced from the estimated mean was 131,042 AON (31,724 - 747,706[2] AON), compared to 64,991 AON (12,199 - 1,282,896[2] AON) from Method 2. Results show that most likely greater proportions of breeding HG are supported by urban environments, with 79.2% (49.5 - 95.3% CL) from Method 1 and 65.4% (27.4 - 97.2% CL) from Method 2. Natural sites now only support an estimated 34,320 AON (32,340 - 36,481 AON). Results here also show that over half (54.7%) of natural-nesting LB are coastal. There are inconsistent results about the proportions of roof-nesting LB that are coastal, 65 - 87% (Method 1) or 54 - 64% (Method 2).

Colonisation of the urban environment since the 1980s was still only in single-digit percentages for Britain and Ireland in 1994 (Raven & Coulson, 1997), and by the turn of the century had grown to c.30% for HG and c.10% for LB in England (Mitchell et al, 2004 adjusted by JNCC, 2020). Although these proportions are likely to be under-estimates, given the inherent detectability issues when surveying the urban environment using GBS (Coulson and Coulson, 2015).

This report reveals dramatic changes in HG and LB breeding populations in England with apparently diverging fortunes in their natural-nesting and roof-nesting sub-populations. New population baselines exist from which conservation efforts and policy outcomes can be monitored at the national scale.

Increased reliance by national populations on the urban environment invites speculation about the origins of colonial recruitment and drivers for diverging population trends, but these aspects of gull demography are not investigated by this report. Whatever the reasons, urban colonisation shown in this report highlights the potential for growing conflicts with human interests (e.g. Spelt et al., 2019) and therefore a need to recognise the changing conservation requirements of both species in the course of their management to resolve such conflicts.

Some population estimates provided here are heavily caveated, accompanied by wide confidence levels and are inconsistent. Correction models and analysis require refinement to reduce propagating errors. In addition, a confounding issue identified in this analysis, which if addressed would assist in producing more precise population estimates, is the reduced sample size of comparable GBS and DAS squares, in particular those survey square with high densities of gulls.

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[2] These are not the mean CLs but refer to the lower CL of the LCL estimated AON and upper CL of the UCL estimated AON

Contents

1 Introduction

Britain and Ireland host internationally important numbers of breeding seabirds along their coasts, also inland at freshwater bodies and moorland sites; and in recent decades increasingly in towns and cities. Established in 1986, the Seabird Monitoring Programme (SMP) is an annual monitoring programme of 25 breeding seabird species. It aims to provide representative sample data to inform breeding abundance trends at regional and national scales; and periodically through more coordinated censuses to estimate breeding population abundance. During the last census, *Seabird 2000* (1998-2002), the UK held internationally significant numbers of breeding Herring Gull (HG) and Lesser Black-backed Gull (LB), approximately 17% and 65% of the biogeographic population, respectively[1].

Annual monitoring by the SMP, using standardised methods (Walsh et al., 1995), provides reliable indications of natural (rural) HG and LB nesting population abundance trends. These trends have indicated long-term declines for both these species (JNCC, 2020a), which contributed to HG being red-listed in BoCC4 (Eaton et al., 2015). LB was amber-listed in BoCC4, however, this decision was not due to natural population declines and instead is an indication of its localised colonial distribution vulnerability and its international status (Eaton et al., 2015).

There is currently no reliable equivalent data via the SMP for the urban (roof) nesting sub-population of HG and LB, due to issues with using traditional survey methods in urban environments, and even without this, the sample size of colonies surveyed is insufficient to calculate population trends (JNCC, 2020a). It is not safe to assume that natural and urban nesting gull population trends are harmonised, neither in terms of the direction nor magnitude of change. To date, censuses have been the only viable approach for obtaining national breeding population estimates of urban nesting HG and LB.

Published national population estimates of HG and LB from the *Seabird 2000* census (Mitchell et al., 2004) are thought to be unreliable due to under-estimates of roof-nesting gulls that resulted from gaps in coverage and from detectability issues using the vantage point methods (identified by Coulson & Coulson, 2015).

There are inherent detectability issues with all GBS methods in the urban environment, which poses a challenge for attempts to produce accurate urban nesting gull population estimates (Coulson and Coulson, 2015; Ross et al., 2016). These detectability issues are not consistent either, for example, gull nests located on high buildings in an inner-city landscape are shown to have a lower detectability rate than those on lower residential buildings. Use of high vantage points or aerial surveys have been shown to improve or overcome these issues (Ross et al., 2016). However, at large spatial scales, given the extent of the current distribution of urban nesting HG and LB in England (Balmer et al., 2013), gathering sufficient data from sampling through this method alone would be prohibitively expensive.

To find suitable solutions for producing urban nesting gull estimates, an ‘urban gull sub-group’ was formed comprising representatives from the *Seabirds Count* steering group[2]. With advice from the sub-group, a Defra commissioned BTO in 2018 and 2019 to develop species-specific correction models to account for variable GBS detectability. This study was published (Woodward et al., 2020). Supported by NE funding, JNCC then expanded this research in 2020 (Burnell, 2021). The sub-group was also instrumental in producing a new survey methodology and sampling regime. The latter was based upon the recommendation for a digital aerial survey (DAS) sampling regime by Thaxter et al., (2017).

The new methodology and sampling regime were applied the urban gull census and this equated to the selection of >4,600 random 1 km urban squares in England from four urban fabric strata. The census commenced in 2019 but by the end of that season only about 10%

of the required number of survey squares had been surveyed by volunteers, an inadequate sample for producing meaningful population estimates for the two species.

Meanwhile, prior to 31 December 2019, the lethal control of HG and LB in England was permissible under various types of licences that, for certain purposes, species and age-classes, were unrestricted in the number of nests, eggs and gulls that could be controlled (Defra, 2019). A stricter licensing regime for these two species was introduced from 2020 for various reasons and one of these was the absence of reliable national population estimates of HG and LB, except for awareness that, at least, natural-nesting populations of these two species had significantly declined (*pers comms* Natural England, 2020).

With a more urgent need for up-to-date gull population estimates to underpin large gull licensing reforms introduced in England in 2020, Defra funded the acceleration of the urban gull survey. This led to professional contractors successfully completing GBS of an additional 3,796 urban 1 km squares and of these a complimentary 99 squares also received DAS. Added to previously surveyed squares in 2018 & 2019, this increased sample sizes enough to allow analyses at the England scale.

This report presents up-to-date breeding population estimates for HG and LB in England plus estimates of the rural (natural nesting) and urban (roof-nesting) sub-populations, and these split and expressed as either inland or coastal nesting. Natural nesting population estimates are derived from SMP data with refinement of previous estimates to allow direct comparisons. Urban population estimates draw on *Seabirds Count* data, enabled through the Defra uplift in 2020, and are produced from two methods of extrapolation up to an England scale and corrected using species-specific correction models. The details, limitations and strengths of these methods, along with recommendations for improving their reliability, are discussed.

[1] Estimates of biogeographical population are different to those stated in the online SMP report and the official *Seabird 2000* publication, as they are based on the corrected totals found in the results of this report.

[2] The urban gull sub-group comprises of representatives from eight conservation bodies: Joint Nature Conservation Committee (JNCC), Natural England (NE), Natural Resources Wales (NRW), NatureScot, Department of Agriculture, Environment and Rural Affairs (DAERA), Royal Society for the Protection of Birds (RSPB), British Trust for Ornithology (BTO) and BirdWatch Ireland

2 Methods

2.1 Rural ('natural-nesting') populations

A HG or LB natural-nesting site is anywhere that is not on top of elevated urban fabric. Typical natural-nesting sites include cliffs, moorland, agricultural land, freshwater margins and islands. Natural-nesting sites can exist within a townscape on unbuilt ground, for example, a site cleared for development.

2.1.1 Data collection and validation

Since 1986, AON data from natural-nesting HG and LB, from a sample of colonies, have been collected to calculate interannual 'Index of Abundance' trends (JNCC, 2020a). Since 2015, more comprehensive coverage of natural-nesting sites has taken place as part of *Seabirds Count* - the fourth breeding seabird census for Britain and Ireland. Volunteer and professional surveyors follow standardised species specific methods, outlined in the Seabird Monitoring Handbook for Britain and Ireland (Walsh et al., 1995). Count data collected from known breeding sites and newly established colonies are entered onto the Seabird Monitoring Programme's (SMP) online database.

To calculate population estimates for England, HG and LB data were extracted from the SMP online database. The dataset was then filtered to remove sites with names pertaining to urban environments such as (but not limited to) those containing the words: city, town, factory, industrial, street, roof (tops) etc. Remaining data were then mapped onto a shapefile consisting of polygons of urban areas in England (Office for National Statistics, 2017). Any additional colonies located within urban areas were then removed. The remaining colony sites were then sense checked to remove any residual urban sites.

To produce site by site comparisons between censuses, the data were split by species and sample year; with records from 1998-2002 (*Seabird 2000*) and from 2015-2020 (*Seabirds Count*) creating four census datasets. The SMP database holds both the accepted counts for *Seabird 2000* as well as counts that are part of annual monitoring. A matching exercise was completed to ensure that only the correct *Seabird 2000* counts were extracted from the SMP dataset for comparison with *Seabirds Count* records. Sites excluded from this exercise that existed in either the *Seabird 2000* or *Seabirds Count* datasets were kept aside for use in the final analysis. Sites counted after the *Seabird 2000* period but had yet to be counted in *Seabirds Count*, and therefore will not have been picked up in either census datasets, were extracted from the full SMP dataset. These formed the final datasets.

Most natural-nesting sites are only visited once during the census period, however, there are exceptions to this. The data relating to *Seabirds Count* were, therefore, given an accuracy score to ensure the most appropriate count was used in the analysis. The accuracy scores were based on preferred unit, correct method-unit combination, count accuracy (if it was an estimate or accurate count) and if the survey occurred within the preferred date range. These rules followed the same process used for selecting *Seabird 2000* records. For sites where there were multi-year visits, the count data with highest accuracy score was used; if all records had the same accuracy score the most recent count was used. For sites where multiple visits were recorded in one season, it was first established if these were additive (e.g. a cumulative count based on separate sections along a section of coast) or a series of visits. If additive, these were then summed and used as a single record for the site. If the counts were a series of full count visits, the highest count was used if it fell within the specified date ranges, if not the record with highest accuracy was used.

For accepted records that expressed counts as IND, instead of the preferred AON (or AOT), an adjustment of 0.5 x IND was applied; this adjustment was also used in the previous census. Any site within 5 km of mean high water mark was assigned as ‘coastal’, which is consistent with SMP and *Seabird 2000* (Mitchell et al., 2004).

2.1.2 Data analysis

For each comparable census site, a percentage change from *Seabird 2000* to *Seabirds Count* was calculated and, from that, annual percentage change (% change / number of years between the two counts) was derived for each site. The mean of annual % change, with 95% confidence limits (CLs), was then calculated for eight regions of England: North East, North West, East Midlands, East Anglia, London, South East, South West and West Midlands. This method assumes linear change, which although in reality is unlikely, given good survey coverage of approximately 80% (HG) and 96% (LB) of populations, the effect of this linear assumption would be negligible.

For sites where there had been no count since *Seabird 2000*, an estimate of current AONs was made by multiplying the number of years between the *Seabird 2000* count year and 2020 by the respective regional mean annual % change (from the calculation above), and then multiplying the *Seabird 2000* count by % change over the whole period. Calculations were also made using the upper and lower CLs for the mean annual % change to produce upper and lower CLs for the site.

For sites where colony counts exist between, but not during either census, a similar calculation to that above was used with slight adjustments by using the nearest corresponding counts. For example, a new site (not surveyed during *Seabird 2000*) may have had a count in 2008 but had yet to be surveyed for *Seabirds Count*. In this situation, the number of years since *Seabird 2000* and number of years to *Seabirds Count* was multiplied by the regional average % change to extrapolate over those years forward and backward in time to estimate site abundances, which then contribute to overall natural nesting population estimates.

To produce up-to-date natural-nesting population estimates, all mean estimated and actual *Seabirds Count* data were summed. Additionally, the actual *Seabirds Count* data and all summed with upper and lower CL to produce estimates with 95% upper and lower CLs for the overall England population and the inland and coastal sub-populations.

2.2 Urban (‘roof-nesting’) populations

Urban and roof-nesting gulls have been interchangeable terms since at least Mitchell et al. (2004) and is followed here that ‘urban’ is defined by Natural England as “all man-made (non-natural) habitats, including, but not limited to, buildings and other structures found in villages, towns, cities and industrial land” (BL Ecology unpubl., 2020). Classification of urban strata was based on the CORINE dataset (CLC, 2012) a full description of the process is presented in Woodward et al. (2020).

2.2.1 Data collection and validation

Successfully trialled in 2018, a new survey methodology and design were implemented to the census from 2019 to provide the necessary data for the new correction factor and extrapolation models to operate and thus to calculate population estimates more accurately. An annotated example of the GBS recording form can be found in Appendix 1. A stratified random sample of the total 30,256 urban 1 km squares (those with more than 2% urban fabric cover in the square) was selected in England, which unintentionally excluded the Isles of Scilly. Stratification was based on county and four urban habitat strata:

- **Sub**: suburban, discontinuous, essentially residential urban fabric;
- **Ind:** industrial, commercial or transport urban fabric types;
- **Ind/Sub mix**: industrial/suburban mix, a relatively even mix of 'Sub' and 'Ind'; or
- **Most Urb**: mostly urban, high density and continuous urban fabric comprising a mix of urban fabric types;

A detailed explanation of the process for delineating these strata is presented in (Woodward et al., 2020).

To ensure that at least one square of each urban stratum was selected in each county, a sample of approximately 15% of the available urban squares, in each stratum and each county was selected randomly equating to 4,629 1 km squares to survey in England. Data were collected from 4,075 of the selected 1 km urban squares in England during the 2019 and 2020 breeding seasons, by both volunteer and professional surveyors, respectively.

Surveyors in 2020 submitted their records via an online form and records were checked regularly to enable feedback of any potential discrepancies or errors to surveyors. Data from surveys conducted in 2019 were submitted on a recording sheet and either scanned, mailed, or emailed to JNCC. All survey data received by JNCC were checked for any potential errors. Validation included the following checks – i) that totals across the count types were cumulative, e.g. if 2 AON were recorded there must be at least 2 AOT); ii) the proportion of each square that was un-surveyed was calculated correctly as relating to potential urban breeding habitat, not natural habitats, or habitat with no nesting potential, e.g. the sea or open freshwater. Adjustments were based on surveyors' survey form comments, or surveyors were contacted directly for information.

The recorded estimated percentage of urban area in each square that could not be surveyed was used to adjust counts up to 100% coverage of the urban area within each surveyed square. The amount of urban fabric not surveyed is not the amount of roof space that could not be viewed when surveying each building; instead, the correction models attempt to account for this issue. Also, the estimated percentage of urban area surveyed is not the same as public accessibility, since it was acceptable to conduct the GBS from any reasonable distance. Despite this, survey coverage was a common issue, for instance within attempts to survey central buildings on MOD sites. Each square with less than 70% survey coverage of its urban fabric were removed from subsequent analysis, since this was deemed to potentially add too much error. After this screening out exercise, a total of 3,872 1 km survey squares were available for analysis. The correction model analysis screened out squares with less than 50% coverage to maximise sample size, which testing found not to cause issues for that analysis step.

2.2.2 Method 1: data exploration and analysis

Both the correction factor study report (Burnell, 2021) and Woodward et al. (2020) found that IND count data from GBS was the most reliable unit to produce realistic AON estimates, as determined by comparison to DAS data. All statistical analysis and extrapolation were conducted using the statistical coding programme R (R Core Team, 2018).

Through exploration of the data, in addition to the already assigned urban strata and proximity of a survey square to the coast, the proportion of each square covered by urban fabric (PROP) (low ≤50% coverage, high >50%) suggested this too was an important factor that influenced the IND counts of both gull species. Strata and PROP were also found to be important during the analysis for the correction factor model, which is not unexpected since both factors were anticipated to influence gull nesting potential. To account for influences by combinations of various factors, species-specific 'groupings' were created. These comprise different combinations of urban strata and proximity to the coast). Each survey square was assigned to a species-specific ‘grouping’; see Table 1 and 2. This further stratification, beyond the county-stratum stratification from the random selection of survey squares, would usually be produced prior to survey and feed into the survey design. However, up to date information on how these factors affected the population and distribution of the species was not apparent until the survey had been completed.

Further exploration of the data indicated geographical variation in abundance. To account for this and to continue to ensure that each grouping had sufficient sample sizes remained, the sample survey data were assigned to courser geographical areas. For HG this was into ‘areas’ - North, South and Midlands; and for LB this was into ‘regions’ - North East, East-England, London, South East, South West and North West. Since the original sampling regime design selected 15% of each urban stratum from each county, this additional geographical stratification was sufficiently captured; see Tables 3 and 4.

Table 1. Herring Gull groupings for method 1 extrapolation up to England level.

Group number	**Herring Gull grouping** (number of squares in the sample)
1	Suburban & Inland (2,671)
2	Suburban, Coastal & high urban density (343)
3	Suburban, Coastal & low urban density (171)
4	Industrial/Suburban mix & Inland (209)
5	Industrial/Suburban mix, Coastal & high urban density (66)
6	Industrial/Suburban mix, Coastal & low urban density (21)
7	Industrial & Inland (151)
8	Industrial, Coastal & high urban density (12)
9	Industrial, Coastal & low urban density (18)
10	Most Urban & Inland (171)
11	Most Urban, Coastal & high urban density (9)
12	Most Urban, Coastal & low urban density (30)

Table 2. Lesser Black-backed Gull groupings for method 1 extrapolation up to England level.

Group number	**Lesser Black-backed Gull grouping**
1	Inland & High urban density (1,104)
2	Inland & Low urban density (2,098)
3	Coastal & Low urban density (430)
4	Industrial, Coastal & high urban density (21)
5	Most Urban, Coastal high urban density (30)
6	Suburban, Coastal & high urban density (171)
7	Industrial/Suburban mix, Coastal & high urban density (18)

Table 3. Herring Gull areas and the contributing counties to those areas, based on the SMP administrative areas. Numbers of survey sample squares in each area are given in brackets.

Area (numbers of squares in the survey sample)	**Contributing counties**
South (1,431)	Gloucestershire, Wiltshire, Somerset, Dorset, Avon, Berkshire, Surrey, Cornwall, Devon, Greater London, Oxfordshire, Buckinghamshire, East Sussex, West Sussex, Kent, Hampshire, Isle of Wight.
Midlands (1,489)	Lincolnshire, Nottinghamshire, Northamptonshire, Leicestershire, Derbyshire, Hereford and Worcester,

	West Midlands, Warwickshire, Staffordshire, Shropshire, Norfolk, Suffolk, Cambridgeshire, Bedfordshire, Hertfordshire, Essex.
North (952)	Northumberland, Cleveland, Tyne and Wear, Durham, Cumbria, Merseyside, Lancashire, Cheshire, Greater Manchester, North Yorkshire, Humberside, West Yorkshire, South Yorkshire.

Table 4. Lesser black-back gull regions and the contributing counties to those areas, based on the SMP administrative areas. Numbers of survey sample squares in each area are given in brackets.

Regions (number of squares in survey sample)	**Contributing counties**
East England (1,071)	Lincolnshire, Nottinghamshire, Northamptonshire, Leicestershire, Derbyshire, Norfolk, Suffolk, Cambridgeshire, Bedfordshire, Hertfordshire, Essex.
North East (587)	Northumberland, Cleveland, Tyne and Wear, Durham, North Yorkshire, Humberside, West Yorkshire, South Yorkshire
North West (365)	Cumbria, Merseyside, Lancashire, Cheshire, Greater Manchester.
South East (907)	Berkshire, Surrey, Greater London, Oxfordshire, Buckinghamshire, East Sussex, West Sussex, Kent, Hampshire, Isle of Wight.
South West (418)	Gloucestershire, Wiltshire, Somerset, Dorset, Avon, Cornwall, Devon.
West Midlands (418)	Hereford and Worcester, West Midlands, Warwickshire, Staffordshire, Shropshire.

To produce England scale population estimates, Method 1 extrapolated this from a random sample and replacement process, known as ‘bootstrapping’. Firstly, for each species and square, GBS IND counts were adjusted to AON counts using the species-specific correction factor model (Burnell, 2021) using the predict function (R Core Team, 2018). These corrected estimated mean AON counts, and their corresponding confidence limits (CLs) for each survey square were then used for the bootstrapping process.

Bootstrapping randomly sampled survey square results from ‘area-groupings’ (HG), or ‘region-groupings’ (LB), with replacement up to the cumulative number squares within each of those groupings in England. The mean estimated AONs were then summed and the process repeated 999 times. This process was also performed separately on the estimated AON lower and upper CLs. This step produced three datasets for each species, each with 999 estimates of urban gull AONs in England. These datasets were then sorted in ascending order and the 499th (median) and the 25th and 975th (2.5 and 97.5th percentiles) reported for each. This entire process was also conducted separately on the inland and coastal filtered datasets and, therefore, those estimates may not sum to the whole England estimate.

Propagating errors across the two-step process was challenging, and the method presented here is only a partially effective compromise. In the correction model analysis (Burnell, 2021;

Woodward et al., 2020) the simulate function (R Core Team, 2018) was used to capture error from the correction step. This function does not exist when predicting from a new dataset. Time and computing constraints prohibited this analysis from exploring other options further so resolving this is recommended as a priority area for future analysis, including re-estimations of populations using these data.

2.2.3 Method 2: data exploration and analysis

Given the substantial sample size of the urban dataset, a modelled approach to the distribution and abundance of GBS IND counts of HG and LB was also explored.

Several other potential environmental variables were mapped and calculated for each square (all measured in metres) – i) distance from coast, ii) distance to nearest freshwater source, iii) distance to nearest known gull colony; and iv) distance to nearest confirmed breeding (observed during survey) that wasn't itself part of the survey. Exploration of these environmental variables suggested that, for both species, relationships exist between the IND and each of these factors. In addition, v) region, vi) PROP and vii) urban strata assigned to the square were also important variables. For some of these variables, relationships differed between inland and coastal squares.

Two types of regression models were tested to find an appropriate method to predict IND counts based on these environmental variables. The MASS (Ripley et al., 2020) and DHARMa (Hartig and Lohse, 2020) packages were used to produce and test the validity of negative binomial (NB) regressions, respectively. However, given the large number of zeros in the GBS IND dataset (over 50% of all responses) it was decided zero-inflated negative binomial (ZINB) regressions were also worthwhile exploring. The package pscl (Jackman et al., 2020) was used to produce and carry out validation tests on ZINB regressions. ZINB were chosen over hurdle models due to the existence of two processes for zeros being present in the dataset – either birds were not detected (sampling error), or there was genuine absence of birds in the square (structural) (Hu et al., 2011).

The top models of each regression type were chosen based on the root-mean-square error (RMSE) and Akaike information criterion (AIC) values. Additionally, a comparison of the observed total count of IND and the model predicted total IND count was used. Once the best models were decided for each type of regression, the most appropriate was chosen using several comparative tests, described in the next section.

Once the best model was selected for each species, it was then used to predict the IND counts for every urban square across England (30,256). Using the predict function (R Core Team, 2018) GBS IND counts for each species for produced for each square in the dataset. Using the boot package (Canty and Ripley, 2021), with a strata argument for inland and coastal, these predictions were bootstrapped 999 times. Producing 999 estimates of IND for each of the 30,256 squares. Each predicted IND was then corrected to AON drawing from the species-specific correction model, once again using the predict function to produce a mean AON estimate and lower and upper CLs for each of the 999 IND values for each of the 30,256 squares. Each square's mean AONs, were then summed to produce 999 estimates for England overall, and the process was repeated for inland and coastal squares. These were sorted in ascending order and the 499th and 25th and 975th were presented (median, 2.5th and 97.5th percentiles). This process was also performed separately on the estimated AON lower CLs and upper CLs for each species. This then shows error ranges from both the extrapolation and correction steps. The same self-assessment conclusions and recommendations as those given in Method 1 above also apply to Method 2.

3 Results

Although the number of the natural-nesting sites surveyed were different between censuses, due to the establishment of new colonies and desertion of others, or absence of survey data during census periods, all known colonies were surveyed using standardised methods. Also, since calculations that account for the dynamic nature of colonies and data gaps were relatively straightforward, comparisons between sets of adjusted census results of natural-nesting HG and LB populations are considered to be reliable and accurate.

Urban-nesting gulls present considerable challenges and censuses have been inconsistent in their approaches and in their success at addressing these. As a result, direct comparison between *Seabird 2000* and *Seabirds Count* urban-nesting and overall population estimates are unreliable.

3.1 Rural ('natural-nesting') population estimates

Adjusted natural-nesting population estimates from the *Seabird 2000* period (Table 5 and 6) were produced from counts at sites used in the original dataset, plus counts at additional sites made available after publication of the original results. Clerical errors discovered by this current study in the original dataset had the effect of considerably increasing the LB count and consequently decreasing the HG count. This was due to the original counts of a large site being assigned to the wrong species.

Seabirds Count is incomplete and not all known gull colony sites have been surveyed. As a result, estimates presented for HG in this study have broader CLs, compared to those associated with the LB estimate. This is due to lower survey coverage for HG, with c.80% of the population being surveyed, compared to c.96% of LB population, in England.

The natural-nesting populations of HG (-38%) and LB (-45%) in England were found to have significantly declined since *Seabird 2000* (Table 5 and 6).

The distribution of natural-nesting HG has remained predominantly coastal with >97% in both censuses. However, the proportion of coastal natural-nesting LB has dropped from 70% to 55% between censuses and this appears to be due to proportionately greater declines in coastal colonies compared to those located inland. England's largest LB colony is inland at Bowland Fells SPA, where SMP counts of 18,518 AON in 2001 and 14,627 AON in 2018 dominate the overall inland estimate.

Table 5. Estimated natural nesting Herring Gull breeding abundance (AON) for two breeding seabird censuses, *Seabird 2000* (1998-2002) and *Seabirds Count* (2015-present) for: inland and coastal English colonies and the total population.

Survey	Inland (AON)	Coastal (AON)	**TOTAL (AON)**
Seabird 2000 (adjusted)	733	27,617	**28,350**
Seabirds Count (with CL)	504 (393-635)	17,069 (13,299-23,324)	**17,573 (13,693-23,959)**

Table 6. Estimated natural nesting Lesser Black-backed Gull breeding abundance (AON) for two breeding seabird censuses, *Seabird 2000* (1998-2002) and *Seabirds Count* (2015-present) for: inland and coastal English colonies and the total population.

Survey	Inland (AON)	Coastal (AON)	**TOTAL (AON)**
Seabird 2000 (adjusted)	18,947	43,744	**62,691**
Seabirds Count (with CL)	15,533 (14,837-16,486)	18,787 (17,503-19,995)	**34,320 (32,340-36,481)**

3.2 Urban ('roof-nesting') population estimates

3.2.1 Method 1 results

Method 1 used a more traditional process of 'bootstrapping' to extrapolate population estimates, with an initial step to convert GBS IND counts into estimates of 'true' AON counts from the correction models. This two-step process means propagating error through both steps is difficult, so for transparency and completeness, three sets of three population estimates were produced using the predicted mean AON and the lower and upper CLs, with the 95% CLs from the extrapolation for each set of separate calculations performed to produce estimates for the inland, coastal and overall England populations of both species, presented in Table 7 and 8.

Table 7. Method 1 urban-nesting **Herring Gull** population estimates for inland, coastal and all 1 km urban squares in England. Totals shown for England are not the sum of inland and coastal since these were calculated independently. CLs refer to the confidence around the correction from IND counts to AON, not from the extrapolation method.

	Lower CL predicted AON	Mean predicted AON	Upper CL predicted AON
Inland (24,156)	9,177 (8,569 - 9,804)	23,948 (22,213-25,898)	59,245 (53,649 - 65,293)
Coastal (6,100)	26,194 (24,403 - 28,181)	69, 615 (63,754 - 75,690)	215,735 (190,490 - 241,009)
All England (30,256)	35,407 (33,440 - 37,179)	93,703 (87,442 - 99,619)	274,676 (249,252 - 302,011)

Table 8. Method 1 urban-nesting **Lesser Black-backed Gull** population estimates for inland, coastal and all 1 km urban squares in England. Totals shown for England are not the sum of inland and coastal since these were calculated independently. CLs refer to the confidence around the correction from IND counts to AON, not from the extrapolation method.

	Lower CL predicted AON	Mean predicted AON	Upper CL predicted AON
Inland (24,156)	12,385 (11,343 - 13,486)	31,687 (29,077 - 34,516)	77,435 (70,364 - 84,836)
Coastal (6,100)	22,688 (19,181 - 26,508)	98,572 (78,184 - 119,692)	510,613 (376,981 - 649,942)
All England (30,256)	35,266 (31,724 - 39,634)	131,042 (109,904 - 154,438)	594,066 (458,375 - 747,706)

This method produced wide confidence limits for both species, which are likely driven by a combination of correction model fit and that some of the groupings had small sample sizes.

3.2.2 Method 2 results

Model 2 used an extrapolation method based on a modelled approach of abundance and distribution. To ensure that uncertainty around the correction factor models were captured appropriately, GBS IND were modelled rather than applying the correction factor first. The models tested the hypothesis that GBS IND counts could be predicted by a combination of environmental and geographical predictors. To model GBS IND counts as a function of a suite of explanatory variables, both negative binomial (NB) with log link functions and zero-inflated negative binomial (ZINB) GLMs were fitted for each species.

From exploration of the data, five continuous (cn) and two categorical (ct) variables were taken forward for modelling:

1. Distance to coast (cn), obtained through measuring the distance from the centre of square to the mean high water mark using the OSM mean high water shapefile (OpenStreetMap, 2019);
2. Distance to nearest known gull colony (cn), obtained through mapping the known gull gull colonies in the SMP online database (Seabird Monitoring Programme, 2020);
3. Distance to nearest confirmed breeding square, AONs recorded within this survey, but not itself (cn);
4. Distance to nearest freshwater source (cn) obtained through the Land Cover Map 2015 shapefile (Rowland et al., 2007) which is both standing open water sources, rivers and canals;
5. Proportion of square covered in urban fabric, PROP (cn) obtained through the CORINE data classes and refers to the amount of the square covered by all urban classes (CLC, 2012);
6. Urban fabric type, strata (ct);
7. Region of England the square is situated (ct) explained above for the LB regions (Table 4).

It was also noted that for some of these the relationship between them and the GBS IND count was different between inland and coastal squares.

The R packages MASS (Ripley et al., 2020) and DHARMa (Hartig and Lohse, 2020) were used to produce NB models with the above explanatory variables for both species. The best performing models i.e. with lowest AIC and RMSE values, were validated using various tests, the outputs from which can be found in Appendix 2. These validation steps suggested potential over-dispersion in the HG model residuals with possible significant deviation away from the assumed distribution also apparent in the residuals. LB model residuals however, appeared to show promising relationships with no apparent over-dispersion or deviation away from distribution and no apparent zero-inflation.

Although no significant zero-inflation was found in either model for either species, zero-inflated models have been shown to improve model predictions and fit in other count based studies (Littlewood et al., 2019; Lyashevska et al., 2016; Zipkin et al., 2014). Additionally, the high number of zero counts for HG (80%) and LB (86%) in the GBS IND datasets suggested ZINB models were worthwhile exploring. To fit these the pscl package (Jackman et al., 2020) was used. However, checks and validation steps were taken to compare the two models and ensure the best fitting model was taken forward.

The favoured ZINB model, the results for which can be found in Appendix 3, equations only differ from the favoured NB through the addition of model for zero counts, therefore the two models NB equations are the same. The best performing ZINB models (Figure 1), lowest AIC and RMSE, were then tested against the best fitting NB model for each species.

Results from the chi-squared likelihood ratio tests and the Vuong tests, suggest the ZINB performed better than the NB for both species (Table 9). Likewise, model predicted IND counts from ZINB were closer to the observed compared to those derived from NB models (Table 10). Similarly, the number of zeros predicted by the ZINB models were closer to the observed than those predicted by the NB models (Table 10).

Herring Gull Zero-inflated model

Count model (Neg -bin): Individuals ~ log(Dist' from coast+1) * Inland.Coastal **+** Region*PROP **+**
log(Dist' to nearest confirmed breeding square)*Inland.Coastal **+**
log(Dist' to nearest gull colony +1)*Inland.Coastal **+**
log(Distance to nearest freshwater source +1)*Inland.Coastal **+** Strata

Zero-inflation model: log(Distance from coast+1) **+** PROP **+** log(Distance to nearest gull colony +1) **+**
log(Distance to nearest confirmed breeding square)

Lesser Black-backed Gull Zero-inflated model

Count model (Neg-bin): As above.

Zero-inflation model: log(Distance from coast+1) **+** Proportion of square covered in urban fabric **+**
log(Distance to nearest gull colony +1) **+**
log(Distance to nearest confirmed breeding square) **+**
log(Distance to nearest freshwater source +1)

Figure 1. Favoured ZINB model equations for predicting IND counts of herring and Lesser Black-backed Gulls in urban 1 km squares in England. Due to the structure of ZINB the Neg-bin models stated in these equations are also those of the favoured NB.

Table 9. Comparative test results between ZINB and NB models for HG and LB.

Species	X^2 Likelihood ratio test statistic (p-value)	Vuong test z-statistic (p-value) *(where model one in the test was the ZINB*)*
Herring Gull	396.33 (<0.001)	AIC – 9.41 (<0.001) BIC – 8.65 (<0.001)
Lesser Black-backed Gull	188.63 (<0.001)	AIC – 6.73 (<0.001) BIC – 5.30 (<0.001)

** Vuong z-statistic is dictated by the position of model in the coding, if the statistic is positive and significant the preferred model is model 1. In this case model 1 in the coding was the ZINB models.*

Table 10. Comparisons of IND counts and number of zeros between those observed in the dataset and the NB and ZINB model predicted results for HG and LB.

	IND counts for survey squares			Number of zeros in dataset		
Species	Observed	NB predicted (95% CL)	ZINB predicted (95% CL)	Observed	NB predicted	ZINB predicted
HG	10,802	20,137 (9,887 - 42,065)	11,010 (5,599 - 21,032)	3115	3075	3129
LG	5,142	6,396 (2,729 - 14,795)	4,871 (1,960 - 10,184)	3335	3317	3333

These results suggested that ZINB models were more appropriate than NB models, for both species, for predicting IND counts in urban squares. Although these were accepted as a means of extrapolation, the models were not free from error and for both species ZINB models did not perform well for squares with higher observed counts of gulls (Figures 2 and 3). Additionally, some square estimates, particularly those with smaller observed counts, had large confidence limits around predicted IND counts (Figures 2 and 3). Interrogation, of the data suggests these issues could stem from inadequate sample sizes in some ‘groupings’ of data and also from the relatively few squares that held >100 IND gulls for HG (n=18) and LB (n=6). Therefore, statistical power to predict these large counts is reduced.

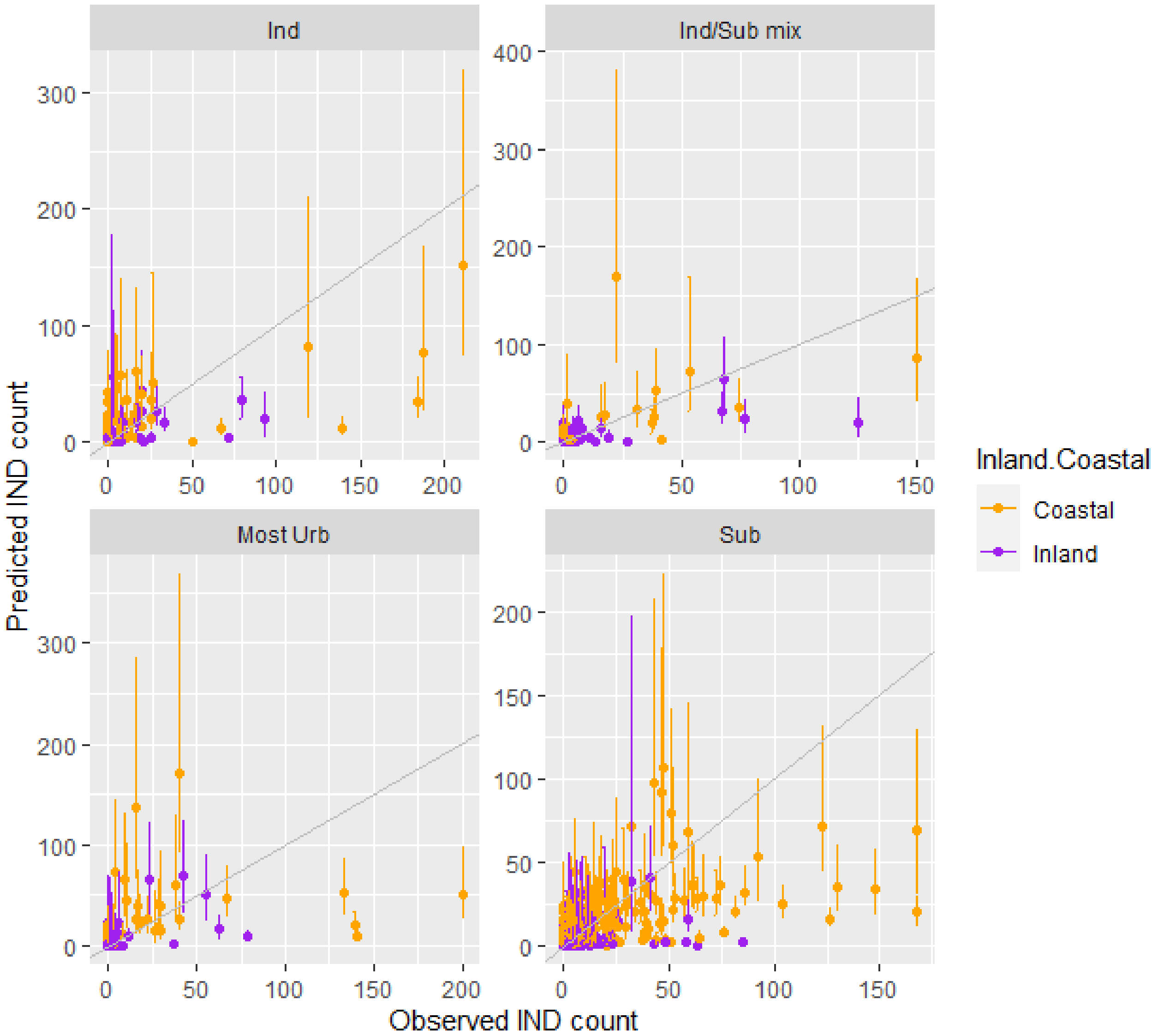

Figure 2. Comparison of observed and predict Herring Gull IND counts from the favoured ZINB model, with associated 95% CLs, split by the urban strata and coloured by the squares location coastal (orange) or inland (purple). X=Y line added for guidance.

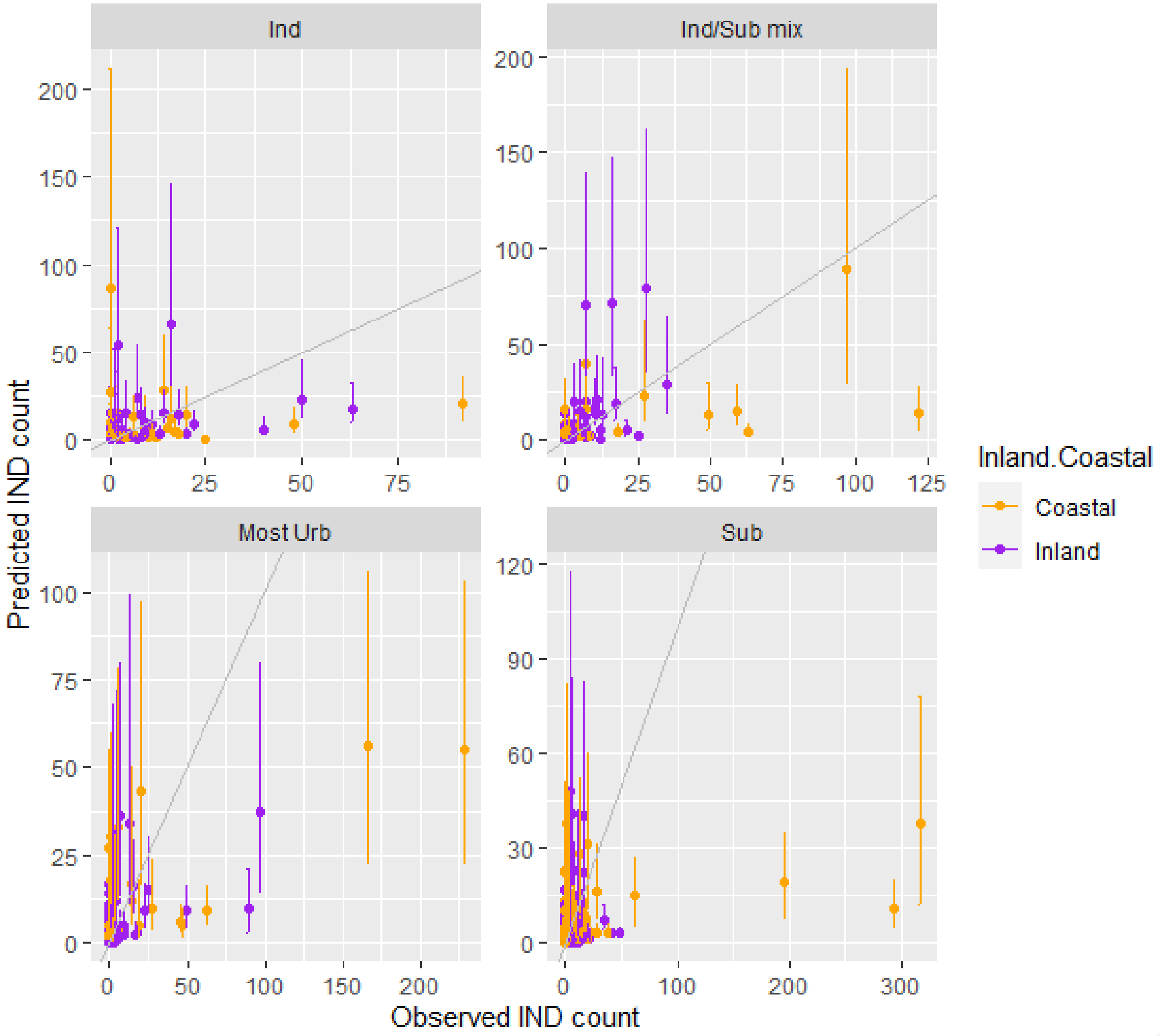

Figure 3. Comparison of observed and predicted Lesser Black-backed Gull IND counts from the favoured ZINB model, with associated 95% CLs, split by the urban strata and coloured by the squares location coastal (orange) or inland (purple). X=Y line added for guidance.

To capture the error around the ZINB model predicted INDs for each species, the predicted IND (obtained through the predict function (R Core Team, 2018)) was bootstrapped 999 times, meaning that for each 1 km square there were 999 IND estimates; see Section 2.2.3 above for details about next steps in the analysis to produce three sets of estimates for each species. Overall estimates for England and for inland and coastal sub-populations are presented in Table 11 for HG and Table 12 for LB. The spatial distribution of the predicted mean estimated AON, based on the predicted mean IND produced by the ZINB models, for each species can be found in Figures 4 and 5.

Table 11. Method 2 urban-nesting **Herring Gull** population estimates for inland, coastal and all 1 km urban squares in England. Totals shown for England are not the sum of inland and coastal since these were calculated independently. CLs refer to the extrapolation error, in this case the modelled IND prediction.

Population area (number of squares)	LCL estimated AON	Mean estimated AON	UCL estimated AON
Inland (24,156)	4,820 (4,613 - 5,066)	12,387 (11,885-12,990)	25,992 (24,644 - 27,972)
Coastal (6,100)	17,434 (16, 439 - 18,509)	41,198 (37,880 - 45,099)	109,031 (95,102 - 126,510)
All England (30,256)	22,264 (21,197 - 23,315)	53,006 (50,266-57,562)	135,145 (121,164-153,159)

Table 12. Method 2 urban-nesting **Lesser Black-backed Gull** population estimates for inland, coastal and all 1 km urban squares in England. Totals shown for England are not the sum of inland and coastal since these were calculated independently. CLs refer to the extrapolation error, in this case the modelled IND prediction.

Population area (number of squares)	LCL estimated AON	Mean estimated AON	UCL estimated AON
Inland (24,156)	6,271 (5,500 - 7,784)	26,383 (17, 361 - 55, 308)	193,966 (59,009 - 850,993)
Coastal (6,100)	7,340 (6,215 - 8,798)	36,516 (23,548 - 59,680)	311,360 (127,656 - 782,289)
All England (30,256)	13,706 (12,199 - 15, 628)	64,991 (46,403 - 98,663)	544,166 (249,146 -1,282,896)

Method 2 improved CLs for HG compared to method 1 although, these are still relatively wide. In contrast, the LB estimates using method 2 have wider CLs than method 1 especially for the upper CL estimated population.

3.3 England population estimates

Tables 13 and 14 present overall current breeding population estimates for HG and LB in England derived from the sum of the natural-nesting estimate with either the Method 1 or Method 2 urban-nesting estimates. Since both methods used to estimate the urban-nesting population involved two-step processes, the full extent of errors generated from both steps are presented as three sets of three population estimates produced from the predicted mean, LCL and UCL summed with their corresponding mean, LCL and UCL estimates from the single method that produced the natural-nesting estimates.

Table 13. Current estimated total population of breeding **Herring Gull** (AON) in England. Calculated by the summation of natural-nesting estimates and each of the two urban-nesting gull estimates from the two methods of extrapolation.

	Natural-nesters (AON)	Urban-nesters (AON)	**TOTALS (AON)**	
			Method 1	**Method 2**
Seabirds Count (2015-2020)	17,573 (13,693 - 23,959)	n/a	**LCL 52,980** (47,133 - 61,138)	**LCL 39,837** (34,890 - 47,274)
Method 1 LCL	n/a	35,407 (33,440 - 37,179)	**Mean 111,276** (101,135 - 123,578)	**Mean 70,579** (63,959 - 81,521)

Method 1 mean		93,703 (87,442 - 99,619)	**UCL 291,837** (262,945 - 325,970)	**UCL 152,718** (134,857 - 177,118)
Method 1 UCL		274,676 (249,252 - 302,011)		
Method 2 LCL	n/a	22,264 (21,197 - 23,315)		
Method 2 mean		53,006 (50,266-57,562)		
Method 2 UCL		135,145 (121,164-153,159)		

Table 14. Current estimated total population of breeding **Lesser Black-backed Gull** (AON) in England. Calculated by the summation of natural-nesting estimates and each of the two urban-nesting gull estimates from the two methods of extrapolation.

	Natural-nesters (AON)	Urban-nesters (AON)	**TOTALS (AON)**	
			Method 1	**Method 2**
Seabirds Count (2015-2020)	34,320 (32,340 - 36,481)	n/a	**LCL 69,586** (64,064 - 76,115) **Mean 165,362** (142,244 - 190,919) **UCL 628,386** (490,715 - 784,187)	**LCL 48,026** (44,539 - 52,109) **Mean 99,311** (78,743 - 135,144) **UCL 578,486** (281,486 -1.319,377)
Method 1 LCL	n/a	35,266 (31,724 - 39,634)		
Method 1 mean		131,042 (109,904 - 154,438)		
Method 1 UCL		594,066 (458,375 - 747,706)		
Method 2 LCL	n/a	13,706 (12,199 - 15, 628)		
Method 2 mean		64,991 (46,403 - 98,663)		
Method 2 UCL		544,166 (249,146 - 1,282,896)		

4 Discussion

4.1 Natural-nesting (rural') population estimates

Estimates from the *Seabird 2000* census period in this report differ from those presented in the annual SMP report (JNCC, 2020a) and from the *Seabird 2000* publication (Mitchell et al., 2004). Clerical errors subsequently found in the original dataset, along with colony counts submitted after the publication necessitated these retrospective adjustments. No CLs are given around *Seabird 2000* estimates since coverage for both species was considered to be adequate. Since the current census, *Seabirds Count*, continues to collect data, overall estimations stated here are given CLs due to the need to calculate estimates at some colonies where counts are yet to be made. It is, therefore, likely that *Seabirds Count* estimates for HG and LB in England will change in the final census publication. Nevertheless, this change is not likely to be large given that coverage was already high in England for HG (80%) and LB (96%) populations.

Survey methods and colony boundary definitions are standardised across the two censuses, leading to robust comparisons between the two surveys for natural-nesting colonies. In many cases, a site by site comparison of counts between the two surveys, could be made. These allowed for average annual change calculations to be provided and ultimately realistic regional annual changes that were then used to predict un-surveyed sites. As such, results stated and comparisons made in this report for the natural-nesting sub-populations of HG and LB in England are considered reliable and robust.

Declines in England's HG (-38%) and LB (-45%) natural-nesting populations since *Seabird 2000* differ in scale from that in the 2018 SMP annual report for HG (-68%), but this is due to these being generated from a different time period of between 1986-2018. Additionally, difference seen between the SMP annual trend and those in produced in this report are likely a result of differences in approaches to calculating change between the two surveys. SMP annual trends up to 2018 were based on a sample of sites that were only included in the trend analysis if they held at least two data points. Since 2018, analysis for trends has required three data points. This means that new sites are not brought into that calculation, nor those that have yet to be surveyed in *Seabirds Count*, and those that have only been surveyed once before. The other important difference is that censuses pool data from several years' worth of surveys rather than from one year, so even though annual trends will be pulling in *Seabirds Count* data, this will not be for any particular year of that survey. From SMP data, the LB trend is less predictable due to very broad CLs (JNCC, 2020a).

The split between coastal and inland natural nesting HG populations in England is consistent between *Seabird 2000* and *Seabirds Count* with a little under 3% nesting inland. Updated *Seabird 2000* data showed c.70% of natural-nesting LB in England were coastal and results here from *Seabirds Count* show this proportion has dropped to c.55%. This finding validates findings of a similar study at the UK scale which used a similar datasets, minus the new *Seabirds Count* data (Nager and O'Hanlon, 2016).

The roles of productivity, recruitment, colony exchange and other demographic functions in the observed change is unclear. The figures presented here, at least, show a greater decline in the coastal sub-population compared to the inland sub-population and, as shown by results in section 3.1 above, the inland population change could alone be explained by modest decline in the huge Bowland Fells SPA population.

Since the overall LB population comprises of fewer and much larger colonies compared to HG, hence its amber-listed status (Eaton et al., 2015), dramatic losses in the coastal population are largely driven by declines in those major colonies, such as at Alde-Ore Estuary SPA (23,400 AON in 2000 cf. 1,953 AON in 2017) and at Morecambe Bay and Duddon Estuary SPA (19,487 AON in 1999 cf. 2,782 AON in 2018) (SMP, accessed online Dec 2020).

4.2 Urban-nesting ('roof') population estimates

In this report two methods for producing urban-nesting HG and LB population estimates are presented. Both methods have their own strengths and weaknesses around their performance and appropriateness for use, described in section 4.2.1 and 4.2.2. The two methods produced considerably different population estimates for both species and with CLs given around the LCLs and UCLs for the two step processes that were necessary in both methods. Method 2 produced less wide CLs for HG than Method 1; by contrast, Method 2 produced wider CLs for LB than Method 1, especially for the upper CL estimate; overall, estimate ranges from both methods are particularly broad; see Tables 7, 8, 11 and 12.

Extracting proportions of inland and coastal urban-nesting HG from the results is confounded by the independency of those calculations, and by each set of results which is presented with CLs around CLs generated from the two-step processes performed in both methods. These issues have the effect that sums of inland and coastal estimates do not equal all England estimates.

For HG, if the sum of mean predicted AON for inland and coastal estimates was assumed to equate to the overall England total, the results from both methods are reasonably consistent indicating that approximately three-quarters of urban-nesting HG are coastal - 74.4% (74.1 - 74.5% CL) from Method 1 and 76.9% (76.1 - 77-6% CL) from Method 2.

For LG, following the same assumptions as above, results from the two methods are inconsistent – 75.5% (72.9 - 77.6% CL) from Method 1 and 58.0% (51.9 - 57.6 % CL) from Method 2. The Method 2 proportions have a similar split seen in the natural-nesting inland and coastal sub-populations of LB but, there is no evidence to suggest that natural-nesting and roof-nesting populations are synchronised in this way and the similarity could be just a coincidental artefact of independent results.

Expansion inland by nesting LB has been seen in the Netherlands and is possibly a response to loss of marine fishery discards and an ability to exploit terrestrial food resources (Gyimesi et al., 2016). *Seabird 2000* data does not allow comparisons to be made with which to infer trends about the inland colonisation of urban-nesting gulls.

Although the two methods produced highly dissimilar results, both methods indicate that urban-nesting sub-population are now larger than their natural-nesting counterparts. There is greater confidence in this assertion for HG where comparison of the LCL for the natural-nesting estimate with the LCL of the LCL estimate for the urban-nesting estimate, suggests that 60.8% (Method 1) or 70.9% (Method 2) of the HG population nests in the urban environment (Table 15). Using mean estimates, at least three-quarters of HG in England are reliant on built structures for nesting.

The picture for LB is less clear; see Table 16. CLs around urban population estimates are particularly wide and indicate from as little as about a quarter to as much as nearly all of the LB population nest in urban environment. Using mean estimates, it seems likely that between approximately two-thirds and four-fifths of LG in England are be reliant on built structures for nesting in England.

Unlike natural-nesting estimates, for reasons explained by Coulson and Coulson (2015), the urban population estimates derived from *Seabird 2000* data cannot be reliably manipulated and re-analysed for comparative purposes.

Table 15. Percentage of **Herring Gull** population nesting in the natural and urban environment, taken from the estimated means of the natural and urban AON from both methods, with the extreme ranges from LCLs and ULCs comparisons expressed in brackets.

Urban population method	Natural-nesting (%)	Urban-nesting (%)
Method 1	**15.8** (7.4 - 29.1)	**84.2** (70.9 - 92.6)
Method 2	**24.9** (13.5 - 39.2)	**75.1** (60.8 - 86.5)

Table 16. Percentage of **Lesser Black-backed Gull** population nesting in the natural and urban environment, taken from the estimated means of the natural and urban AON from both methods, with the extreme ranges from LCLs and ULCs comparisons expressed in brackets.

Urban population method	Natural-nesting (%)	Urban-nesting (%)

Method 1	**20.8** (4.7 - 50.5)	**79.2** (49.5 - 95.3)
Method 2	**34.6** (2.8 - 72.6)	**65.4** (27.4 - 97.2)

4.2.1 Method 1 Limitations and Strengths

Method 1 could be described as the more traditional of the two extrapolation methods. The process has been used successfully for many sample based surveys at UK and England scales to produce population estimates of various bird species (Austin et al., 2007; Banks et al., 2007; Jackson et al., 2006; Rehfisch et al., 2002). Given its more prevalent use, this method could be considered the more reliable approach for this survey.

However, unlike those studies, analysis here needed a two-step process to produce urban-nesting HG and LB population estimates. Error from both steps have been conveyed in this report by producing three estimates, with 95% CLs from the bootstrapping, based on the LCL, mean and UCL estimated AONs from the correction step. More efficient methods for propagating these errors are potentially available e.g. hierarchical Bayesian frameworks.

Extrapolation also experienced problems around its assumption of randomness in the distribution of breeding gulls and number birds in each survey square. Although the creation of ‘groupings’ went some way to alleviate these problems, random allocation from each area-grouping (HG) or region-grouping (LB) into each county-group combination does not properly accommodate the colonial nesting or clustering characteristics of these species. Whilst models discovered a suite of environmental variables that affect distribution and numbers, there are likely to be additional factors with more complex interrelationships than accounted for in the creation of these ‘groupings’, which were limited by the information made available. Even with more detailed information about environmental variables to create finer-scale ‘groupings’, their exploration could become limited due to sample sizes becoming too small for the bootstrapping stage.

There was also the assumption that sampled survey squares within each area-grouping (HG), or region-grouping (LB) were representative with regard to the numbers of gulls present and their urban fabric make-up. Although the urban strata will account for some of the latter, the proportion of urban fabric is more variable. The sample size achieved of 12% per stratum in each county is a reasonable size but this is less than the 15% target minimum below which issues with lack of representativity could manifest. Re-running the analysis with fewer groupings but that are characterised by the most pertinent influential variables may help to reduce CLs produced by this method.

4.2.2 Method 2 Limitations and Strengths

Zero-inflated models have been shown to be a reliable method for incorporating or accounting for over-dispersion and excess zeros in count data (Hu et al., 2011; Martin et al., 2005; Zeileis et al., 2008). Additionally, their use for modelling distribution and abundance as a function of habitat or/ and environmental factors, is thought to be reliable (Boyce et al., 2016). Although not commonly used on breeding seabird data, they have been used successfully in other systems that involve count data (Himsworth et al., 2014; Littlewood et al., 2019; Lyashevska et al., 2016; Maclean et al., 2013). Similar to the findings in Hu et al., (2011) and Martin et al., (2005) this report found that ZINB models performed better than their NB counterparts, based on comparative tests and the reduced confidence around estimated IND counts.

Method 2, although less conventional, appears to have merit for HG, judging by CL widths, but appears less robust for LB. The cumulative effect of wide confidence intervals produced by the correction model and the ZINB IND model are likely the main drivers.

The relative abundance maps presented in this study (Appendix 4, Figures 4 and 5) were produced from Method 2 results. Acknowledging that equivalent maps produced for the Bird Atlas 2008-11 (Balmer et al., 2013) are presented at courser scales (10 km^2), plot breeding in all habitats (not just urban) and derive from data collated about a decade ago, informal comparisons can be made - HG (https://app.bto.org/mapstore/StoreServlet?id=237, and LB - (https://app.bto.org/mapstore/StoreServlet?id=236 (Balmer et al., 2013). Consistencies exists with HG maps in showing localised concentrations in certain coastal areas and where HG has colonised inland. High concentrations of LB are shown in coastal towns and cities, notably inland, for example, in Birmingham, Bristol, Peterborough, Swindon, Scunthorpe and Manchester, which although is shown at a courser scale, roughly matches that shown in the corresponding relative abundance map from Bird Atlas 2008-11.

Similar to the correction model analysis, the ZINB models for both species were not good at predicting the higher density squares (Burnell, 2021). These were few and far between in the survey data which meant that power to predict these was low but, it could also suggest these are in reality infrequent. Further research into identifying environmental variables that better characterise locations where these high concentrations exist should be conducted.

As in Method 1, the two step process in Method 2 created a challenge when trying to propagate that error through the steps, as a compromise this was conveyed in the same way by producing three sets of population estimates using the LCL, mean and UCL AON estimates from the correction step with 95% CLs given for each from the extrapolation step, with ZINB predicted IND error created in Method 2.

There are other attributes of error to consider that may inhibit the effectiveness of the modelled approach. The unit IND used for the modelling has its own associated error since it is the count unit that is most likely to be double-counted, assumed by the inherent mobility of IND adults. The sometimes high proportions of non-breeding IND that may be present in breeding squares (Calladine and Harris, 1997) also confounds this. Squares in close proximity and that are surveyed at different times on different days could be sensitive to this type of error since numbers of IND present could be driven by opportunistically available food sources, an ephemeral factor not adequately explored in this analysis.

4.2.3 Recommendations for Future Analyses

In order to acquire data from at least one survey square from within each of the four urban strata categories, and from within each county, it was calculated that stratified sampling would require c.15% of the 30,256 1 km urban squares (as identified by CORINE, 2012), equating to a total of 4,629 squares for GBS. During 2019 and 2020, only 4,075 squares were surveyed and of these, some lacked sufficient survey coverage of urban fabric (<70%). After these were screened out from the dataset, 3,872 1 km squares remained for analysis. This is 83.6% of the envisaged minimum requirement and represents only 12.8% of urban 1 km squares in England.

Similar to this study, the correction factor analysis also had issues with GBS coverage of some squares, as well as insufficient coverage from DAS (Burnell, 2021). This was a significant issue that reduced the sample from a potential 257 to only 235 directly comparable squares. Of these, very few squares supported high densities of gulls, which inhibited the models performance (Burnell, 2021).

In summary, analysis suffered from numerous confounding factors:

- Small sample sizes of ‘groupings’ (at extrapolation stage) and sets of environmental variables with gull presence (at the correction factor stage);
- Variable and incomplete coverage of survey squares that was needed for the correction factor and population estimation analyses for urban-nesting gulls;
- Overwhelming complexity of influence on gull distribution and upon GBS detectability by finer scale physical characteristics of environment variables that ‘groupings’ were only partially able to account for; and
- Strongly colonial characteristics of nesting gulls, particularly by LB.

Within their limitations, the estimates that derive from the methods used are reliably presented and repeatable. Within this ‘frequentist framework’, there are inevitable challenges to propagating errors when using two-step processes to produce estimates. In a Bayesian framework, error can potentially be accumulated or incorporated as part of the final model. Estimates could be refined and be more effective and efficient in their inclusion of potential error from both steps of the process. It would also be worth considering fitting a model that accounts for spatial autocorrelation. This would hopefully account for error caused by movements of gulls between squares.

In either broadly philosophical statistical approach to this, the acquisition of more data to reduce CLs at one or both steps are desirable. This can be achieved by targeting surveys at squares that are poorly represented in ‘groupings’; or by refining the arrangement of existing environmental predictors in ‘groupings’. Gathering more and finer detailed information about environmental predictors, such as categorisation of urban strata by roof types (material, aspect and slope; chimney presence/ absence) may help. Additional factors, such as distance to likely food sources, may also help to alleviate variations in the model. However, in all instances, additional survey data will be required to avoid over-fitting.

Another primary concern is with correction models and how well they account for variation seen between DAS and GBS counts. With more comprehensive samples, correction models could be further refined and thus improve subsequent estimates. It should also be noted that, regardless of how refined correction models are, field survey methods with inherent weakness in detectability will always produce some level of error.

5 Conclusion

Results in this report are statistically robust in showing that England's natural-nesting populations of HG and LB have declined over the past 20 years since *Seabird 2000*. Decline is particularly significant in coastal natural-nesting LB, which is likely driven by declines in major colonies, including those supported by SPAs. The inland natural-nesting LB population is largely influenced by fortunes of the largest colony in England at Bowland fells SPA. The distribution of natural-nesting HG has remained predominately coastal.

Although the two methods employed to estimate urban-nesting populations produced considerably different results, both clearly indicate for the first time that approximately three-quarters of England's breeding population of HG are now utilising our towns and cities for nesting and that approximately three-quarters of HG in England overall nest in coastal locations. Results suggest that whilst HG colonisation inland has been predominantly into the urban environment, rather than into natural habitats inland. There may exist limiting factors or some other reasons for lack of attractiveness to HG to colonise natural habitats inland.

LB results are less clear to interpretation due to broad CLs and inconsistencies. It is likely that the larger proportion, perhaps two-thirds to four-fifths, of the overall England breeding population now rely on our towns and cities for nesting, but confidence around these proportions is poor. And between perhaps half and three-quarters of urban-nesting LB are coastal.

Due to methodological problems with *Seabird 2000* data that resulted in irreparable under-estimations of roof-nesting HG and LB, direct comparisons with *Seabirds Count* urban-nesting, and therefore also with overall England population estimates, are unreliable.

Care is advised when communicating and utilising populations estimates, proportions and trends. This is especially true for urban population results due to the particularly wide confidence limits produced through the two-step processes of both methods, which allude to uncertainty about robustness of the correction models and to inherent flaws in the survey methodology. However, this report has identified and made sophisticated attempts to address those confounding issues, and has been transparent about these issues expressed through the presentation of errors as CLs.

Population estimates are expected to change as the submission of counts from additional sites continues in England as part of *Seabirds Count*. Population estimates presented in this report can therefore be regarded as provisional in respect of the current and ongoing seabird census.

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Appendix 1

Urban Gulls: Survey methods

Urban nesting gulls have expanded their distribution over the last 15-20 years. To be able to produce population estimates and distribution maps, a new method and sample regime has been created. The method consists of surveying a sample of 1 km squares in urban areas, across the UK and Ireland. The sample is a stratified random sample based on the number urban squares, and the ratio of strata[1], found within each county. With such sampling, it is likely that some squares will not contain gulls, however, for the data to be statistically robust and not biased, these squares will still need to be surveyed **and reported on**.

The methods for the survey are **ground-based** counts of herring and Lesser Black-backed Gulls, vantage points are not necessary for this survey. If a clear sight of several roofs can be gained at the top of street or at a junction etc, there is no need to walk the entirety of the street. You only need to survey the urban areas of the square, surrounding fields etc, do not need to be included in the count. Not all birds will be able to be viewed from the ground, please do not worry if you think you have missed some, just record what you can see. The analysis of these results adjusts for the lower detection rate that occurs in urban environments.

Counts for each species will need to be reported on separately, for each 1 km square. A count for **all three** units below will need to be recorded for each species:

- **AON** = apparently occupied nest (well-constructed nest or scrape nest, either containing eggs or young, or capable of holding eggs (possibly attended by an adult) or an apparently incubating adult),

- **AOT** = apparently occupied territory (estimated by the spacing of birds or pairs on different rooftops and observations of apparent territorial behaviour, when actual nests cannot be discerned. Any AONs should also be considered a territory, so the number of AOTs will always be equal to or greater than the number of AONs.)

- **IND** = Individual adults (Count the total number of birds in full adult plumage. Individual birds should only be counted once. However, where movement occurs it will sometimes be impossible to be certain whether some birds have already been counted in which case you should use your best judgement to decide. Birds in flight can be counted if it is clear they are using rooftops in the square, but birds observed flying over the square should not be counted)

Please note these counts are cumulative i.e. the count of AOT's will be the number of AON's observed plus any extra AOT's seen, the number of individuals will be all birds seen, including those on the nests and territories. An example recording sheet is attached at the end of this document to highlight this. Proposed survey timings can also be found at the end of this document.

Also needing to be recorded:

- The survey square, date of survey, time of survey and weather. These should be noted on the count sheets in the appropriate box.
- A note of any use of gull deterrents or control measures e.g. netting on roofs will be much appreciated.
- A rough estimate of the % of the square that **could not** be accessed for survey (i.e. private land)
- Finally, use the comment section for any other observations made.

Please use this link to submit your data: http://bit.ly/JNCCgullsurvey

Survey timings

*These timings are only a guide, local knowledge and own judgement based on when the gulls are at peak incubation should be used.

Survey Timings	
England	Late April – Late May (Southern & inland surveys closer to the earlier period progressively getting later further North & towards the coast)
Wales	Late April – Late May (Inland colonies closer to the earlier period, coastal can be towards the latter half)
Northern Ireland	May (coastal colonies may breed closer to the latter half of the period)
Scotland	May – 1st week June (coastal colonies may breed closer to the latter half of the period)

Seabirds Count – Urban Gull Survey Form

Recorder name:	Daisy Burnell	County/ district of survey:	Cleveland
Recorder contact details: Phone:	01224 266577	email:	Daisy.burnell@jncc.gov.uk

Square grid ref	Species Code	Date (DD/MM/YY)	Start Time (24hr)	End Time (24hr)	Counts			% of square that could not be accessed	Weather			Gull deterrent present	description	Additional comments
					AON	AOT	IND		Vis.	Wind	Rain			
NZ5233	HERGU	20/05/19	10:00	11:30	10	13	16	5	1	1	2	No		90% of this square is the sea. The % not accessed is based on the urban area that actually contains urban fabric.
NZ5233	LBBGU	20/05/19	10:00	11:30	4	7	9	5	1	1	2	No		As above
NZ5133	HERGU	19/05/19	12:30	13:30	4	9	8	10	1	0	1	Yes	Netting on some rooftops	Some birds resting on water, but not included in count.
NZ5133	LBBGU	19/05/19	12:30	13:30	0	0	3	10	1	0	1	Yes	Netting on some rooftops	As above

The number of AOTs will always be either equal or more than the count of AONs. In this example 10 AONs could be discerned and 3 AOTs could be seen therefore the total AOT is 10+3=13

The number of IND is the number of all adult birds present on suitable nesting habitat. This includes those sitting on nests or associated with territories. It will almost always be equal or more than the AOT count. However, a nest counted as an AON/AOT with no adult present will mean a lower count of IND.

Zero counts are very important, do not forget to report these. If there are adults around on suitable urban nesting habitat don't forget to count them, even if there are no visible nests.

Appendix 2

Table 17. Model outputs for the favoured NB GLM equation for Herring Gull.

GBS IND ~ log(Dist’ to gull colony +1)*Inland.Coastal + log(Dist’ from coast+1)*Inland.Coastal + log(Dist’ to confirmed breeding square)*Inland.Coastal + Region*PROP Strata +log(Dist’ to freshwater+1)*Inland.Coastal.

p-values in bold are significant <0.05.

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	4.267904	0.51788	8.241	**<0.001**
log(Dist.from.coast + 1)	-0.09203	0.030506	-3.017	0.002555
Inland	10.05014	1.036637	9.695	**<0.001**
East Midlands	-0.05842	0.41156	-0.142	0.887124
London	2.308969	0.569368	4.055	**<0.001**
North East	2.081621	0.392937	5.298	**<0.001**
North West	0.478867	0.343203	1.395	0.162929
South East	0.486217	0.288774	1.684	0.092235
South West	1.233923	0.281682	4.381	**<0.001**
West Midlands	-0.5916	0.575639	-1.028	0.30408
Yorks and Humber	-0.46669	0.405888	-1.15	0.250222
PROP	0.028748	0.003998	7.191	**<0.001**
log(Dist.to.confirmed.breeding.sqr)	-0.15576	0.03089	-5.042	**<0.001**
log(Dist.to.gull.colony + 1)	-0.2511	0.049581	-5.064	**<0.001**
log(Dist.to.freshwater + 1)	0.013898	0.033158	0.419	0.675116
Ind/Sub mix	0.05732	0.267912	0.214	0.830587
Most Urb	-0.14603	0.25828	-0.565	0.571802
Sub	-0.63837	0.181664	-3.514	**<0.001**
Interaction: log(Dist.from.coast + 1)	-0.82865	0.094086	-8.807	**<0.001**
Interaction: East Midlands	-0.01893	0.007491	-2.526	**0.011523**
Interaction: London	-0.03128	0.007643	-4.093	**<0.001**

Interaction: North East	-0.02619	0.007115	-3.68	**<0.001**
Interaction: North West	-0.01206	0.005805	-2.077	**0.037771**
Interaction: South East	-0.00024	0.005292	-0.046	0.963499
Interaction: South West	-0.01505	0.005458	-2.757	**0.005825**
Interaction: West Midlands	-0.0096	0.008101	-1.186	0.235772
Interaction :Yorks and Humber	-0.00526	0.006983	-0.753	0.451313
Interaction: log(Dist.to.confirmed.breeding.sqr)	-0.28287	0.0425	-6.656	**<0.001**
Interaction: log(Dist.to.gull.colony + 1)	-0.01826	0.068398	-0.267	0.789498
Interaction: log(Dist.to.freshwater + 1)	-0.07713	0.038367	-2.01	**0.044405**
AIC	7552.5			
RMSE	27.14			
DHARMa dispersion (p-value)	0.4266 (**<0.001**)			
DHARMa uniformity (p-value)	0.031687 (**<0.001**)			
DHARMa zero-inflation (p-value)	1.0136 (0.072)			

Table 18. Model outputs for the favoured NB GLM equation for Lesser Black-backed Gull.

GBS IND ~ log(Dist' to gull colony +1)*Inland.Coastal + log(Dist' from coast+1)*Inland.Coastal + log(Dist' to confirmed breeding square)*Inland.Coastal + Region*PROP Strata +log(Dist' to freshwater+1)*Inland.Coastal.

p-values in bold are significant <0.05.

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	2.91967	0.639518	4.565	**<0.001**
log(Dist.to.gull.colony + 1)	-0.15037	0.062875	-2.392	**0.016777**
Inland.CoastalInland	4.013452	1.287066	3.118	**0.001819**
log(Dist.from.coast + 1)	-0.05861	0.039664	-1.478	0.139483
log(Dist.to.confirmed.breeding.sqr)	-0.14956	0.039192	-3.816	**<0.001**
log(Dist.to.freshwater + 1)	-0.07946	0.042419	-1.873	0.061045
PROP	0.037365	0.004429	8.436	**<0.001**
East Midlands	-1.68645	0.541318	-3.115	**0.001837**
London	0.353097	0.745179	0.474	0.635612
North East	1.069127	0.487116	2.195	**0.028177**
North West	0.40407	0.385906	1.047	0.295068
South East	-1.37966	0.379716	-3.633	**<0.001**
South West	-0.58072	0.346477	-1.676	0.093723
West Midlands	0.406621	0.401077	1.014	0.310666
Yorks and Humber	-1.28035	0.480014	-2.667	**0.007646**
Ind/Sub mix	0.438367	0.308048	1.423	0.154723
Most Urb	0.032124	0.301183	0.107	0.91506
Sub	-0.63343	0.213713	-2.964	**0.003037**
Interaction: log(Dist.to.gull.colony + 1)	0.05748	0.085205	0.675	0.499922
Interaction: log(Dist.from.coast + 1)	-0.33436	0.11709	-2.856	**0.004296**
Interaction: log(Dist.to.confirmed.breeding.sqr)	-0.21972	0.052754	-4.165	**<0.001**
Interaction: log(Dist.to.freshwater + 1)	0.014969	0.048036	0.312	0.755336

Interaction: East Midlands	-0.01608	0.008797	-1.828	0.067563
Interaction: London	-0.02586	0.009711	-2.664	**0.007732**
Interaction: North East	-0.04665	0.008997	-5.185	**<0.001**
Interaction: North West	-0.0157	0.006589	-2.383	0.017184
Interaction South East	-0.00516	0.006649	-0.776	0.437993
Interaction: South West	-0.00489	0.006619	-0.739	0.460001
Interaction: West Midlands	-0.01783	0.006666	-2.675	**0.00748**
Interaction: Yorks and Humber	-0.00611	0.008004	-0.763	0.445194
AIC	5432.4			
RMSE	10.8			
DHARMa dispersion (p-value)	0.53254 (0.12)			
DHARMa uniformity (p-value)	0.017902 (0.167)			
DHARMa zero-inflation (p-value)	1.0051 (0.44)			

Appendix 3

Table 19. Model outputs for the favoured ZINB model for Herring Gull.

Neg-bin model: Individuals ~ log(Dist' from coast+1) * Inland.Coastal + Region*PROP +
log(Dist' to nearest confirmed breeding square)*Inland.Coastal +
log(Dist' to nearest gull colony +1)*Inland.Coastal +
log(Distance to nearest freshwater source +1)*Inland.Coastal + Strata

Zero-inflation model: log(Distance from coast+1) + PROP + log(Distance to nearest gull colony +1) +
log(Distance to nearest confirmed breeding square)

The p-value that are significant are in bold (<0.05)

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	4.167176	0.407382	10.229	**<0.001**
log(Dist.from.coast + 1)	-0.0253	0.022016	-1.149	0.250585
Inland	5.592797	1.088238	5.139	**<0.001**
East Midlands	0.595603	0.496426	1.2	0.230224
London	2.688419	0.664026	4.049	**<0.001**
North East	1.493906	0.396275	3.77	**<0.001**
North West	0.36519	0.359746	1.015	0.310042
South East	0.65729	0.334647	1.964	**0.049515**
South West	0.786762	0.294822	2.669	**0.007617**
West Midlands	0.091813	0.652464	0.141	0.888093
Yorks and Humber	-0.08496	0.507159	-0.168	0.866965
PROP	0.016794	0.003804	4.415	**<0.001**
log(Dist.to.confirmed.breeding.sqr)	-0.12553	0.021461	-5.849	**<0.001**
log(Dist.to.gull.colony + 1)	-0.16819	0.037976	-4.429	**<0.001**
log(Dist.to.freshwater + 1)	0.001819	0.025019	0.073	0.942035
Ind/Sub mix	-0.09868	0.229804	-0.429	0.667623
Most Urb	-0.04161	0.218363	-0.191	0.848887
Sub	-0.61849	0.158091	-3.912	**<0.001**

Interaction: log(Dist.from.coast + 1)	-0.70305	0.108662	-6.47	**<0.001**
Interaction: East Midlands	-0.02017	0.007638	-2.641	**0.008268**
Interaction: London	-0.03568	0.008312	-4.292	**<0.001**
Interaction: North East	-0.02414	0.006129	-3.938	**<0.001**
Interaction: North West	-0.00725	0.005349	-1.356	0.175116
Interaction: South East	-0.00383	0.005404	-0.709	0.478451
Interaction: South West	-0.01045	0.004769	-2.192	**0.0284**
Interaction: West Midlands	-0.01801	0.008424	-2.138	**0.032496**
Interaction :Yorks and Humber	-0.00965	0.009056	-1.065	0.286723
Interaction: log(Dist.to.confirmed.breeding.sqr)	-0.0551	0.031646	-1.741	0.081681
Interaction: log(Dist.to.gull.colony + 1)	0.134034	0.052104	2.572	**0.010098**
Interaction: log(Dist.to.freshwater + 1)	-0.01136	0.032582	-0.349	0.727296
Log(theta)	-0.51786	0.076475	-6.772	**<0.001**
Zero-inflation model coefficients (binomial with logit link):				
(Intercept)	-9.78983	0.80801	-12.116	**<0.001**
log(Dist.from.coast + 1)	0.23063	0.04232	5.45	**<0.001**
PROP	-0.01805	0.0024	-7.52	**<0.001**
log(Dist.to.gull.colony + 1)	0.50219	0.07531	6.669	**<0.001**
log(Dist.to.confirmed.breeding.sqr)	0.54212	0.05067	10.699	**<0.001**
AIC	7166.1			
RMSE	10.8			

Table 20. Model outputs for the favoured ZINB model for Lesser Black-backed Gull.

Neg-bin model: Individuals ~ log(Dist' from coast+1) * Inland.Coastal **+** Region*PROP **+**
log(Dist' to nearest confirmed breeding square)*Inland.Coastal **+**
log(Dist' to nearest gull colony +1)*Inland.Coastal **+**
log(Distance to nearest freshwater source +1)*Inland.Coastal **+** Strata

Zero-inflation model: log(Distance from coast+1) **+** Proportion of square covered in urban fabric **+**
log(Distance to nearest gull colony +1) **+**
log(Distance to nearest confirmed breeding square) **+**
log(Distance to nearest freshwater source +1)

The p-value that are significant are in bold (<0.05)

	Estimate	Std. Error	z value	Pr(>|z|)
(Intercept)	3.058109	0.658819	4.642	**<0.001**
log(Dist.to.gull.colony + 1)	-0.07232	0.062425	-1.158	0.246677
Inland.CoastalInland	1.874167	1.451721	1.291	0.196705
log(Dist.from.coast + 1)	0.013485	0.035802	0.377	0.706417
log(Dist.to.confirmed.breeding.sqr)	-0.11161	0.033202	-3.362	**<0.001**
log(Dist.to.freshwater + 1)	-0.06234	0.0394	-1.582	0.113579
PROP	-1.53526	0.634085	-2.421	**0.015468**
East Midlands	0.09894	0.852494	0.116	0.907605
London	0.683894	0.576205	1.187	0.23527
North East	0.281229	0.511536	0.55	0.582475
North West	-1.40676	0.483551	-2.909	**0.003623**
South East	-1.07521	0.417259	-2.577	**0.009971**
South West	0.134328	0.585408	0.229	0.81851
West Midlands	-1.24814	0.653706	-1.909	0.056219
Yorks and Humber	0.021817	0.005294	4.121	**<0.001**
Ind/Sub mix	0.261947	0.296872	0.882	0.377584
Most Urb	0.104958	0.282005	0.372	0.709757
Sub	-0.79301	0.224357	-3.535	**<0.001**
Interaction: log(Dist.to.gull.colony + 1)	0.031694	0.082626	0.384	0.701286

Interaction: log(Dist.from.coast + 1)	-0.18959	0.135631	-1.398	0.162165
Interaction: log(Dist.to.confirmed.breeding.sqr)	-0.0947	0.044238	-2.141	**0.0323**
Interaction: log(Dist.to.freshwater + 1)	0.049915	0.045856	1.089	0.276369
Interaction: East Midlands	-0.01628	0.008917	-1.826	0.067846
Interaction: London	-0.0268	0.010659	-2.515	**0.011913**
Interaction: North East	-0.04065	0.008791	-4.624	**<0.001**
Interaction: North West	-0.0154	0.007147	-2.154	**0.03124**
Interaction South East	-0.01092	0.007638	-1.43	0.152855
Interaction: South West	-0.00294	0.006827	-0.431	0.666729
Interaction: West Midlands	-0.01906	0.007831	-2.434	**0.014925**
Interaction: Yorks and Humber	-0.01176	0.009925	-1.185	0.235939
Log(theta)	-1.24185	0.102734	-12.09	**<0.001**
Zero-inflation model coefficients (binomial with logit link):				
(Intercept)	-6.87754	1.208039	-5.693	**<0.001**
log(Dist.to.gull.colony + 1)	0.331986	0.097812	3.394	**<0.001**
PROP	-0.02968	0.003645	-8.142	**<0.001**
log(Dist.from.coast + 1)	0.185807	0.053491	3.474	**<0.001**
log(Dist.to.confirmed.breeding.sqr)	0.408748	0.07501	5.449	**<0.001**
log(Dist.to.freshwater + 1)	0.09049	0.033091	2.735	**0.006246**
AIC	5255.8			
RMSE	9.45			

Appendix 4

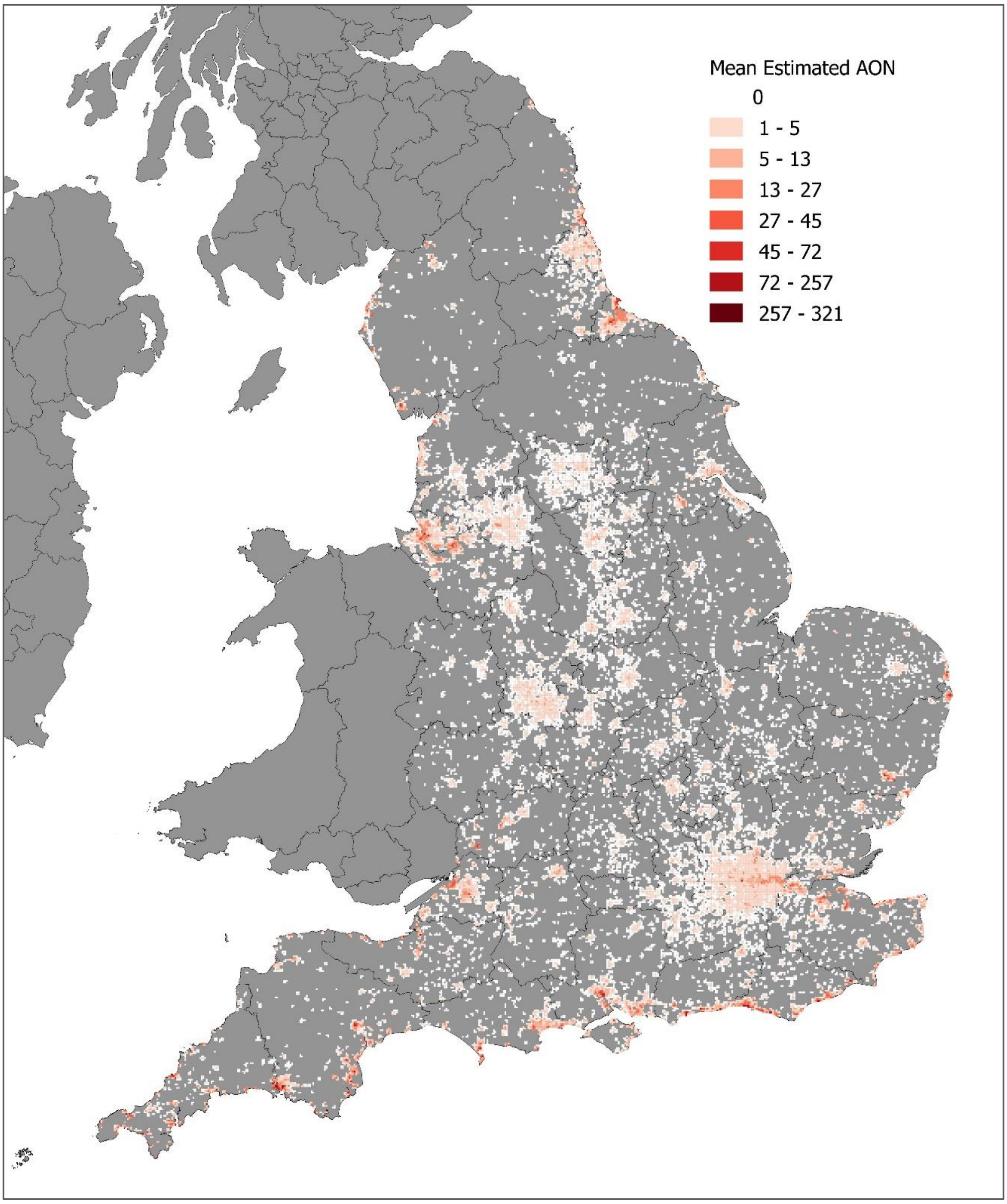

Figure 4. Mapped mean estimated **Herring Gull** AONs per 1 km urban square in England, based on ZINB model predicted mean IND Herring Gulls in each 1 km square. Note these should be interpreted with caution as they do not show the error around these estimates.

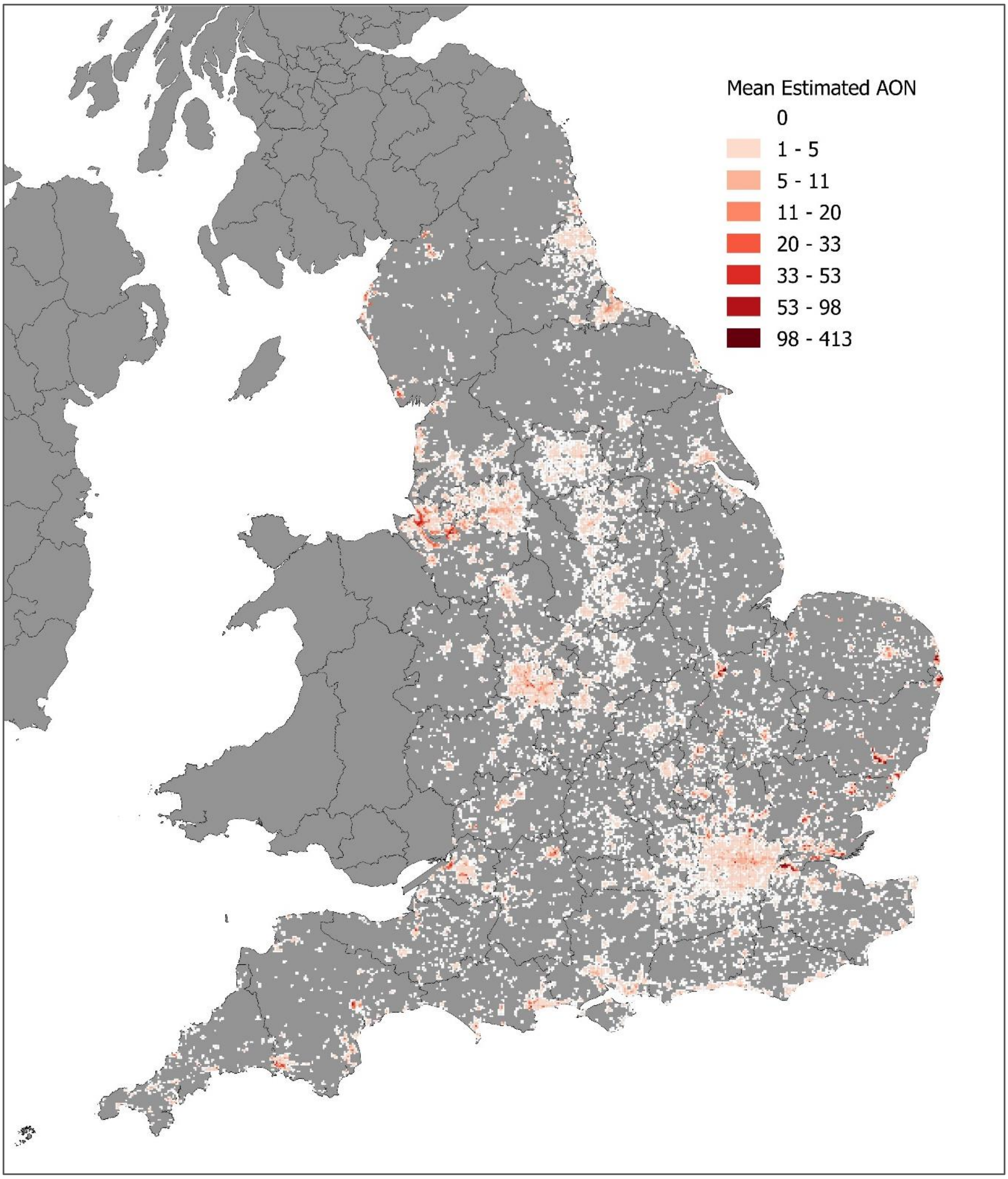

Figure 5. Mapped mean estimated **Lesser Black-backed Gull** AONs per 1 km urban square in England, based on ZINB model predicted mean IND Lesser Black-backed Gulls in each 1 km square. Note these should be interpreted with caution as they do not show the error around these estimates.