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The JNCC Terrestrial Biodiversity Surveillance Schemes: An Assessment of Coverage

Border, J., Gillings, S., Newson, S.E., Logie, M., August, T.,
Robinson, R.A. & Pocock, M.J.O.

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For further information please contact:

Joint Nature Conservation Committee
Monkstone House
City Road
Peterborough PE1 1JY
www.jncc.gov.uk

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Summary

Biodiversity information is needed to provide a sound evidence base for decision-making, including operational needs, statutory reporting requirements and strategic needs. In this study we sought to assess aspects of coverage so as to identify gaps in taxonomic, thematic (habitat) and spatial coverage which might need to be addressed in future.

The UK Surveillance Schemes (many, though not all, of which are supported by JNCC) can be broadly divided into two categories:

(i) those that employ some form of structured sampling; and
(ii) those that support the collection of unstructured observational records.

We assess coverage provided by these two contrasting scheme types separately, since they differ substantially in their nature and scope. Note, this report only considers schemes that are supported by JNCC, including structured monitoring and unstructured recording (supported through the Biological Records Centre).

With a focus on terrestrial biodiversity, we assessed coverage by taxon (how well different groups are monitored, with an emphasis on the quality of data for assessing stock and change in distribution, abundance and demographic rates), habitat (to what extent can inferences be made for specific broad habitat types), and spatial coverage at multiple spatial scales: national, regional and local.

Taxonomic coverage
While there is variation within and between taxonomic groups in our ability to record and monitor populations, species groups can broadly be divided into three on the basis of current coverage. The first group, for which biological recording or monitoring is most well developed, includes birds, flowering plants and ferns, butterflies and macro moths, fish and bats. Structured surveillance schemes are in place for many of these. The second group includes other mammals and amphibians and reptiles; species which are generally popular with recorders and so large numbers of records are submitted. The third group includes stoneworts, mosses and liverworts, lichens, invertebrates excluding butterflies and macro moths, fungi and slime moulds; most of which are challenging to record for a variety of reasons.

National-scale structured sampling data are available for four groups of biodiversity: bats, birds, butterflies and plants. Coverage of species by monitoring schemes for these groups was assessed in conjunction with hectad atlas data which provide a more exhaustive inventory of species in a particular location. The unstructured datasets came from the National Recording Schemes and Societies supported by the Biological Records Centre at CEH, of which we used data from 32 of these to estimate trends in occupancy.

Data required to produce trends
Rules of thumb are used as an indication of when there are sufficient data for a species to warrant production of a trend or other metric. The existence and derivation of rules of thumb varies between structured and unstructured data. For structured data we adopted the often-used rule of thumb that for analysis a species should have an average of at least 30 counts (or occupied sites) per year. For the unstructured data, there was no equivalent rule of thumb, so we developed these based on whether trends were deemed to meet a threshold of precision (determined by expert evaluation). Importantly, we note that meeting the precision threshold is not evidence that the resulting trend is accurate or representative of the species.

From our production of rules of thumb for unstructured data, we derive some key principles:

(i) sites should be visited on more than one occasion and on more than one year for data to be useful for analysis of trends;
(ii) surprisingly little data is required for use with our Bayesian occupancy models with weakly informative priors to produce estimates of yearly occupancy that met the precision threshold;
(iii) the better-recorded years are most influential on the precision of the outputs, compared to years of average effort; and
(iv) the minimum level of precision consistently increased with the number of records of the focal species, so increased recording of biodiversity will support better analysis to support decision-making and management decisions. A rule of thumb for the minimum requirements is that the best 10% of years should have >29 presence records for the focal species (plus records for other species in the taxonomic group from which to infer non-detections) to be confident of producing a trend that meets the precision threshold, although the required number of presence records is lower for rarer species.

Spatial coverage

We considered spatial coverage in relation to the Nomenclature of Territorial Units for Statistics 1 (NUTS1) boundaries as they are widely used in policy, well established and large enough to potentially contain sufficient records for generating species trends, yet small enough to allow regional variations to be detected. The 2015 Land Cover Map was used to determine habitat coverage of surveys due to its wide use and policy relevance, with habitats delineated into nine broad classes.

Considering the structured surveys; those of birds have the greatest spatial coverage in terms of number of squares sampled, followed by butterflies, then bats, then lastly plants; however, the National Plant Monitoring Scheme has been running for the shortest span of time and was designed to assess habitat quality, rather than a comprehensive species survey as the other schemes. Birds are the only taxon for which it was possible to calculate trends for at least some species across all regions from structured survey data; butterfly trends were possible for at least some species in all regions apart from Northern Ireland; no trends could be generated for bat and plant species in four of the regions. Coverage by the structured surveys was best in the two southern regions of England, which have good recorder bases, and poorest in the largely urban areas of London and the West Midlands of England, but also in the more rural regions of Northern Ireland and North-east of England.

There was substantial variation across taxa and across regions in the proportion of species predicted to be modelled with high or acceptable precision using the unstructured data. This variation was not directly related to human population density, although the South-east of England tended to be best recorded and both Wales and Scotland are relatively poorly recorded. A few taxa are well-recorded (e.g. butterflies and moths) by the unstructured schemes and many species in those taxa are predicted to have high precision trends across many regions. Some taxa (e.g. pollinator groups, which have been the focus of recent attention) are reasonably-well recorded with an ability to produce outputs for some species in many regions. For many taxonomic groups, however, most species are predicted to produce trends with that did not meet the precision threshold at the regional level.

Habitat coverage

In general, the coverage of habitats by the structured schemes matches the UK habitat distribution fairly well with the exception of built-up areas and broadleaved woodland, which are over-sampled for all four taxa, and mountain/heath/bog which is under-sampled for all four taxa. Across all taxa, broadleaved woodland has the best coverage, closely followed by arable and improved grassland; mountain/heath/bog has by far the worst coverage. For birds and butterflies it was possible to generate trends for some species for all habitats, but for bats and plants, trends could not be generated for any species for four of the habitats with lower coverage.

We applied the rules of thumb for the unstructured data to predict which species would be modelled with acceptable or high precision within each habitat. In general, there are four habitats that appear to have relatively good species coverage: broad-leaved woodland, built-up areas, intensive grassland and arable.

Fine-scale spatial coverage
We assessed fine-scale spatial coverage of the structured schemes by:

(i) quantifying the proportion of species in a hectad for which published national trends were available, and
(ii) estimating the spatial precision with which individual species trends could be produced using the rule of 30 counts to produce a trend.

Published trends are produced for a high proportion of established butterfly species (mean 99.8%) and this does not show much spatial variation. For birds, a high percentage (mean 90.5%) of established species have published trends, but there was more spatial variation, with the Highlands of Scotland and the Outer Hebrides having the lowest percentages of established bird species reported in national trends. A smaller percentage of surveyed bat species are included in published trends (mean 72.5%).

Birds show the highest inter-species variation in the area needed to encompass 30 counts, then plants and butterflies with bats showing the least variation between species. Although there is some spatial variation, in general, an area covering around 200km from a site is far enough to get 30 samples for the majority of species across birds, bats and butterflies; for plants, however, this value is closer to 300-400km. These figures give a broad indication of the likely spatial resolution with which multi-taxa indicators might be constructed.

Implications for conservation monitoring
Taxonomically, we have a particularly poor understanding of the population status of lower plants and many invertebrates. It is important, though, to identify and prioritise groups that would yield most benefit from improving coverage. For example, species of conservation and international importance (e.g. lichens), or those that form an important functional part of most ecosystems (e.g. Coleoptera, Symphyta and Chironomidae). An alternative approach would be to identify species that are sensitive to particular environmental pressures (e.g. nocturnal invertebrates or bryophytes).

Spatially there is a general trend for lower coverage towards the north and the west of Britain, with highest coverage for most taxa for most taxa in the south-east of England, however, perhaps surprisingly, the greater London region is relatively under-represented. More in line with expectations, moor/heath/bog habitats were also identified as poorly covered, despite the fact that these include some that are scarce at a European level and for which Britain has particular responsibility.

There is a need for suitable monitoring data to assess the impact of conservation and policy interventions. We used taxonomic, spatial and habitat coverage of the surveillance schemes (gathering structured and unstructured data) as a proxy for ability to address questions of concern. However, this report highlights there may be limits to which this can be achieved through volunteer-based monitoring, and we have identified areas where there is relatively poor coverage, both spatially (by region and habitat) and taxonomically which might be valuable areas of focus in future prioritisation. However, it is essential to focus on the questions of interest in further consideration of coverage, because many questions (e.g.

about the success of policy interventions) may require different patterns or levels of data coverage. Ensuring good coverage of biodiversity data across taxa and areas does though act as insurance, making it more likely that a range future questions could be addressed with these data.

In understanding biodiversity responses to environmental change (anthropogenic or natural), there is a hierarchy of information:

(i) where do things occur (addressed using either structured or unstructured data);
(ii) how many are there (typically requires structured data); and
(iii) why are things changing? Gaps in data to address these may be caused by either incomplete or biased coverage, or simply by an inability to answer questions of interest. Arguably the latter is more relevant, but it is more challenging to generically identify such ‘gaps’ since it depends both on the particular question being asked and its context (e.g. in space or time).

There is a range of solutions that could be used to address the gaps in data that we identified in this report. Not all solutions will be applicable in all cases, and their ease of implementation will vary between groups. Most current monitoring is based around volunteer contributions. Volunteers require support, both to coordinate the effort and to effectively capture the data in a form that is amenable to analysis. Maintaining a well-motivated volunteer base is critical to the success of biodiversity recording. This can be facilitated in a number of ways, including:

(i) provision of mentoring or training materials;
(ii) promotion of community networks;
(iii) improved availability of identification resources; and
(iv) the use of technology to support or automate the identification process.

The extent to which technology can aid biodiversity and environmental monitoring is increasing rapidly and while most biodiversity recording has been based on volunteers making records through their own identifications or sampling, technological developments offer the potential to fill some gaps, either taxonomic or spatial. These include sampling environmental DNA, Earth observation techniques, and deployment of passive recorders.

There may be opportunities to establish levels of co-located cross-taxa recording, involving either the same individuals recording different taxa, or by different recorders participating in different schemes recording at the same sites.

In general, for a given number of sampling sites, structured recording (i.e. recording that is defined, consistent and repeated), generally gives the greatest statistical power to determine temporal or spatial trends, but relies on sufficient volunteers with relatively high levels of commitment and able to follow set protocols at set sites at specific times, which will be possible for only a few taxonomic groups. While unstructured recording can have lower power to detect changes, this may be offset by a greater number of samples with wider coverage. Given the need to make maximum use of the data available, we particularly recommend further investment in developing joint analyses of different biodiversity data (e.g. combining structured, semi-structured and/or unstructured data) and further investigation in combining biodiversity data with other environmental data. This could assist spatial extrapolation to unsampled areas and potentially help assess the impact of environmental drivers and management interventions.

Contents

1 Introduction

Biodiversity information is needed to provide a sound evidence base for decision-making, including operational needs, statutory reporting requirements and strategic needs. The JNCC terrestrial Surveillance Schemes run in partnership with NGOs and research organisations are very important in providing this information but use of data and information from the schemes critically depends on aspects of dataset quality such as taxonomic and spatial coverage and the precision with which inferences can be made. In this study we sought to assess aspects of coverage so as to identify gaps in taxonomic, thematic (habitat) and spatial coverage which might need to be addressed in future. The work builds on an assessment of the needs for biodiversity evidence in the country nature conservation bodies (CNCBs) and related public sector bodies (Pocock 2018, hereafter referred to as the needs assessment). In this work, we undertake a coverage mapping or gap analysis of the schemes to assess how well they meet the identified needs.

The UK Surveillance Schemes (many, though not all, of which are supported by JNCC) can be broadly divided into two categories:

(i) those that employ some form of structured sampling (e.g. selected sites, possibly selected at random, which are revisited on a set timescale using a specific protocol), and
(ii) those that result in the collection of unstructured observational records, which are likely to be more heterogeneous, but may come from a broader range of locations.

Examples of the former are the UK Butterfly Monitoring Scheme and of the latter are the more than 80 schemes contributing to data held by the Biological Records Centre, which provides a focus for the collation, management, dissemination and interpretation of these observations. The data provided by these two contrasting scheme types and the analyses required to extract trends are so different that it makes sense to assess coverage separately. For the purposes of this report, BTO led on the assessment of coverage for structured schemes and CEH led on the analyses of unstructured data. These are described in more detail in Section 3 and the results presented in Sections 4 and 5. To allow comparison in coverage between the schemes, where possible the same metrics are calculated for both types of surveys and displayed in concurrent sections.

Using a questionnaire-based approach, Pocock (2018) determined the UK's public environmental bodies use biodiversity information for a wide range of purposes, from strategic reporting to more immediate operational needs (e.g. planning advice and regulation). While the specific environmental policies may vary across the four countries of the UK, the biodiversity information required to meet and assess these policies is broadly similar. The assessment highlighted a need to increase both spatial and taxonomic coverage. Increased taxonomic coverage is regarded as helping to provide a broader perspective of ecosystem health, which is an increasingly important metric; whereas increased spatial coverage for already-well-monitored taxa is regarded as more feasible to achieve. In particular, there is a need for suitable monitoring data to assess the impact of conservation and policy interventions. More generally, there is a need to increase both spatial and taxonomic coverage, accepting that there can be limitations on what is achievable. In this work, we set out to identify where the key gaps in coverage are and provide some suggested approaches for how they may be overcome.

1.1 Objectives

The main objective of this work was to review coverage of biodiversity sampling. This was considered in terms of taxonomic coverage, habitat and geographically, and covered both structured and unstructured sampling. In particular we produced the following:

a) An initial qualitative assessment of how well different taxonomic groups are monitored, with emphasis on the quality of data for assessing stock and change in distribution, abundance and demographic rates;
b) For existing schemes, an assessment of the extent to which inference can be made for specific broad habitat types;
c) For existing schemes, an assessment of spatial coverage of the monitoring data at multiple spatial scales: national, regional and local.

A further key objective of this work was to develop some ‘rules of thumb’ to help us judge when there was sufficient unstructured recording available to be able to produce trends with a useful level of precision. These rules of thumb were used in the work to assess coverage, but have wider utility and we have used them to draw out recommendations for recording.

The final objective was to consider the implications of this work for conservation organisations and surveillance scheme organisers. Recommendations have been highlighted throughout the different sections of the report, and the final section focusses particularly on the ways some of the gaps we have identified might be filled. The discussion considers other work within the ‘Terrestrial Surveillance Development and Analysis’ project, including the Needs Assessment (Pocock 2018), as well as drawing on outcomes from workshops held in 2017 and 2018 by the ‘Terrestrial Evidence Partnership of Partnerships’ (TEPoP), a group comprising representatives from the JNCC supported surveillance schemes and policy representation from country conservation bodies and governments.

1.2 Scope

This report only considers schemes that are supported by JNCC, including structured monitoring and unstructured recording (supported through the Biological Records Centre, which is jointly funded by CEH and JNCC).

It should be noted that other monitoring and recording in the UK is undertaken by other public bodies and NGOs (e.g. Common Standards Monitoring of SSSIs, Countryside Survey, water quality monitoring, woodland monitoring, and collation of records by Local Environmental Records Centres (LERCs), but due to resource and data access constraints we have not addressed these here unless their data is added to the JNCC-supported schemes. In Wales in particular, the LERCs collate a significant amount of unstructured data that is not in the BRC database.

In this report we firstly assess the coverage of taxonomic groups (Section 2). Then we briefly summarise the structured and unstructured datasets[1] that are available for monitoring purposes (within the TEPoP; Section 3) before considering ways of assessing coverage (Section 4). Next, we consider the regional coverage and the habitat coverage of both the structured and unstructured data (Section 5). After this we will examine fine-scale spatial coverage for the structured data (Section 6); this level of detailed analysis is not feasible to

[1] Structured data are those that stem from a scheme with a designed sampling protocol, such as randomly selected survey squares and/or the use of rigid field survey techniques (e.g. strip transects). Unstructured data are collections of distribution records collated from field recording including ad hoc casual records, expeditions and other field recording that does not rely on fixed locations or rigidly standardised sampling procedures.

undertake for the unstructured data. Finally (Section 7), we outline ways in which some of the gaps that we have identified might be filled.

2 Taxonomic coverage

In this section, we consider current taxonomic sampling coverage of terrestrial species in the UK, and the extent to which we are able provide a precise (i.e. with a small degree of uncertainty) and accurate (i.e. unbiased) assessment of the status and change in status of different species groups. We do this as an initial qualitative assessment of how well different taxonomic groups are recorded or monitored, with emphasis on the quality and extent of data that is available for assessing stock and change in distribution, abundance and demographic rates.

2.1 Methods

We consider this question in relation to six species **population measures** that are captured by one or more of the Surveillance Schemes and which are likely to be of interest to stakeholders. These were selected based on our judgement of the parameters that have strong biological relevance. Although some of these measures may be difficult to collect for certain taxa, they are all relevant in conferring something about the conservation status of species and their ability to provide ecosystem services:

1. Population size/abundance
2. Trends in population size/abundance
3. Distribution
4. Trends in distribution
5. Demographic rates
6. Trends in demographic rates

Many of these have direct policy relevance, either for international reporting (population size), for indicator metrics (trends in population/abundance) or for contributing to assessments of ecosystem health (demographic rates). We focus here on species-specific measures, which form the basis of much biodiversity recording, monitoring of cross-species measures, such as community composition, taxonomic diversity and the strength and nature of species interactions is, in many cases, currently less well developed.

Within each of the six population measures above, we define **subcategories** that relate to the extent and quality of existing biological recording or monitoring data that are available for each species group, and so our ability to provide a precise and accurate assessment of that population measure for each species group (Table 2.1). For each subcategory, we associate a single **quality score** of 1…*n*, where 1 is the lowest extent and quality of data, to *n* the highest extent and quality of data. For example, for the measure Population size, the lowest quality subcategory was Ad-hoc counts (assigned a quality score of 1) and the highest quality subcategory was Complete census (assigned a quality score of 6) (Table 2.1).

For each species group and population measure, we also distribute a **taxonomic coverage score out** of 10, based on an expert qualitative assessment by BTO and CEH on the proportion of species in each species group, which fall into each subcategory. This was initially done by giving one or two experts responsibility for each species group, and then moderated independently by the authors to ensure that there was consistency in scoring across species groups.

Where it was believed there were likely to be clear differences in scoring within a species group, for example nocturnal birds are poorly recorded compared to widespread and abundant breeding bird species, we also split species groups into sub-groups to reflect this. In some cases, scores are assessed separately for sub-groups of species with particular policy or reporting needs (e.g. migratory waterbirds) and for mammals we grouped species for which monitoring scores were similar, rather than taxonomically, to provide a clear picture of the different levels of monitoring being undertaken. This inevitably leads to us highlighting "known unknowns" but where we know that there are differences in our ability to monitor particular species or groups of species within taxonomic groups, is important to capture this, as it provides the most accurate information that we can for a taxonomic group. Also, this is most often the case for high level groups that are generally well covered (e.g. birds). For groups that are generally poorly covered (e.g. insects) it has to be assumed that subcategories within such groups are also likely to be poorly covered unless specified otherwise. We are not expecting to be exhaustive in doing this, and only do this where the scores across species within a group do not vary widely, but we hope that this will help the reader interpret situations where values across a species group are assigned to several sub-categories and to identify monitoring gaps of key importance.

A single aggregate score for each population measure was then calculated as the product of the **quality score** and the **taxonomic coverage score** and expressed as a proportion of the maximum score (to account for the fact that the maximum possible scores in some categories differed). In doing this we were able to provide a single score for each species group (and species subgroup), and population measure, which quantifies how well we can monitor each population measure for each species group / species subgroup. We further provided a single value for each species group, calculated as the sum of scores across population measures as a proportion of the maximum score.

2.2 Results

An assessment of the current extent and quality of biological recording or monitoring data to provide a precise and accurate assessment of six population measures is summarised for each species group / species subgroup in Table 2.2 (see Annex 1 for raw scores).

There was a tendency for species groups and subgroups that scored well or poorly to do so across population measures (Table 2.2). In order, birds were the best recorded / monitored species group, whilst slime moulds were the most poorly recorded across population measures (Figure 2.1).

However, demographic rates and trends in demographic rates could only be produced with reasonable precision and accuracy for some birds, fish and mammal species. Population size could also only be estimated for some species groups, which in addition to birds, fish and mammals, included some estimates for butterflies, moths, dragonflies, damselflies, grasshoppers and allies.

2.3 Discussion

We provide an initial qualitative assessment of how well different taxonomic groups are monitored, with emphasis on the quality of data for assessing the population stock and changes in distribution, abundance and demographic rates. While there is variation within and between species groups in our ability to record and monitor populations, species groups can broadly be divided into three, illustrated in Figure 2.1. The first group includes birds, flowering plants and ferns, butterflies and macro moths, fish and bats, for which biological recording or monitoring is most well developed, and for many of which structured surveillance schemes are in place. The second group includes other mammals and amphibians and reptiles; species which are generally popular with recorders and so good numbers of records are submitted. The third group includes stoneworts, mosses and liverworts, lichens, invertebrates excluding butterflies and macro moths, fungi and slime moulds. Many of these species are challenging to record for a variety of reasons, for example identifying taxa maybe challenging, or sampling may be time-consuming, and hence recording is less extensive and less amenable to a structured sampling design approach. It is important to note that the scores do not necessarily reflect the accessibility of data. For example, for some mammal and invertebrate groups, data are not easy to gain access.

Within these broad groups, and where data are available, there are differences based on status. Rare and/or localised species are better monitored than average species in some taxa (notably amphibians and reptiles and non-volant mammals), but less well-monitored in others (flowering plants, birds and bats). Recorder interest in rare species is likely to be high in all cases and the differences likely reflect differing challenges in detecting, recording and collating observations, so the strategies needed to fill these gaps will likely be different. For rare species, the links between population status and environmental variables can be more straightforward to disentangle (because such species are often dependent on particular habitats and/or conditions and it is easier to cover a large proportion of the population (e.g. Aebischer *et al.* 2000). However, it can be harder to collect data, especially through broad-scale citizen science schemes; more targeted data collection is usually needed. On the other hand, some groups would benefit from better monitoring of common or widespread species. Efforts are underway to tackle some of these through the establishment of new schemes (e.g. the National Plant Monitoring Scheme), or by utilising technological advances (passive acoustic recorders (Newson *et al.* 2017b)).

It is worth bearing in mind that as knowledge of individual taxonomic groups increases the nature and number of gaps in our understanding of monitoring needs changes. Thus, for groups for which survey coverage is poor, the ‘unknowns’ might be unknown, beyond a need for basic information better characterising general patterns, but, as our knowledge increases the ‘unknowns’ become better known. For example, without already having reasonably good knowledge of bat ecology, we might not know that there is a specific need to know more about their hibernacula. Although the gaps might then appear more numerous, they will usually be better defined and hence easier to address. Pragmatically, improved biodiversity surveillance, and hence the decision-making processes that can then be informed, will likely require some reduction in both the ‘unknown unknowns’, through large-scale, ‘diffuse’ surveys, and the ‘known unknowns’ through more targeted effort, probably involving a subset of volunteers.

Recommendation: Data limitations vary by taxa, as do the levels of surveillance effort which are feasible, hence and hence the aims will differ; meeting the knowledge requirements will require maintenance of a diverse volunteer base to flexibly address such challenges.

Recommendation: For some taxa there is a relative lack of data for rarer species, while for others there is a relative lack of data for commoner species. Considering the results of our taxonomic gap analysis will help identify which groups may require additional investment in, or incentivising of, recording.

3 Existing survey schemes

This section briefly describes the main data sources used in quantitative analyses in sections 4 to 6. It also describes the data cleaning, filtering and preparation, including the standard trend production carried out for use in the analyses.

3.1 Structured data

The main data assessed here are annual surveys carried out by skilled volunteers in selected 1km squares. In most cases these provide data on abundance (not simply presence) and the data are used to generate national trends in abundance by the coordinating organisation and which can feed into national level indicators, such as the UK Biodiversity Indicator. We restricted these data to the ten most recently available years to ensure any inferences concerning coverage relate to the current situation. However, the different taxa surveys varied in the span of years they covered. In particular, the National Plant Monitoring Scheme only began in 2015. Bird, bat and butterfly data for 2017 had not been fully input and validated at the time of writing, so only data from 2007 up until 2016 were used.

In addition to the survey data, we also used independent atlas style data sets (i.e. records of presence/absence in an area) for each taxon to obtain a list of species present for each 10km square (or hectad) in the UK. These data were coarser resolution than the survey data - just presence or absence rather than abundance - but the coverage was more comprehensive and involved combining several years of data, as described for each dataset. Both the survey data and the atlas data were acquired for the whole of the UK; data from the Channel Islands were removed due to their distance from the rest of the UK and small size.

3.1.1 Common and widespread breeding birds

The bird survey data used in this project were acquired from the BTO/JNCC/RSPB Breeding Bird Survey (BBS, Freeman *et al.* 2007). In this survey, randomly selected 1km squares are surveyed twice per year, the first between mid-April and mid-May the second between mid-May to the end of June. Two parallel line transects are walked, ideally 500m apart and 250m from the boundary of the 1km square, but in practice, the precise transect route varied depending upon localised habitat and access restrictions. These transects are split into 200m sections and all birds within each 200m section are recorded within distance bands from the line. Square selection follows a stratified random design, with more squares selected in regions (roughly equating to counties) with high human population density. Weightings are used in the production of trends to account for the stratification. The atlas data set used to determine presence in a hectad was lists of probable breeding and confirmed breeding species in each hectad from *Bird Atlas 2007–11* (Balmer *et al.* 2013).

3.1.2 Bats

The bat survey data came from four surveys carried out by skilled volunteers targeting different species which collectively form the National Bat Monitoring Programme (NBMP). (Barlow *et al.* 2015). The NBMP is run by Bat Conservation Trust (BCT), in partnership with JNCC and Natural Resources Wales, and supported and steered by Natural England, Northern Ireland Environment Agency and Scottish Natural Heritage. The surveys are outlined in more details below:

- Field survey: 1km squares are assigned via a stratified random sampling approach. On two dates in July (at least five days apart) surveyors walk a 3km long transect in the 1km square, 20 minutes after sunset. Each transect is split into 12 sections; while walking the transect surveyors count the number of passes of noctule and serotine (or Leisler's bat in Northern Ireland) using a bat detector. At the end of each of the 12 sections they stop and do a two-minute point count of common and soprano pipistrelles.

- Waterways survey: Surveyors are allocated a grid reference along a water course >2m wide, and from this map out a 1km transect. Ten-point counts of Daubenton's bat passes are conducted for four minutes each at evenly spaced points along the transect in two dates in August (at least five days apart), 40 minutes after sunset.
- Roost survey: This consists of emergence counts from summer roosts at sunset or 15 minutes prior to sunset depending on the species. This survey targets Natterer's bat, serotine, brown long-eared bat, grey long-eared bat, common pipistrelle, soprano pipistrelle, greater horseshoe bat, lesser horseshoe bat.
- Hibernation survey: Two daytime visits are made to hibernation sites, one in January and one in February. Surveyors search for bats along a standard route around the site covering open areas and crevices. This survey targets Daubenton's, natterer's, whiskered/Brandt's/Alcathoë, greater and lesser horseshoe bats, barbastelle, Nathusius' pipstrelle and Bechstein's bat.

These surveys provide a mixture of presence/absence and count data. There is no atlas style hectad resolution presence data available for bats so instead we used a series of spatial polygons depicting each species' range recently generated by JNCC for the 3rd UK report for Habitats Directive Article 17 reporting.

3.1.3 Butterflies

Butterflies were surveyed via a mixture of transect and non-transect surveys as part of the UK Butterfly Monitoring Scheme (UKBMS), a partnership between Butterfly Conservation, CEH, and BTO, with fieldwork provided by thousands of volunteers. The UKBMS includes approximately 1,500 self-selected sites visited weekly for ~26 weeks per year. The same transect is walked and all species counted. A second component of the UKBMS, the Wider Countryside Butterfly Survey (WCBS), covers roughly 800 x 1km squares; squares are randomly selected within regions but only two visits per year are required. The same transect methodology is used, and around one-third of WCBS squares are also BBS squares, providing valuable col-location of recording. Additionally, there are three surveys to target butterflies that are very scarce, hard-to-detect (as adults) or in remote areas. Specifically, these surveys are egg counts, timed counts and larval counts. For atlas style hectad presence data we used the Butterflies for the New Millennium (BNM) data from 2006 to 2015 (Asher *et al.* 2001).

3.1.4 Plants

The plant data came from the National Plant Monitoring Scheme (NPMS, Pescott *et al.* 2019a), a partnership between Plantlife, CEH, the Botanical Society of Britain and Ireland, and JNCC. The NPMS involves the following tiers of participation: the Wildflower survey, the Indicator species survey and the Inventory survey. The Wildflower and Indicator surveys both use species lists to restrict the survey; the Wildflower survey species are a subset of the species included in the Indicator survey. For the Inventory survey, all species within the survey area are recorded. Volunteers are assigned a stratified random 1km square and asked to survey up to five plots of mostly 5m by 5m (although plots in some habitats are 10m x 10m or 1m x 25m), twice a year (spring and late summer). For hectad presence information we used data from the Botanical Society of Britain and Ireland (BSBI) flora atlas, which includes records since 1987 (https://bsbi.org/maps).

3.1.5 Data cleaning and filtering

For all taxa, any hybrid or exotic (neophyte) species were removed; additionally, family or order-level records were excluded as were records where the species recorded was not certain. For birds, seabirds were removed as the focus of this work is on terrestrial species, but we included coastal squares and some species found also in coastal habitats, such as curlew and oystercatcher. The bat 1km survey data includes records on non-target species for each survey; these were removed for the purposes of this analysis as these records are not used when calculating trends. For the plant 1km data we focussed on the list of 209 NPMS wildflower species, and we then used data on these species from all three tiers of the 1km survey scheme (Wildflower, Indicator and Inventory). The bat, butterfly and plant data occasionally had duplicate records, where a species was recorded twice in a 1km square in a particular year. These may be the result of repeat visits, different parts of the square being sampled, or different types of surveys being carried out (i.e. the bat field survey and the bat waterways survey). To ensure consistency between the taxa we removed duplicated records to ensure each species only had one data point per square per year.

3.2 Unstructured data

3.2.1 Datasets

The unstructured datasets came from the National Recording Schemes and Societies supported by the Biological Records Centre at CEH (Pocock *et al.* 2015). These are termed ‘unstructured’ because the overall dataset is not collected under consistent sampling methods, even though individuals or groups of recorders may undertake recording according to their own consistent methods (e.g. consistent coverage of all squares in a regional atlas).

There are 85 biodiversity recording schemes in Great Britain: they are defined by their taxonomic scope (e.g. Odonata, butterflies, lichens, *etc*.), mostly led by volunteers, and since most records are submitted by people voluntarily, we regard them as a form of ‘citizen science’, albeit that sufficient expertise is required to accurately identify the species, so only a subset of ‘citizens’ currently participate. We regarded each recording scheme (or taxa within recording schemes, where these are recorded differently, e.g. bees, wasps and ants) as an individual ‘project’. The records in each project’s database have undergone quality assurance by experts in the recording scheme. We treated data from each project as independent from all others.

The datasets comprise lists of species recorded in ‘visits’. A ‘visit’ is a list of one or more species reported at a specific place (here, a 1km UK Ordnance Survey grid square) and time (here, a specific day). Therefore, visits to the same 1km grid square on the same date are considered to be a single visit, even if they are from different recorders or different locations within the grid square. Records that were not specific to species, a date or to a 1km grid square were excluded. Species within the taxonomic remit of each project but not reported on a list are inferred as ‘non-detections’ (i.e. an ‘absence’ in the terminology of ‘presence-absence’ data); many of these will be ‘true’ absences (the species was not present), others will be ‘false’ absences (i.e. the species was present but not searched for, or searched for but not detected) (see Table 3.1). Revisits to sites within a year provide information to estimate detection probability (Mackenzie & Royle 2005).

3.2.2 Data preparation

We used trend outputs from Bayesian occupancy analyses and the species occurrence datasets used for these analyses. The analyses had already been run for a total of 10 967 species from 34 projects as part of existing work (median species per project = 168; range =

9 - 1002); 5,293 of these trends are publicly available (Outhwaite *et al.* 2019). These analyses were run for data encompassing the 40 years to the end of 2016. Recording intensity has increased over these four decades, so we also used the last ten years of each dataset and of each occupancy trend output. Each species was therefore included twice: for the 40-year and ten-year trend.

Occupancy models are a valuable approach for analysing occupancy data (Isaac *et al.* 2014; Mackenzie & Royle 2005) but can require a lot of data when reporting annually varying estimates of occupancy (=presence). Recent approaches have implemented a random walk within the time-based occupancy which acts as a data smoother, so allowing occupancy trends to be modelled over time even with relatively sparse datasets (Outhwaite *et al.* 2018). Of course, the successful modelling of a trend does not mean that the trend is representative (see Discussion).

4 Rules of thumb

In the assessments of sampling coverage in sections 5 and 6 we had to decide when we had sufficient data to produce trends. To do this we applied rules of thumb to the datasets. Rules of thumb are used as an indication of when there is sufficient data for a species to warrant production of a trend or other metric. The existence and derivation of rules of thumb varies between structured and unstructured data.

4.1 Structured data

For the analysis of structured data we used a rule of thumb of an average of 30 counts (or occupied sites) per year. This threshold is routinely used in the production of regional trends from the BBS (Harris *et al.* 2017) and contrasts with the threshold of 40 sites per year used for national analyses. For butterflies, due to the scarcity of some species a lower threshold of five squares is sometimes used (M. Botham, pers. comm.). However, for consistency we measured coverage by applying a threshold of an average of 30 occupied sites per year across all taxa.

4.2 Unstructured data

Previously there have been guidelines for the use of occupancy data for analysis (Mackenzie & Royle 2005), although these are typically applied to designed studies when allocation of recording effort can be controlled. In our case, recording effort is ‘opportunistic’, but our models shared information on detection and occupancy between years (Outhwaite *et al.* 2018). Therefore, it was valuable to develop ‘rules of thumb’ indicators for when these unstructured occurrence data were suitable for analysis. This work also enabled us to draw out recommendations on how to improve sampling coverage to maximise the number of trends being produced. This is summarised here, but for full details see Pocock *et al.* (2019).

The construction of rules of thumb for unstructured data required answering three questions:

a) When are outputs from occupancy analysis good enough?
b) What metrics from the datasets should be used to distinguish good from bad outputs?
c) How are the dataset metrics used to distinguish good from bad outputs?

4.2.1 Criteria to define good outputs from occupancy models

Firstly, we needed to define which of our occupancy trend outputs had ‘adequate’ precision based on user assessment. Outputs that did not meet this precision threshold were deemed too imprecise to be useful. Note that this was not a description of accuracy (i.e. if the occupancy model was mis-specified, or the data were biased, the trend outputs could be precisely estimated but deviate from the 'true' value of occupancy). It would have been too time-consuming to individually assess each of the trend outputs, so we decided to classify a subset of the outputs and use these to statistically define the threshold. To classify the subset of trend outputs we asked three experts (Nick Isaac, Charlie Outhwaite and Gary Powney) to assess 100 selected trend outputs from across the range of precision and occurrence values of our Bayesian occupancy analysis outputs. These three experts have worked together on projects using these occupancy analysis outputs in the past, e.g. the UK's *State of Nature Report* (Hayhow *et al.* 2016), and so they were not strictly independent from each other, but they undertook the assessments individually. Each person was provided with images of the trend outputs showing yearly occupancy estimates with 95% credible intervals and information on convergence of the parameter per year.

We considered precision of the outputs to be acceptable when two or three of the three experts scored them to be acceptable. The result of decision tree analysis was that the threshold for 'acceptable' precision was when mean annual precision of arcsine-transformed posteriors (hereafter 'arcsine-transformed precision') >70.4; we defined this value as the 'precision threshold' and used it in the remaining analyses. Full details are in Pocock *et al*. (2019).

4.2.2 Metrics used to describe data

We developed seven metrics, each describing different attributes of the data that we predicted would influence the success of occupancy analysis. These metrics included different aspects of the dataset including records, visits and non-detections (Table 4.1).

4.2.3 Using metrics to construct rules of thumb to predict acceptable and high precision trend outputs

We calculated these data metrics for each of 15344 species occupancy trends and classified each output according to whether it met the precision threshold. We used decision tree analysis to construct simple rules: using the dataset metrics (Table 4.1) to predict whether trend outputs would meet the precision threshold.

Prior to calculating the metrics, we undertook 'site filtering', following the method for calculating the trends, because without this the power of the occupancy analysis was reduced (see Isaac *et al.* 2014). Specifically, we removed sites within each project where records had been obtained in only one year across the whole dataset (even if there had been multiple records of one or more species within that year). In doing this, we removed an average of 32% of all records from projects (range: 7-65%). More records were removed for taxa that are relatively poorly recorded (e.g. centipedes) or require specialist skill for their identification (e.g. bryophytes) (Table 4.2).

We assessed the classification success of the decision tree as specificity (also called the true negative rate; the proportion of trends below the threshold that were classified as such) and sensitivity (also called the true positive rate; the proportion of trends above the threshold that were classified as such). Higher specificity provides a higher confidence that a species meeting the simple rules will have outputs above the precision threshold, in other words, higher specificity provides a more conservative decision. Higher sensitivity gives a stronger guarantee that species that could have outputs above the precision threshold will meet the threshold, i.e. fewer acceptable datasets are discarded unnecessarily.

One method of adjusting the specificity of the decision tree is to give the data points above and below the precision threshold different weights. We therefore ran two decision trees. One tree ('equally-weighted') weighed the two classes equally and so sensitivity and specificity were balanced. The second ('high specificity') weighed data above the precision threshold ten times those below. This allowed us to preferentially prioritise high specificity rather than high sensitivity, and so provided a more conservative target for our data, i.e. the data that passed the simple rules would have a high probability of being suitable for modelling, in having precision that exceeded the precision threshold.

The classification trees resulted in the following rules of thumb (full details are available in Pocock *et al.* 2019):

- For very high confidence (98%) that a trend output from unstructured data will meet the precision threshold, the 10% best recorded years need at least 29 records of the focal species per year (plus other records from the taxonomic group, permitting the inference of non-detections).
- There is a good confidence (80%) to produce trends that meet the precision threshold if the 10% best recorded years have at least 7 records per year.
- More rarely-recorded species (<1-4% of visits) make up the majority of species in the dataset and require even less data (above 10 or 3 records in the 10% best recorded years, for very high or good confidence, respectively).

These classifications are illustrated in Figure 4.1. It is notable that information on both the presences and the absences was retained in the rules of thumb.

4.2.4 Precision is affected by the number of records

It is important to note that the thresholds from the classification trees for acceptable and high precision should not be considered as a target, but as a minimum, because there is an overall trend that precision is increased when the number of records is increased. Specifically, the lower bound of precision (from a quantile regression) is positively related to the number of records per year: 5^{th} percentile of arcsine-transformed precision = 12.155 x (90^{th} percentile yearly number of records ^0.649) (Figure 4.2). However, there is great variation and even species that typically have a relatively low number of records recorded each year can have high precision. This is to be expected given the complexity of the unstructured datasets, which we have simplified to a couple of metrics. Also, as the number of records increases, we anticipate that the number of sites with revisits within a year and the spread of sites would increase, so increasing the likely accuracy and representativeness of the estimates.

4.2.5 Outputs with good precision are not necessarily accurate or representative

Despite the positive potential for occupancy modelling, because site selection is not subject to a sampling design, trend outputs with good precision does not mean that they are accurate or representative of the species (Mackenzie & Royle 2005). Obviously, it is impossible to determine accuracy without an independent measure of the true occupancy and its trends (and if that was available, we would not be undertaking this analysis anyway). However, it is important to ask whether the trends can be considered representative. This relies on expert assessment of the dataset compared to the expected distribution of the focal species.

Two key questions to consider whether occupancy trend outputs with good precision should be considered further:

1. Are the occurrence and non-detection data distributed across the range of the species in the region of interest? Note that the data do not need to be distributed across the whole region of interest if the focal species is limited in its range.
2. Are the visited sites a representative subset of the region of interest? Due to the nature of recorders' behaviour, the sites visited are likely to be biased, e.g. with disproportionately more visits to nature reserves than to farmland. The key question is whether species trends are likely to be different across different types of sites, because if they are then biased recorder behaviour will lead to biased estimates of trends.

It requires expert assessment to consider whether the outputs with acceptable or high precision should be considered further. Care should be taken though about the circularity in decision-making, otherwise only the trends that agree with the expert opinion will be considered.

4.2.6 Lessons from our 'rules of thumb' for rapid assessment of data for occupancy analysis

For our approach of assessing outputs from Bayesian occupancy analysis run by CEH, we could derive some key recommendations for these types of datasets.

Recommendation: Sites should be visited in more than one year, and some sites need to be revisited within a year. In our analysis an average of one-third of records for a taxon were removed prior to analysis because they were from sites that had been visited in only one year. The opportunity is that if sites that have been visited once are revisited, then both the current and the prior records become available for analysis.

Recommendation: Analysis of unstructured data could be effective because surprisingly few data are the minimum requirement for estimates of yearly occupancy with our Bayesian occupancy analysis to meet the precision threshold (29 records in the best-recorded years to be confident that the trend will have acceptable precision: less for the rarest species). However, care needs to be taken with interpretation because trends are not necessarily accurate or representative.

Recommendation: With unstructured records, it is generally better to have more records, for increased precision. However, further work is required to assess which has greater impact on the bias of the outputs between having more records from the same place and increasing spatial coverage.

5 Coverage of the schemes by habitat and region

5.1 Methods

The aim of these analyses is to determine how coverage varied by region and the degree to which monitoring data for different taxa can be used to infer status in specific habitat types of interest. Specifically, we applied the rules of thumb to the datasets to assess where we could produce trends for different taxa within different regions and habitats.

5.1.1. Assigning sampling to regions

The Nomenclature of Territorial Units for Statistics 1 (NUTS1) boundaries were used to determine regions: nine of these regions are in England, with Scotland, Wales and Northern Ireland each counting as a region. NUTS1 regions were selected as they are widely used in policy, well established and large enough to potentially contain sufficient records for generating species trends, yet small enough to allow regional variations to be detected. A shape file of the NUTS1 regions was overlaid onto a 10km square grid and each hectad was assigned to the NUTS1 region that covered more than 50% of its area. This information was then used to assign each 1km survey square a region. For the structured data, the atlas hectad data was then used to filter 1km square records for each habitat, removing records for species where there was no evidence of presence in the parent hectad from the atlas data. The aim of this process was to remove transient anomalies in the dataset likely to be due to migratory passage or chance translocation that does not accurately represent a species' established range. From these filtered 1km square records a species list was generated for each region of those species for which we have breeding evidence/ evidence of presence from the atlas.

5.1.2. Assessing potential for trend generation by region

For each region-specific species list, we calculated the percentage of species for which our rules of thumb determine that a trend could be generated. For the structured data, these were the species with an average of 30 or more records per year in that region. For the unstructured data we applied the rules of thumb described in Section 4.

5.1.3. Assigning sampling by habitat

The 2015 Land Cover Map (LCM2015, Rowland *et al.* 2017) was used to determine habitat coverage of surveys due to its wide use and policy relevance. Habitat was classified into nine broad classes (see Table 5.1 for more detailed classifications); we did not include seawater. Then percentage cover was calculated by summarizing the LCM2015 25m resolution raster, each 1km square includes 1600 25m pixels. As seawater is not included, and integer values are given which induces some rounding errors, cover for some squares will not necessarily sum to 100%. The LCM2015 covers Britain and Northern Ireland but it does not cover the Isle of Man. Different habitats may have very variable coverage with some habitat never occurring in large amounts (Table 5.1). Because of this, we classified a 1km square as a particular habitat type if it covered more than the median percentage cover for a particular habitat, calculated over all 1km squares in the UK (excluding squares with 0% cover of that habitat); Table 5.1 gives the median percentage cover values. Note that using this method it is possible for a 1km square to be classified as two or more habitat types, hence, for example, a square could contribute to knowledge about woodland and arable habitats. This is a practical means to obtain an overview across all surveys and squares in a consistent way, but it should be noted that individual surveys do not cover the whole surveyed square so the area sampled may have a different habitat composition to that of the overall square.

All survey squares for each taxon were included here regardless of the survey method, producing an assessment of sample sizes for individual habitats. However, it should be noted that there may be quite a bit of variation in the coverage of the 1km squares included. For example, the BBS is considered to fully cover a 1km square whereas some of the butterfly structured non-transect methods are much more targeted to specific habitats. As some of the bat data from hibernation surveys were highly sensitive and could only be provided at 10km resolution, these locations had to be removed from the analysis. Note also that these analyses measure coverage without reference to stratification techniques used in trend models. Hence, although we may find that a survey technically oversamples a particular habitat, this may be planned and accounted for in the production of population trends by that scheme.

5.1.4. Assessing potential for trend generation by habitat

Though general metrics on habitat coverage are useful to give a broad overview, the most critical knowledge is whether coverage for each habitat is sufficient to calculate a habitat specific trend. To assess this, we followed the same process as used for the regional trend analysis above: for each habitat we generated a list of species established there from the 1km survey squares included in that habitat (> median %cover for a habitat), this list was filtered by the hectad atlas data to remove transient records and then we used the rules of thumb (for structured and unstructured data) to work out the percentage of species for which it was possible to generate a trend.

5.2 Results and discussion

The UK covers an area of 247,567.3km^2; in terms of 1km grids this translates to 257,529 squares as coastal squares may be partly over the sea.

5.2.1 Regional coverage with structured data

The bird surveys have the largest coverage, followed by butterflies, then bats, then lastly plants (Table 5.2). However, the plant survey has also been running for the least time (only since 2015). The number of squares covered varies between years, though this variation is minimal (Table 5.2). For birds, coverage fluctuates around an average of 3,500 squares per year, but dipping during the period of fieldwork for the 2007/11 BirdAtlas, as volunteers prioritised that fieldwork, suggesting a finite capacity among volunteers. For bats the number of squares covered has decreased by 10% since 2010, something which BCT is reviewing; the number of squares surveyed for butterflies has increased by 30% in a similar time period. The decrease in coverage of bats may be due to changes in survey technology (less use of heterodyne detectors), but it is anticipated that the forthcoming British Bat Survey based on full spectrum static acoustic detectors will lead to greatly enhanced coverage.

Birds are the only taxon for which it was possible to calculate trends for at least some species across all regions; on average it was possible to calculate trends for 41.6% (+/- 1.7%) of bird species across all regions (Figure 5.1). Butterfly trends were possible for at least some species in all regions apart from Northern Ireland, but sampling effort was the most variable between regions: the proportion of species it was possible to generate trends for ranged from 0-68%, with sampling biased towards the south of England. There was generally poor coverage in Northern Ireland: with our rules of thumb for structured data no trends could be generated for bats, plants and butterflies[2], but coverage of bird surveys in this region was not substantially lower than other regions. No trends could be generated for bat species in four regions (Figure 5.1), and the percentage of species possible to generate trends for was less than 27% in all other regions, with the exceptions of Scotland (43%) and southeast England (40%). Plants had least coverage; it was only possible to generate trends for any species in four regions (Figure 5.1), and even in those regions the percentage of species trends could be generated for was low (max 8.1%). It must be understood that this monitoring scheme was set up to address the policy need of assessing habitat quality, rather than assessing plant abundance directly. It was designed for this purpose, and the design

[2] Note, this is slightly different to results currently reported through the NBMP and UKBMS, where trends are published for one bat species and several butterfly species. This is likely due to slightly different data/filtering protocols and approaches being used in this analysis. However, the overall message of poor coverage for producing trends in NI for these taxonomic groups is consistent.

(plot-based recording, with a focus on indicator species) would have limited its ability to provide trends in abundance of plant species, but see Pescott *et al.* (2019b) for further consideration of the use of these data. In addition, the scheme had only been running for three years at the point at which we extracted data for these analyses, so there had been less time to encourage uptake and publicise the scheme. The region where most plant trends could be generated was southeast England, where the human population density is higher, giving a larger pool of potential surveyors to engage. In a similar way London could potentially be an area where it would be possible to recruit enough volunteer surveyors and the smaller number of species likely to be present in the city would potentially reduce the skill level required, opening the surveys up to a wider range of potential participants. Free training days or well-publicised simplified unstructured surveys are possible ways to engage more people. These could then support the smaller sample of more structured data considered utilising current analytical developments that increasingly allow structured and unstructured data to be combined to provide more robust trends and inference.

Across all taxa, coverage was poorest in the largely urban areas of London and the West Midlands of England, and the more rural regions of Northern Ireland and North-east of England. Coverage was best in the two southern regions of England, which have good recorder bases. Northern Ireland and the North-east of England are relatively sparsely populated, so it is likely that a shortage of recorders accounts for at least some of the lack of coverage in these areas. Clearly, though, simple population density is not a good predictor of survey coverage since coverage is also poor in the most densely populated areas of England (London and the West Midlands). This could be for one of two reasons, either people in urban areas are less interested in (local) wildlife, and so less willing to participate in surveying activities, or they perceive such wildlife to be less ‘interesting’ or ‘valuable’ to survey. This is not the case, urbanisation is one of the largest threats to biodiversity (Maxwell 2016), and biodiversity in urban areas is declining (Seto *et al.* 2011). Urban areas can also contribute to biodiversity assets, through the use of green infrastructure and other planning initiatives (Andersson *et al.* 2014), although not without risk as sometimes habitats that appear suitable are not (Garmendia *et al.* 2016), so understanding the requirements of biodiversity (though survey monitoring and other activities) is of critical importance. Provision of such areas not only increases biodiversity but also has added societal benefits such as improved human health, which engaging with monitoring and related activities can help deliver (Tzoulas *et al.* 2007). Targeted education and promotion of the value of such biodiversity, and the need to know more about it, may help to redress the balance.

5.2.2 Regional coverage with unstructured data

We applied the rules of thumb for the unstructured data to predict which species would be modelled with acceptable or high precision within each region (Figure 5.2). In doing this we strictly subsetted the data by regions. An alternative approach to our strict subsetting by region would be to share information on detection probability across regions. This has the benefit of making efficient use of the data and allowing more occupancy trends to be estimable with good precision. We expect that sharing information in this way would almost always result in more trends having acceptable or high precision, so our results (Figure 5.2) should be taken as a conservative estimate. Because this was a rapid assessment, we assessed only the predicted precision of the outputs and not the representativeness of these trends (see Section 4.2.6).

Despite these potential limitations, the outputs do provide a helpful way to broadly consider gaps and opportunities in biodiversity recording at a regional level in GB (Figure 5.2, there were insufficient data for Northern Ireland). As expected, there was substantial variation across taxa and across regions in the proportion of species predicted to be modelled with high or acceptable precision. The variation across regions is unrelated to human population density of the region, although the South-east of England is the region that tends to be best recorded and has highest accessibility to the population due to its proximity to London. There are clearly other factors that are needed to explain regional variation because South-west England is well-recorded despite having low population density, while North-west England has a relatively high population density (at least in parts) but is relatively poorly recorded. Both Wales and Scotland are more poorly recorded and have low population density. The variation across regions is particularly important because environmental policy is devolved to the four countries of England, Scotland, Wales and Northern Ireland (three of which are considered as 'regions' here under the NUTS1 regional classification), and there is increasing impetus to develop within-region policies, e.g. Area Statements within Wales. It is also partly scientific, because regional trends can provide a better understanding of how biodiversity is affected by potential drivers of change, many of which vary across GB.

As expected, there is also great variation across the taxonomic projects. A few taxa are well-recorded (e.g. butterflies and moths) and many species in those taxa are predicted to have high precision trends across many regions. Some taxa are reasonably-well recorded with the majority of species predicted to produce outputs that are at least acceptable (and so likely to be useful for multi-species indicators) in many regions; these include pollinator groups such as bees and hoverflies that are the focus of much recent attention. However, for many taxonomic groups, most species are predicted to produce trends with poor precision, making it unlikely that they are suitable for regional analysis. Despite this, up to one half of species in these groups are predicted to produce trends with at least acceptable precision at the extent of GB. These would need to be assessed for their representativeness, but this does demonstrate the potential for unstructured recording in GB to produce acceptable outputs, e.g. for multi-species indicators, for an incredibly wide range of taxa. Further consideration would enable us to understand why some region/taxa combinations are particularly well-recorded (e.g. wasps in South-east England, aquatic bugs in the East Midlands or Gelechiid moths in North-east England); it is likely that this is due to one or more particularly keen recorders in those regions. It could also help identify key gaps that could be filled with targeted support and recruitment of volunteer recorders (e.g. bees in the East Midlands).

5.2.3 Habitat coverage with structured data

We first considered the representation of habitats within the structured schemes, and whether this was consistent over time. Over all the years considered, the number of 1km squares covered by schemes holding a substantial proportion of a particular habitat (defined as greater than the median value of the habitat occurrence in all monads in the UK) are shown in Table 5.3; this is also shown year by year in Figure 5.3. In general, the coverage of squares with more than the median value for a habitat matches the UK distribution fairly well with the exception of built-up areas and broadleaved woodland, which are over-sampled for all four taxa and mountain/heath/bog which is under-sampled for all four taxa, probably due to difficulties in access. For the bird surveys (BBS) this apparent oversampling of certain habitats is at least partly an outcome of the survey design, the stratified random design of BBS purposely samples lowland areas, which tend to be more populated and so have a greater pool of volunteers, at higher intensity, with this then accounted for in trend production. For all schemes, the habitat coverage does not vary substantially between years, though there are a couple of small deviances. For example, in 2011 bird sampling is biased more towards built-up areas and broadleaved woodland at the expense of semi-natural grassland and mountain heath and bog. For butterflies, in 2009, when the Wider Countryside Butterfly Survey was introduced to the UKBMS, the oversampling of broadleaved woodland becomes less pronounced, and arable habitat which had been under-sampled becomes a lot more representative of the UK level.

The percentage of established species for each taxon for which it is possible to calculate trends is displayed in Figure 5.4. Across all taxa, broadleaved woodland has the best coverage, closely followed by arable and improved grassland. Mountain/heath/bog has by far the worst coverage. For birds and butterflies it was possible to generate trends for some species for all habitats, but for bats and plants, trends could not be generated for any species for four of the habitats with lower coverage (coastal, coniferous woodland, and semi-natural grassland for both groups, and additionally freshwater habitats for plants and mountain/heath/bog for bats (and also for very few plant species). Bats showed the highest between habitat variation in the percentage of species trends could be generated for (0–64.3%), closely followed by butterflies (16.3–72.4%). As found from the regional analysis, the plant survey suffered most from under-sampling, with this group having the lowest percentage of species for which it was possible to generate habitat specific trends for all habitats.

It is important to note though that because a species was recorded in a square with a higher than average coverage of a habitat, this does not mean that the species was recorded in that habitat, or that it even occurs in that habitat. As squares are assigned to habitat types based on the median percent cover of a habitat, for some habitats only a small amount is needed for a square to be assigned to it (e.g. 3% for freshwater or 4% for built up areas).

It is not entirely surprising that the under-sampled habitats are those which are more remote and therefore harder to access. For BBS the Upland Rovers scheme, where volunteers were encouraged to visit under-sampled squares and allowed to visit these remote squares just once rather than twice, has gone some way to improving coverage. Other options have been considered but all involve greater departures from the original BBS study design (e.g. swapping uncoverable random squares with nearby similar squares) but the preference has been to test other low impact interventions first so as to maintain the “gold standard” of the scheme. Similar practices could be introduced to the other schemes to encourage coverage in remote habitats. There is also the option of paying for professional surveyors to fill in significant coverage gaps, as was successfully undertaken in England for a few years to improve coverage in upland areas, though this is obviously funding dependent.

5.2.4 Habitat coverage with unstructured data

We applied the rules of thumb for the unstructured data to predict which species would be modelled with acceptable or high precision within each habitat (Figure 5.5). All the limitations applied to the regional analysis also apply to this analysis (see Section 5.2.2).

In general, there are four habitats that appear to have relatively good species coverage: broad-leaved woodland, built-up areas, intensive grassland and arable. These are the same as with the structured data. These are also the habitats that have the highest number of squares, which will influence the ability to produce trends. This is useful because here we are interested in the ability to produce trends, rather than, for instance, the relative intensity of coverage of these habitats.

It is important to remember, as noted previously, that because a species was recorded in a square with a higher than average coverage of a habitat, this does not mean that the species was recorded in that habitat, or that it even occurs in that habitat. (This is why we can obtain trends for some fish for intensive grassland.) These figures would need careful interpretation for future action.

Recommendation: Further investigation of the reasons for regional and habitat variation in the rates of recruitment to schemes would be valuable. While some less densely-populated regions have lower coverage, as expected, there is not a clear association between human population and coverage: regions such as northern England and London have lower coverage. This may have been influenced by targeted training or volunteer recruitment and retention.

Recommendation: Fine scale data on habitat is important to provide habitat-specific trends, and likewise impacts of interventions within habitats. Our analysis was limited in that all data was provided at the 1km resolution, so we could not specifically determine the actual habitat where the record was made. Habitat information has been collected for some structured schemes and used in some previous analyses. Habitat information associated with each high spatial resolution record could be valuable but may incur biases in recording. It would be valuable to consider the strengths and limitations of habitat-specific recording.

6 Fine-scale spatial coverage for structured data

6.1 Methods

Two main analyses were conducted to assess fine-scale spatial coverage. One sought to assess spatial variation in the proportion of species in a hectad for which published national trends were available. The other focussed on estimating the spatial precision with which individual species trends could be produced using the rule of 30 counts to produce a trend. Together these measures help in assessing where coverage is adequate and poor for each taxon and may identify multi-taxa coverage gaps.

6.1.1 Proportion of species for which there are published national trends

Maps were produced of the percentage of species established in each hectad (from the atlas data) for which a national trend has been published (Brereton *et al.* 2017; Harris *et al.* 2017; Bat Conservation Trust 2018). As the plant data has only been collected since 2015, no national trends on plants are yet published. It is important to note here that the number of records required for published trend is not always equivalent to our rule of thumb of an average of 30 squares per year. Some species of butterfly are very rare and localised so as few as five samples have been used to produce a trend in these cases (M. Botham pers. comm.). Nevertheless, the aim of these analyses was to give a standardised overview of the spatial pattern of coverage for each taxon.

6.1.2 Spatial precision potential for individual trends

To investigate the spatial precision for which individual species trends can be produced, first the 1km survey data for each taxon were aggregated to hectad level. Atlas data were used to generate a list of established species for each hectad of the UK and this list was then filtered by the survey data to remove any species that were not recorded in the hectad-level survey data despite the evidence of their presence from the (more comprehensive) atlas records. Next, for each of the listed species within a hectad, the minimum distance from the hectad centroid needed to get an average of 30 occupied survey sites for each survey year was determined (see Figure 6.1). The distances to hectad centroids were used as opposed to distances to the 1km square centroids as it reduced the computational intensity of the calculation efficiently while resulting in minimal loss of accuracy. However, any assessment of the distance needed to get a certain number of samples will present coastal areas and small islands as having a low coverage because there is less land in these areas and therefore the distance needed to reach the threshold level of counts will be higher. To reduce this effect, additional maps were produced with the distance value for each species in each hectad standardized by dividing by the distance needed to find 30 hectads from the focal hectad if all squares in the UK were included (Figure 6.2). The results were summarised for species across all hectads, and for hectads across all species.

6.2 Results and discussion

6.2.1 Proportion of species for which there are published national trends

Across the whole of the UK the respective atlases recorded 200 bird species, 65 butterfly species, 17 bat species and 209 plant species in the NPMS *Wildflowers* survey level. Published trends are produced for a high proportion of established butterfly species (mean 99.8%) and this does not show much spatial variation (range 90–100%)(Figure 6.3). For birds, the percentage of established species for which published trends are produced shows more spatial variation ranging from 0–100%, mean = 90.5%. The Highlands of Scotland and the Outer Hebrides are two of the areas where the lowest percentages of established bird species are reported in national trends. This could be due to two factors, firstly it may represent a higher proportion of the communities being made up of scarce and localised species (such as habitat specialists). Secondly, it may show species which are heavily impacted by anthropogenic change, and therefore are now restricted to more remote areas. A small percentage of surveyed bat species are included in published trends compared to birds and butterflies (mean for bats = 72.5%). The lowest proportion of established bat species for which national trends are produced is in Northern Ireland. Atlas data for bats is missing for the Highlands of Scotland and the Outer Hebrides. There were no plants with published national trends because the NMPS is too recently a launched survey for trends to be published.

This analysis would suggest that efforts need to be made to improve bat survey coverage, especially in the more remote areas in Scotland. Currently there are four main surveys in the bat monitoring scheme. To a large extent different people are involved in the different surveys (at least field/waterways surveys compared with more intensive e.g. hibernacula checks), and each has value in its own right. The proposed British Bat Survey (BBats) using static detectors, has the potential to deliver more comprehensive national bat monitoring. Static automated bat detectors can be used to record all bat species and may be an option to improve coverage in some areas, and it should be noted that deploying detectors is likely to appeal to a different subset of (potential) recorders, which may also broaden participation. In addition, not all species can be easily distinguished on the basis of audio calls alone (e.g. *Plecotus* and some *Myotis spp*), although there are continuing improvements in knowledge and understanding of species identification, such that this is less likely to be an issue in the future.

Recommendation: It would be valuable to consider how different recording approaches may attract different types of recorder, as in bat recording where there are different levels of commitment required for different types of recording. Focussing on multiple approaches may support an increase in coverage but would require investment to combine the data for trend outputs.

6.2.2 Spatial precision potential for individual trends

We were able to conduct the spatial precision analysis for 58% of bird species, 71% of butterflies, 53% of bats, and 33% of plants in the NPMS Wildflowers survey level, out of the species in the atlas lists. There are a few species in each taxon which are present according to the atlas data but for which there was no survey data (7 bird, 6 butterfly, 1 bat and 2 wildflower species). There are also a number of species for which there are survey data but for which it is not possible to calculate trends, even if data across the entire UK is acquired (78 birds, 13 butterflies, 7 bats and 138 wildflowers). Most of these species, especially in the case of birds, are vagrants, very recent colonisers, or species with very small and localised populations. The lack of sample size for plants compared to the other taxa is again evident here. Butterflies appear to have better coverage than birds here, and birds appear to have a similar coverage to bats, but it is important to remember that these figures are given relative to the atlas list of species for each taxa and the quality of this atlas data varies between taxa.

Birds show the highest inter-species variation in the distance needed for 30 counts, then plants and butterflies with bats showing the least variation between species (Figure 6.4). But the proportion of inter-species variation in spatial coverage increases with the number of species under consideration, as would be expected. In general, 250km from a site is far enough to get 30 samples for the majority of species across birds, bats and butterflies. For plants, however, this value is closer to 300-400km, again emphasising the low coverage of plant surveyors compared to the other schemes.

The maps using distance (Figure 6.5) are qualitatively similar to the maps using standardised distance (Figure 6.6): standardising the data to remove coastal effects makes very little visible different at the scale the maps are displayed here. Considering all taxa, coverage is lowest in Scotland and highest in southern England, though for butterfly coverage in Scotland is better than for the other taxa (Figure 6.5). Coverage in Northern Ireland is also low compared to other areas for bats, butterflies and plants, but for birds is as good as for southern Scotland. Over all taxa, the best spatial coverage is for birds, with 100km sufficient distance to obtain 30 samples for the majority of hectads in the UK. Butterfly coverage is also good, with 150km sufficient distance to obtain 30 samples for most areas, but coverage in Northern Ireland is poorer. For bats, coverage in the southeast is good, but coverage in the rest of the UK is poorer. A distance of 180km is required to obtain 30 samples for the majority of hectads, but as atlas data is missing for a good proportion of Scotland and some of England this figure is actually likely to be higher if the missing hectads were included. For plants, coverage is currently not high anywhere in the UK, with the shortest median distance to get 30 samples 134km and for the majority of squares, distances over 300km required. Therefore, the same messages are apparent here as in earlier analyses, if we are relying on structured monitoring, survey coverage would need to be improved for plants and bats. Generally, coverage of structured monitoring in Scotland and Northern Ireland is lower than for the rest of the UK.

Recommendation: Consideration should be given to targeting volunteer recruitment in areas of poorer coverage (darker colour in Fig 6.5) to increase spatial representativeness.

7 Main discussion

The UK's public environmental bodies use biodiversity information for statutory reporting, and for operational and strategic needs. There is, thus, a requirement to ensure that such information is balanced and comprehensive, providing a robust evidence base for environmental and conservation management decisions. Overall, a recent review found there was confidence in the current biodiversity surveillance at a national level, which helps organisations to meet their reporting obligations and some of their operational needs (Pocock 2018). In addition, there is a need for suitable monitoring data, such as from the JNCC-supported surveillance schemes to assess the impact of conservation and policy interventions, which requires suitably comprehensive spatial and taxonomical coverage by the surveillance schemes, especially if appropriate baseline data are to be available. However, as this report highlights, there are gaps in coverage. We identify areas where there is relatively poor coverage, both spatially (by region and habitat) and taxonomically.

For most purposes, species are the fundamental unit of assessment and, in general, effective action needs to be targeted at particular species' needs. For rare species, or those deemed of sufficient 'importance' (e.g. listed under Schedule 1 of the Wildlife & Countryside Act), there is clearly a need for knowledge of those individual species trends. However, policy interest is typically in assessing wider environment/ecosystem health and the extent of

Natural Capital (however defined) that is present. Assessing the state of a particular ecosystem (whether that be a functional one, such as a particular habitat type/location, or a broader one, such as a regional assemblage) might be achieved by considering trends averaged in some way over a given set of species by defining a wider ‘functional’ group, but, in order to generate such a multi-species indicator, the individual species trends must be known. Combining species trends also introduces additional issues to consider, namely of representativeness, relative data quality and appropriateness of analytical methodology (Norris *et al.* 2016). Similarly, to effectively target management actions, individual species responses need to be predicted and, indeed, these can provide insight into the mechanisms underlying wider changes, particularly where the demographic causes of the changes are known. A key knowledge gap that remains, however, is how does species occurrence/ abundance translate into Natural Capital? Consequently, there is a need for classification by functional groups, such as has been developed for pollinators, understanding of what general proxies may be useful, and identification of the most critical monitoring targets to prioritise those, although this would need to be a substantive piece of work.

Recommendation: Further work should be undertaken to relate species and taxa to functional groups, to support reporting of natural capital.

In understanding biodiversity responses to environmental change (anthropogenic or natural), there is a hierarchy of information by which responses can be characterised or understood, and hence the data that are needed:

1. Where do things occur, and is this changing?
    - This may be addressed using either structured or unstructured data, although the benefits of each depend on the specific question.

2. How many are there, and is this changing?
    - This typically requires structured data

3. Why are things changing?
    - This can be addressed in various ways, ranging from inferring mechanisms via correlations between trends and drivers (Woodcock *et al.* 2016) through to bespoke detailed research that tests mechanistic causes of changes in vital rates (demography).

The population status of many species varies spatially, either geographically or by habitat (Massimino *et al.* 2015), as a result of the interplay between land-use and climate patterns and changes therein. Quantifying these spatial trends in abundance/occurrence is important for three reasons. Firstly, it provides greater understanding of the causes of change, as drivers are similarly likely to vary spatially, and comparing/contrasting different population trajectories can help disentangle these multiple factors. Secondly, regional (or habitat-based) trends can provide important context to local site-based trends, such as on designated sites. Thirdly, multiple species showing concordant trends, or variation, in particular habitats can indicate wider issues around the health of the ecosystem, especially for species that occupy a relatively high trophic position (Bell *et al.* 2019). For example, the analysis by Massimino *et al.* (2015) showed previously undetected strong declines in habitat specialist bird species in the south-east of England, which as yet remain unexplained.

7.1 Data Gaps

7.1.1 What is a gap? Matching data availability against the needs assessment for biodiversity data

What is a data gap? This simple question is important to consider. We suggest that gaps can be:

1. *Incomplete or biased coverage*, i.e. a paucity of records, recorders or recorded sites, for example from upland areas or north-west Scotland, or for specific taxonomic groups and at specific spatial scales. This is a conventional way to view data gaps – from the perspective of a scientific sampling design. It is relatively easy to identify ‘gaps’ defined in this way, based on observation of coverage.
2. *The inability to answer questions of interest*. This perspective is about the broader dataset and its suitability for different purposes. Recording effort could be uneven or patchy, but if it can be taken into account through appropriate analysis, then the question could still be answered. It is more challenging to generically identify ‘gaps’ from this perspective because it depends both on the particular question being asked and its context (e.g. in space or time).

In the ‘needs assessment’ related to this work (Pocock 2018), the focus from government agencies was the need to answer questions, such as on the impact of interventions or the assessment of ecosystem health and function, rather than simply having ‘complete’ spatial, habitat and taxonomic coverage *per se*. For instance, respondents to the needs assessment, did not express concern over the paucity of structured and unstructured recording in the uplands, but rather were concerned whether we could answer questions related to specific management policy (which may be possible, even with incomplete coverage), or protected sites (which would require adequate recording within and outside of protected sites to answer the question of interest). The assessment of data being ‘adequate’ was also variable. Raising data quality can also change the perceived data gaps, e.g. previously a taxon may not have sufficient data for trend estimation (a taxonomic gap), but now has sufficient data for trend estimation in some regions but not others (a regional gap). Respondents were less concerned about the ability to answer questions with a pre-determined level of statistical power but, instead, were willing to undertake risk-based assessment of the use of results derived from the available information.

In this report, we have primarily focussed on quantifying incomplete coverage (the first point) because this is tractable and independent of varying policy focus, which will cause the questions of the second point to change. It is also generalizable because having relatively good coverage is likely to permit the answering of many questions; having ‘incomplete’ coverage does not preclude that questions of interest could still be answered adequately, but it does make this less likely.

Recommendation: In future, cross referencing monitoring/recording coverage with the gaps that we identified with the needs assessment for biodiversity data in the public sector will be needed to identify clear priorities for future development in the schemes (strategic gap filling).

7.1.2 Where is coverage relatively poor?

Here we focussed on volunteer-based monitoring activity that supports the first two levels of the information hierarchy (assessment of trends in distribution, and in abundance) through unstructured and structured monitoring. Our assessment identified groups and areas where coverage is relatively poor. Taxonomically, we have a particularly poor understanding of the population status of bryophytes and charophytes, many invertebrates (with the exception of Lepidoptera) and fungi (Figure 2.1). It is important to identify critical gaps, i.e. groups with an important functional role (rather than simply groups with poorer coverage) and prioritise groups that would yield most benefit from improving coverage. For example, species of conservation and international importance e.g. Britain's high international responsibility for some lichens. Invertebrates form an important functional part of most ecosystems, so the lack of knowledge (especially for groups important for ecosystem function such as Coleoptera, Symphyta and Chironomidae) is concerning; similarly, fungi, especially of those in soils, play an important role in nutrient cycling and are also poorly covered. Another approach would be to identify species that are sensitive to environmental pressures e.g. nocturnal invertebrates that are susceptible to increased artificial light at night, or lichens and bryophytes that are indicators of changes (positive and negative) in air quality. We describe below some approaches to improving coverage, which might help address such gaps. It would be valuable to undertake an assessment of the factors that limit recording in different taxonomic groups.

Spatially there is a general trend for lower coverage towards the north and the west of Britain, with highest coverage for most taxa in the south-east of England. This broadly correlates with patterns of human population density; however, it is interesting to note that the greater London region is relatively under-represented in both structured (Figure 5.1) and unstructured monitoring (Fig 5.2). Built-up areas are generally well covered by both structured (Figure 5.4) and unstructured (Figure 5.5) efforts, so this under-representation seems surprising, yet offers opportunities in recruiting people from a (presumably) large potential pool of volunteers, who might gain well-being benefits from engaging with nature through such surveys. It would also improve our understanding of the contribution of urban areas to natural capital and the role of urbanisation in driving biodiversity change. This gradient in coverage also correlated with latitudinal gradients in species-richness (Eglington *et al.* 2015), which means that its impact on our ability to report trends across all species (in aggregate) is perhaps reduced, although, as we expected, it does mean that moor/heath/bog habitats, which tend to predominate in the north and west, were poorly covered. This includes some habitats that are scarce at a European level, for which Britain has particular responsibility, so good assessments of quality and trends are important, and improved data would improve the spatial and temporal resolution of our reporting. It also means that northern or upland-associated species tend to be less-well covered than other more widely-distributed or southerly-distributed groups This has implications for our ability to detect changes in the abundance of what are likely to be the most climate-vulnerable cold-associated species (Pearce-Higgins *et al.* 2017).

7.2 Potential solutions to improving biodiversity recording

There is a range of solutions that could be used to address the gaps in data coverage that we identified in this report. Not all solutions will be applicable in all cases, and their ease of implementation will vary between groups. These options were presented at JNCC's Terrestrial Evidence Partnership of Partnerships (TEPoP) workshop held in Birmingham in October 2018, and the text takes into account responses raised during the subsequent workshop discussions.

7.2.1 Volunteer-based recording

Most current monitoring is based around volunteer contributions. This is cost efficient and contributes extra societal benefits, e.g. potentially contributing to the health and well-being of participants (Blaney *et al.* 2016), but it requires motivated and informed participants. Typically, volunteers who participate in monitoring schemes choose to specialise in one, or occasionally a few, taxonomic groups. This generally means that the records submitted are of high quality (though still requiring verification), but means that opportunities for cross-taxa recording may be limited, especially for those groups requiring detailed identification knowledge (but see below). Volunteer-based surveys may be:

- structured - i.e. volunteers are directed to visit the same locations at repeated intervals following prescribed protocols. Most of the national biodiversity monitoring schemes fall into this category;
- semi-structured - i.e. observers visit particular locations, but not on a set schedule, and provide information on their observation effort, or on taxa not recorded;
- unstructured - i.e. records are submitted from locations selected haphazardly by the observer with little or no information on observation effort).

Clearly, structured surveys require substantial investment in volunteer coordination, and effort from the individuals, than less-structured recording (Pescott *et al.* 2019a). The types of observer contributing to each of these will differ and we recommend that more should be done to understand the motivations/contributions of these different types of volunteers (see Ganzevoort *et al.* 2017) and, perhaps, hence the extent to which they might move up the ‘survey ladder’ of skill and commitment. Semi-structured recording covers a wide range of different approaches to standardise (or record) effort (Kelling *et al.* 2019) and has the potential to act as ‘rungs’ of such a ‘ladder’, offering those who might record occasionally in an unstructured way a mechanism to deepen their engagement with, and improve their outcomes from, biodiversity recording.

In general, for a given number of sampling sites, structured recording (i.e. recording that is defined, consistent and repeated), generally gives the greatest statistical power to determine temporal or spatial trends, but it relies on sufficient volunteers with relatively high levels of commitment who are able to follow set protocols at set sites at specific times. This is possible for only a few taxonomic groups (currently in the UK these are taxa such as birds, bats, butterflies and plants). Structured recording is especially good for quantifying trends in abundance and so is likely to be better at detecting small changes or providing early detection of changes quickly; it can be less efficient, compared to unstructured recording, for quantifying trends in distribution (e.g. for providing rapid assessment of species with rapidly changing distributions). The potential of structured versus unstructured recording will vary by taxonomic group, depending on the feasibility of gathering such data, given the number of recorders available. Three inter-related questions need to be addressed in determining which approach is most beneficial, with the answers likely to be taxon-specific:

1. Benefit: How much extra benefit is gained from structured recording? Or, conversely, what are the limits to the inference that may be drawn from unstructured data? How well do these address evidence gaps that are believed to be important? How much additional benefit can be gained, compared to unstructured recording, by using semi-structured recording, and how best can this be achieved (e.g. complete lists or effort recording)?
2. Feasibility: How feasible is such structured, or semi-structured, recording given the necessity of motivating, and potentially upskilling, volunteers? For semi-structured recording, how feasible is it that volunteers can quantify effort meaningfully (either directly, e.g. by recording visit length, or indirectly, by recording all or a pre-defined list of species).
3. Joint analysis: What additional inference is added by the joint analysis of different data (structured, semi-structured and/or unstructured), for example, combining temporal resolution from a smaller number of sites with less frequently sampled sites over a large spatial area?

7.2.2 Supporting volunteer recorders

Volunteer-based recording requires significant support, to coordinate the effort, to capture the data in a form that is amenable to analysis, and to feed back the results so that volunteers remain motivated and engaged for the longer-term required for monitoring (compared to ‘citizen-science’ activities which often only require a single act of participation). Maintaining a well-motivated volunteer base (whether for structured or unstructured recording) is critical to the success of biodiversity recording. Investment in either supporting existing recorders to improve their identification /sampling skills, or training new recorders has the potential to increase the level of recording and some of this activity might be spatially targeted (as has been done through the National Plant Monitoring Scheme and the Breeding Bird Survey), although for some areas the number of inhabitants and visitors is always likely to limit recording. There are a number of ways in which existing levels of volunteer support can be improved and/or expanded:

Provision of mentoring, training courses and materials
These can be effective and can be targeted in particular areas. While face-to-face workshops appear to be effective in training, motivating and retaining volunteers (but systematic evaluation is lacking), they are expensive to run and reach a limited number of people. Alternatively, or in addition, online materials can be created, curated and disseminated, often more cheaply than traditional printed material, and should not be ignored. Such approaches may work best at improving the skills of those already submitting data, though facilitating entry routes into more systematic surveying through sites like iRecord or BirdTrack could also be explored. Establishing more informal peer mentoring networks could also be considered, for example this has been successful in increasing participation in the BTO/JNCC Nest Record Scheme, by providing entrants with the confidence that their skills are developing in the right direction.

Community-supported identification / social networks
Many volunteer surveyors enjoy community focussed activities around their recording, such as comparing records or helping others with identification. The BTO’s network of regional organisers and ambassadors provides one model for this, whereby survey effort is stimulated and co-ordinated locally and, in the best cases, support and encourage the development of new recorders. In addition, a number of online tools now exist to facilitate such networking with varying degrees of formality and engagement (iSpot is one cross-taxon project in the UK (Silvertown *et al.* 2015)). Schemes should be supported in developing, or working with, such initiatives, which could help to build communities of recorders. While social media (and other) sites are valuable in fostering networks of recorders, when records are submitted only through them, it is often time-consuming to convert identifications to a usable biological record, with good quality information on location, recorder and date, as well as species identity; promotion of appropriate recording practices will be key to their success in building volunteer capacity. It should be noted, though, that older surveyors (usually those with most experience and knowledge) may be less engaged with online technologies and efforts need to be made to ensure they are not ‘left behind’.

Provision of identification resources (including field guides)
An ability to identify species, particularly in the more difficult groups, but even of those which present fewer difficulties, can be a significant barrier to participation, especially for those new to recording. Arguably, one reason why birds, butterflies and flowering plants have a long history of detailed records is because of the ready availability of user-friendly identification guides, pitched at an appropriate level. Support for the creation of such identification resources has the potential to help improve taxonomic coverage, especially when the resources are promoted to new audiences. Online guides, such as those provided by many of the National Recording Schemes and Societies, and other local natural history societies, provide valuable resources to help with species identification. Websites and apps are

particularly suited to the use of multi-access identification keys (whereby recorders use several features simultaneously to identify a specimen), sometimes including information on location and date to refine likely identifications. Despite the potential of these (Burkmar 2013), they have not become widespread. However, in some recording apps (e.g. iRecord Butterflies), species are presented in order of their frequency of records at that time of year in that region. It would be valuable to explore barriers to the use and application of new approaches to identification resources.

Interactive app/mobile technology
Mobile technology is pervasive these days and is being used by field recorders both to help identify species, either passively through reference material loaded on the device or more actively by taking an image/sound/video which is then classified, and to collect data. Particularly where the main method of data collection is unstructured (or perhaps semi-structured), there is the opportunity for mobile devices to prompt recording behaviour based on particular requirements (e.g. location, date and/or past pattern of submitted records). Thus, the device 'nudges' the recorder's behaviour to improve the value of the data collected. Novel analytical tools would need to be developed to draw robust inferences from such 'prompted' data, though, since detectability will be a function of both recorder's and app's behaviour.

Automation (image/sound recognition)
Automated (passive) recording, either acoustic or otherwise, has significant potential to increase the records obtained by assisting identification. Broadly there are two approaches. Firstly, the use of automation (usually image recognition) to aid users in correctly identifying species, to a certain taxonomic level, for example the iNaturalist app and the Merlin Bird ID app (in the USA) use image recognition for this purpose. Secondly, by the use of automatic algorithms to provide probabilistic identifications, typically from passive detectors; in the UK these are currently most commonly of sound files (e.g. Newson *et al.* 2017a), but camera traps are widely used to record, especially nocturnal, mammals (e.g. www.mammalweb.org). Automated recognition can efficiently undertake call detection and identification from large amounts of data and automated image and/or sound recognition algorithms could help in the identification of some species groups, most likely by recorders uploading an image/sound file to a website with an automated classifier. However, availability of such a mechanism is likely to be limited without significant investment, both in the data capture and processing pipelines and in the developing the automated classifiers, and the long-term retention of volunteers is unknown.

7.2.3 Augmenting volunteer recording with professional recording

We traditionally think of biological recording as the purview of volunteer recorders, indeed at the wide scale required for national monitoring, the resources provided by volunteers makes this the only feasible approach in many cases. However, consideration could be given to supplementing this with targeted (possibly multi-taxa) surveying by paid fieldworkers in areas where coverage by volunteers is less feasible. While resources were available, this model has worked well for BBS where, for example, contracted workers supplemented survey coverage in upland areas, and is employed in other countries (e.g. Scandinavian countries to increase coverage of remote, usually more northerly, sites). We suggest that this is most justifiable where there is a clearly stated question, which then defines the need that supports the additional investment.

7.2.4 Technological solutions

The extent to which technology can aid biodiversity and environmental monitoring is increasing rapidly as ground-breaking and transformative technological advances for studying species and environments are devised and implemented (Allan *et al.* 2018). While most biodiversity recording has been based on volunteers making records through their own identifications or sampling, technological developments offer the potential to fill some gaps, either taxonomic or spatial, potentially opening up the concept of monitoring and citizen science to people who have the interest but not necessarily the identification skills to be participants in the traditional sense.

Environmental DNA sampling
Extraction and identification of DNA from environmental samples (eDNA), combined with advances in the analysis of genetic sample (meta-barcoding) is increasingly able to detect and monitor not only common species, but also those that are endangered, invasive, or elusive (Bohmann *et al.* 2014; Thomsen & Willerslev 2015). eDNA has been widely used to determine presence of particular taxa, especially in freshwater environments (e.g. Rees *et al.* 2014), though the range of situations in which it can be deployed continues to expand, for example invertebrates visiting flower heads (Thomsen & Sigsgaard 2019). Collecting eDNA samples can be cheaper than traditional methods of sampling (although analysis can be expensive), but this is not always the case, and information on the nature of individuals present can be lost (Evans *et al.* 2017).

Earth observation
Remote sensing for Earth observation undertaking with satellites or unmanned aerial vehicles (UAVs, commonly called drones) is a maturing technology (Pettorelli *et al.* 2014). Satellite data are becoming available more easily than ever and in ever-greater quantities, for example, through the Sentinel-2 system that provides 10–60m resolution multispectral images every five days. The potential for Earth observation to represent land use change and habitat condition has been and continues to be explored by JNCC (e.g. Medcalf *et al.* 2014b; Medcalf *et al.* 2014a). Earth observation could be particularly valuable for:

- mapping features over large scales, especially in hard-to-access locations, e.g. habitat extent in the uplands or tree cover;
- assessing habitat change in a resource-efficient way, by using an analysis pipeline to update maps and assessments as new data become available, for example, arable crops (CEH's Land Cover Plus: Crops product) or tree growth in plantations, although some features of particular interest, e.g. reedbed habitat, can be difficult to distinguish;
- recording features that are difficult for volunteers to record consistently, e.g. certain measures of quality of habitats, although not all relevant measures of habitat quality that are relevant to biodiversity, and their use of particular habitats, will be amenable to being remotely sensed.

With regards to species monitoring, there is the potential for two-way engagement between volunteer recorders and Earth observation:

- volunteers undertaking recording could provide ground-truthed data for the analysis of Earth observation data. The potential for this should not be over-stated due to potential limitations, including: spatial mismatching of volunteers/surveyed sites and places where ground-truthing is required; knowledge of critical habitat differences and willingness of volunteers to record habitat features. However, habitat data collected by the NPMS have been used in training an Earth observation model by Natural England to produce their 'Living England' habitat map;
- Earth observation data provide information that could be used to direct, or nudge, volunteers to record in particular locations;
- Earth observation data provide valuable covariates for analysis of species datasets, whether from structured or unstructured data.

Passive recording
There is an increasing availability of sensors that could be available to support biodiversity monitoring. There is particular interest in passive acoustic monitoring (Gibb *et al.* 2019), which has a long use for monitoring bats, but more recently there has been interest in birds (including nocturnal migration), cicadas, Orthoptera (Newson *et al.* 2017a), pollinating insects (Miller-Struttmann *et al.* 2017) and soundscapes (Fairbrass *et al.* 2019; Ross *et al.* 2018). The potential for passive recording has increased dramatically with the availability and accessibility of low-cost sensors and processing boards, supported by a community of people interested in DIY approaches to sensors. Crucial to the success of passive recording, which can generate large quantities of data, is the development and application of accurate automated classification to enable useful information to be extracted efficiently.

The potential for this expands to other environmental sensors, which have been used for a long while in environmental recording (e.g. pollutants affecting air and water quality, water height, light levels and so on), but not necessarily for biodiversity. This may be less motivating for many biological recorders but may open new opportunities to different volunteers willing to deploy sensors, although their motivation and long-term retention has not been assessed. With the development of the ‘Internet of Things’ there will be more potential for networks of sensors to become autonomous, and so not require interventions to collect data. An alternative version of this would be to use volunteers to deploy low cost environmental sensors to provide overlapping spatial coverage, improving, presumably, our ability to detect environmental linkages and relationships. This would not necessarily fill biodiversity recording gaps but could be used to address other issues (e.g. ecosystem services, water quality) and, potentially, build links with the wider sector (e.g. Environment Agency).

7.2.5 Location-based solutions

The Environmental Change Network (ECN) family operates a number of sites where co-ordinated recording of various biodiversity and environmental variables occurs (depending on the particular scheme). There may be opportunities to establish simpler levels of co-located cross-taxa recording. This could involve the same individuals recording different taxa (e.g. ringers recording numbers of invertebrates caught on sticky traps during a session) or by different recorders participating in different schemes recording at the same sites, as with current developments of the UK Pollinator Monitoring Scheme co-locating sampling sites with the National Plant Monitoring Scheme, or linking potential WCBS surveyors with BBS squares that lack butterfly surveys. This has number of benefits in terms of improving resolution of ecosystem function recording as different components from the ecosystem are then recorded from the same spatial location. However, sites suitable for recording one taxon may be less suitable, or less attractive, for recording another, and the regional density of recorders may differ. A further challenge is that the appropriate or practical spatial scale over which sampling should or can be conducted varies across taxa: while it is feasible to survey birds across a 1km square it would be extremely difficult/time-consuming to obtain data on carabid beetle density or diversity that was representative of the same square.

An alternative approach is to consider promoting expeditions to particular areas, whereby a group of recorders visits a particular area with poor coverage and aims to target records of multiple groups. This might work especially well where taxa can be recorded together using similar methods, e.g. butterflies (a very well-recorded group) and soldier beetles or plant bugs (less well-recorded groups). Our analysis indicated that it is the best-recorded years that have most influence on the precision of occupancy trend outputs (albeit that the best-recorded years are consistently related to the typical years). It could be valuable, especially for less-well recorded groups, to have ‘campaigns’ every 5–10 years within projects. At a minimum, if sites that have been visited once are subsequently revisited, then both the current and the prior records become available for analysis. It would be valuable to trial methods that identify and publicise sites that have only been visited in one year over the appropriate period, so that a recording scheme can prioritise these for revisits.

7.2.6 Analytical solutions

While gaps in monitoring can clearly be filled by improving the range and quality of data gathering as discussed above, some analytic solutions may also exist that make better use of data that are already collected. Primarily, this is likely to come from ongoing developments in statistical methods to integrate datasets of different kinds. These include:

- *Combining different types of census data*: combining datasets that describe the same pattern, for example combining samples from structured (usually smaller number of records but more information-rich data) and unstructured (usually larger number of records but less-information rich, e.g. limited information on effort) surveys to better characterise patterns of abundance or change, or models based on process.
- *Combining census data with demographic data*: process-based models might focus on the mechanism of change in particular species, integrating information from different sources on demographic mechanisms (e.g. integrated population models, Robinson *et al.* 2014), or models of ecosystem function (e.g. Bayesian belief networks, Landuyt *et al.* 2013).
- *Combining census data and environmental data*: combining taxa records with independent supplementary data on habitat or other drivers of change is a powerful way to identify possible drivers of change.

7.3 Conclusions and recommendations

There is substantial spatial and taxonomic variation in our ability to monitor biodiversity. Our previous Needs Assessment (Pocock 2018) showed that our monitoring largely meets current requirements, however, further enhancement in recording should allow us to address more needs and do so more accurately, precisely, rapidly and with greater taxonomic breadth.

Gaps in biodiversity data should be defined by our ability to adequately address questions of concern, and not by uneven coverage *per se*. However, gaps in spatial/taxonomic coverage are likely to make it harder to address questions of interest, and so coverage (as we have considered in this report) provides a proxy for our ability to address questions of policy relevance.

While there continue to be opportunities to engage more people, collate more records and to improve spatial and taxonomic coverage, achieving substantial step changes in our ability to gather biodiversity data will be challenging. Our current focus, therefore, should be on making best use of what we have, in terms of volunteer capacity and data, to address gaps in taxonomic and spatial data coverage.

Unstructured recording

- We found that the minimum number of records to obtain estimates of occupancy that meet the precision threshold was surprisingly low. While further work is required to consider the accuracy and representativeness of such outputs, this is positive because it has the potential to support the need of government agencies for more ‘taxonomically comprehensive measures of ecosystem health’ (Pocock 2018).
- However, analytical outputs that meet the precision threshold do not necessarily guarantee that the output is accurate or representative. Further work is required to assess how accuracy is affecting by recording effort and uneven coverage and on how best to assess representativeness of the trend.

- More records are better in increasing precision, and potentially better spatial representation and reduced spatial bias, so even small increases in recording effort will be beneficial in improving precision of occupancy estimates. In particular, sites with records in only a single year are excluded from our current analysis, so recording at these sites means that the current and the historic record are added to the analysable dataset.

Structured recording

- While structured schemes are successful at monitoring some key groups (notably birds, butterflies, some mammals) at a national level, regional gaps do exist, particularly in the north and west of the country, and in some urban areas. Efforts to fill the latter should align with wider engagement objectives.
- We noted some evidence that the pool of effort was not infinite (e.g. dips in BBS recording during the atlas period and apparent recent declines in bat recording). Maintaining a motivated network requires substantial co-ordination and effort.
- There are opportunities for co-location for recording of at least some taxa, however, issues regarding suitability of sites for different taxa, and the appetite of volunteers to either record different taxa, or visit new, pre-selected places will need to be addressed.

The volunteer resource

- If we are to increase in number of recorders (bearing in mind that there are limits to the amount of effort volunteers are willing to undertake), we need to ensure that the value we can gain from people's time and effort is maximised, which will require a fuller understanding of their motivation.
- We recommend that there is further consideration of how we can encourage those submitting unstructured records to undertake semi-structured recording (e.g. recording effort), and how feasible this is for different taxonomic groups. Further work is required to assess the value of this in joint analyses of different data types.
- We recommend that further consideration should be made of how people can be incentivised to increase spatial coverage for the taxonomic group they are recording, especially for structured recording schemes. Ideally, this work should be done in collaboration with the volunteers to ensure that proposals are co-created rather than appear to be imposed from analysts.

The data resource

- Bearing in mind limitations in data currently available it is important that we make maximum use of the data that are available.
- We recommend further investment in developing joint analyses of different biodiversity data (e.g. combining structured, semi-structured and/or unstructured data). It will be valuable to assess the trade-off between analytical complexity and gain in benefit from joint analyses.
- We recommend further investigation in combining biodiversity data with other environmental data. This could assist spatial extrapolation to unsampled areas and would help assess the impact of environmental drivers and management/policy interventions.

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10 Appendix

10.1 Tables and Figures

Table 2.1. Population measures and subcategories relating to the extent and quality of existing biological recording or monitoring data.

Attribute	Subcategories relating to the quality of data	Quality score
A. Population size	Ad-hoc counts (sparse data) or estimates difficult to produce	1
	Ad-hoc counts (extensive data)	2
	Local site-level estimates only	3
	Population estimates (national scale resolution) from structured data	4
	Population estimates (regional scale resolution) from structured data	5
	Complete census	6
B. Trends in abundance	Not possible to monitor change in abundance	1
	Trends based on ad-hoc counts	2
	Trends from structured data for subset of sites	3
	Periodically updated trends from structured data spanning the range	4
	Annually updated trends from structured data spanning the range	5
	Complete census each year (real population change)	6
C. Distribution	Occurrence records are sparse in relation to expected distribution, and not suitable for meaningful distribution mapping	1
	Occurrence records are representative in relation to expected distribution, allowing coarse resolution mapping (e.g. 10km resolution)	2
	Distribution recorded (presence/ non-detection) at a sufficient sample of sites, allowing fine-scale modelling of distribution	3
	Occurrence records are abundant within the expected distribution allowing fine resolution mapping (e.g. 1 or 2 km resolution)	4
	Complete census (true distribution fully known)	5
D. Trends in distribution	Distribution change cannot be inferred with confidence	1
	Coarse scale distribution change statistics can be meaningfully produced from time slices of biological records (e.g. comparison of atlases)	2
	Change can be estimated to produce meaningful national-scale trends in distribution	3
	Production of meaningful trends in distribution that are fine-scale, both spatially and temporally	4
E. Demographic rates	Demographic rates not available	1
	Ad-hoc estimates	2
	Local site-level estimates only	3
	Demographic rates from structured data	4
F. Trends in demographic rates	Not possible to monitor change in demographic rates	1
	Trends based on ad-hoc estimates	2
	Trends based on local site-level estimates only	3
	Trends in demographic rates from structured data	4

Table 2.2. Assessment of our current ability to estimate or monitor six population measures that are most likely to be of interest to stakeholders according to species group / species subgroup. The scores here are the product of a quality score that relates to the extent and quality of existing biological recording or monitoring data that are available for each species group, and a taxonomic score, that relates to the breadth of species coverage. In the penultimate column we present a single score calculated as a proportion of the maximum score across population measures. Green shaded rows show the 'headline' row for each main species group, with subgroups below. Where it was believed that there were likely to be clear differences in scoring within a species group, species groups were split into sub-groups to reflect this. In some cases, scores are assessed separately for sub-groups of species with particular policy or reporting needs. This is described in more detail in section 2.1.

Taxonomic Level		Population attribute (see Table 2.1)							
Level 1	Level 2	A	B	C	D	E	F	Sum as % of maximum score	Main person responsible for scoring
AMPHIBIANS & REPTILES	Amphibia and Reptilia: ALL SPECIES	27	39	29	20	11	12	41	Stuart Newson, Andy Musgrove
AMPHIBIANS & REPTILES	Amphibia and Reptilia: Common and widespread species	28	40	16	23	11	12	45	Stuart Newson, Andy Musgrove
AMPHIBIANS & REPTILES	Amphibia and Reptilia: Rare or localised species excluding Natterjack Toad	31	28	32	25	26	25	58	Stuart Newson, Andy Musgrove
AMPHIBIANS & REPTILES	Amphibia: natterjack toad	30	30	34	25	28	20	58	Stuart Newson, Andy Musgrove
BIRDS	Aves: ALL SPECIES	33	42	31	27	26	25	63	Stuart Newson, Andy Musgrove
BIRDS	Aves: Breeding waterbirds and riverine species	35	35	27	26	29	29	62	Stuart Newson, Andy Musgrove
BIRDS	Aves: Common and widespread species	44	50	40	26	26	25	73	Stuart Newson, Andy Musgrove
BIRDS	Aves: Gamebirds	35	37	26	22	29	29	61	Stuart Newson, Andy Musgrove
BIRDS	Aves: Localised breeding birds	26	38	21	25	29	25	57	Stuart Newson, Andy Musgrove
BIRDS	Aves: Nocturnal breeding birds (owls, nightjar, crakes)	29	39	25	24	29	29	60	Stuart Newson, Andy Musgrove
BIRDS	Aves: Rare breeding birds	33	35	40	29	18	18	60	Stuart Newson, Andy Musgrove
BIRDS	Aves: Wintering wetland birds	41	48	27	24	18	18	61	Stuart Newson, Andy Musgrove
FISHES	Gnathostomata and Agnatha: ALL SPECIES	31	22	21	17	30	30	52	Stuart Newson, Andy Musgrove
FLOWERING PLANTS / FERNS	Tracheophytes: ALL SPECIES	12	48	35	23	N/A	N/A	56	Oli Pescott
FLOWERING PLANTS / FERNS	Tracheophytes: Common and widespread species	10	48	36	27	N/A	N/A	58	Oli Pescott, Charlie Outhwaite
FLOWERING PLANTS / FERNS	Tracheophytes: Rare or localised species	12	19	35	22	N/A	N/A	42	Oli Pescott
FUNGI (NON LICHENISED)	Eurotiomycetes: ALL SPECIES	12	10	14	11	N/A	N/A	22	Oli Pescott
INSECTS	Acari: Sarcoptiformes, Trombidiformes and Mesostigmata / Mites	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Acarina: Ixodoidea / Ticks	10	10	10	10	10	10	21	Martin Harvey
INSECTS	ALL SPECIES	14	19	11	11	10	10	26	Martin Harvey
INSECTS	Annelida: Oligochaeta / Earthworms	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Arachnida: Araneae / Spiders	15	11	16	24	10	10	30	Martin Harvey, Charlie Outhwaite
INSECTS	Arachnida: Opiliones / Harvestmen	15	11	14	10	10	10	24	Martin Harvey
INSECTS	Arachnida: Pseudoscorpiones / Pseudoscorpions	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Cladocera / Water fleas	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: (aquatic species) / Aquatic beetles	10	10	15	21	10	10	26	Martin Harvey
INSECTS	Coleoptera: Buprestidae, Cantharidae, Drilidae, Lampyridae and Lycidae / Soldier and jewel beetles, glow-worm and allies	10	10	13	23	10	10	26	Martin Harvey, Charlie Outhwaite

Taxonomic Level		Population attribute (see Table 2.1)							
Level 1	Level 2	A	B	C	D	E	F	Sum as % of maximum score	Main person responsible for scoring
INSECTS	Coleoptera: Carabidae / Ground beetles	10	10	15	24	10	10	27	Martin Harvey, Charlie Outhwaite
INSECTS	Coleoptera: Cerambycidae / Longhorn beetles	10	10	15	12	10	10	23	Martin Harvey
INSECTS	Coleoptera: Chrysomelidae & Bruchidae / Leaf-and seed-beetles	10	10	12	23	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Coleoptera: Coccinellidae / Ladybirds	18	10	18	26	10	10	32	Martin Harvey, Charlie Outhwaite
INSECTS	Coleoptera: Cryptophagidae, Atomariinae / Atomariine beetles	10	10	10	10	10	10	21	* inactive – no scheme
INSECTS	Coleoptera: Curculionoidea / Weevils and Bark Beetles	10	10	13	23	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Coleoptera: Dermestidae (and Derodontidae) / Hide, larder and carpet beetles	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: Elateroidea / Click beetles and allies	10	10	10	10	10	10	21	* data not shared
INSECTS	Coleoptera: Histeridae and Sphaeritidae / Clown Beetles	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: Ptiliidae / Ptiliid beetles	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: Scarabaeoidea / Dung beetles and chafers	10	10	16	19	10	10	26	Martin Harvey
INSECTS	Coleoptera: Scirtidae / Scirtid beetles	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: Silphidae / Carrion, burying and sexton beetles & relatives	10	10	14	12	10	10	23	Martin Harvey
INSECTS	Coleoptera: Staphylinidae / Rove beetles	10	10	10	16	10	10	23	Martin Harvey, Charlie Outhwaite
INSECTS	Coleoptera: Stenini / Staphylinid beetles: stenus and dianous	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Coleoptera: Tenebrionoidea / Darkling beetles	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Collembola / Springtails	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Crustacea (hypogean species) / Cave amphipods and other crustacea	11	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Agromyzidae / Leaf-miner flies	10	10	12	10	10	10	21	Martin Harvey
INSECTS	Diptera: Anthomyiidae / Anthomyiid flies	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Diptera: Calliphoridae / Blow flies	10	10	12	12	10	10	22	Martin Harvey
INSECTS	Diptera: Chironomidae / Chironomid flies	10	10	10	10	10	10	21	* inactive – no scheme
INSECTS	Diptera: Chloropidae / Chloropid flies	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Conopidae, Lonchopteridae, Ulidiidae & Pallopteridae / Conopid, lonchopterid & picture-winged flies	10	10	14	10	10	10	22	Martin Harvey
INSECTS	Diptera: Culicidae / Mosquitoes	10	10	10	11	10	10	21	Martin Harvey
INSECTS	Diptera: Culicoides (Ceratopogonidae) / Biting midges	10	10	10	10	10	10	21	* inactive – no scheme
INSECTS	Diptera: Dixidae / Meniscus midges	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Drosophilidae / Fruit flies	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Empididae, Hybotidae, Dolichopodidae / Empid, Hybotid & Dolichopodid flies	10	10	14	19	10	10	25	Martin Harvey, Charlie Outhwaite
INSECTS	Diptera: Mycetophilidae and allies / Fungus gnats	10	10	12	19	10	10	24	Martin Harvey, Charlie Outhwaite
INSECTS	Diptera: Nerioidea: Pseudopomyzidae, Micropezidae; Diopsoidea: Tanypezidae, Strongylophthalmidae, Megamerinidae & Psilidae / Stilt and stalk flies	10	10	11	10	10	10	21	Martin Harvey
INSECTS	Diptera: Oestridae / Warble-flies and bots	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Pipunculidae / Pipunculid flies	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Diptera: Platypezidae / Flat-footed flies	10	10	10	10	10	10	21	Martin Harvey

Taxonomic Level		Population attribute (see Table 2.1)							
Level 1	Level 2	A	B	C	D	E	F	Sum as % of maximum score	Main person responsible for scoring
INSECTS	Diptera: Sciomyzidae / Snail-killing flies	10	10	12	10	10	10	21	Martin Harvey
INSECTS	Diptera: Sepsidae / Sepsid flies	10	10	12	10	10	10	21	Martin Harvey
INSECTS	Diptera: Soldierflies and allies (Lower Brachycera) / Soldierflies, horseflies, robberflies, snipeflies, stiletto-flies, bee-flies and allies	10	10	16	22	10	10	27	Martin Harvey, Charlie Outhwaite
INSECTS	Diptera: Syrphidae / Hoverflies	10	10	18	24	10	10	28	Martin Harvey, Charlie Outhwaite
INSECTS	Diptera: Tachinidae / Tachinid flies	10	10	13	11	10	10	22	Martin Harvey
INSECTS	Diptera: Tephritidae / Tephritid flies	10	10	14	14	10	10	23	Martin Harvey
INSECTS	Diptera: Tipuloidea & Ptychopteridae / Craneflies	10	10	15	21	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Ephemeroptera / Mayflies	10	10	12	23	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Hemiptera: Auchenorrhyncha / Leafhoppers & froghoppers	10	10	12	11	10	10	22	Martin Harvey
INSECTS	Hemiptera: Heteroptera (aquatic species) / Water bugs	10	10	14	23	10	10	27	Martin Harvey, Charlie Outhwaite
INSECTS	Hemiptera: Heteroptera (terrestrial species) / Plant bugs & allied species	10	10	13	22	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Hemiptera: Heteroptera (terrestrial species) / Shield bugs & allied species	10	10	17	23	10	10	28	Martin Harvey, Charlie Outhwaite
INSECTS	Hemiptera: Sternorrhyncha: Psylloidea / Psyllids	10	10	10	10	10	10	21	* inactive – no scheme
INSECTS	Hymenoptera: Aculeata / Bees, Wasps, Ants	19	13	23	26	10	10	35	Martin Harvey, Marc Botham, Charlie Outhwaite
INSECTS	Hymenoptera: Ichneumonoidea (nocturnal) / Parasitic wasps	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Hymenoptera: Symphyta / Sawflies	10	10	10	10	10	10	21	* inactive – no scheme
INSECTS	Isopoda (non-marine species) / Woodlice	10	10	14	15	10	10	24	Martin Harvey
INSECTS	Lepidoptera: Butterflies - All species	35	50	30	26	10	10	56	Martin Harvey & Marc Botham
INSECTS	Lepidoptera: Butterflies - Common and widespread species	35	50	28	30	10	10	56	Martin Harvey & Marc Botham
INSECTS	Lepidoptera: Butterflies - Localised and rare species	44	48	32	27	10	10	59	Martin Harvey & Marc Botham
INSECTS	Lepidoptera: Crambidae & Pyralidae / Grass & Pyralid moths	10	10	14	12	10	10	23	Martin Harvey & Marc Botham
INSECTS	Lepidoptera: Gelechiidae / Gelechiid moths	10	10	12	23	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Lepidoptera: Incurvarioidea / Longhorn moths and allies	10	10	12	10	10	10	21	* inactive – no scheme
INSECTS	Lepidoptera: Leaf-miners / Leaf-mining moths	10	10	12	10	10	10	21	* data not shared
INSECTS	Lepidoptera: Macro-moths	34	50	18	21	10	10	49	Marc Botham
INSECTS	Lepidoptera: Micro-moths	24	12	14	11	10	10	28	Marc Botham
INSECTS	Lepidoptera: Moths - All species	33	50	18	26	10	10	51	Martin Harvey, Charlie Outhwaite
INSECTS	Lepidoptera: Pterophoridae / Plume moths	10	10	16	11	10	10	23	Martin Harvey & Marc Botham
INSECTS	Mollusca (non-marine species) / Non-marine molluscs	10	10	15	19	10	10	26	Martin Harvey, Charlie Outhwaite
INSECTS	Myriapoda: Chilopoda / Centipedes	10	10	13	20	10	10	25	Martin Harvey, Charlie Outhwaite
INSECTS	Myriapoda: Diplopoda / Millipedes	10	10	13	20	10	10	25	Martin Harvey, Charlie Outhwaite
INSECTS	Neuropterida (Neuroptera, Mecoptera & Megaloptera) / Lacewings, scorpion-flies, snake-flies and allies	10	10	12	11	10	10	22	* inactive – no scheme
INSECTS	Odonata / Dragonflies & damselflies	22	10	20	26	10	10	34	Martin Harvey, Charlie Outhwaite
INSECTS	Orthoptera, Dermaptera, Dictyoptera & Phasmida / Grasshoppers and allies	17	11	21	26	10	10	33	Bjorn Beckmann
INSECTS	Plecoptera / Stoneflies	10	10	12	24	10	10	26	Martin Harvey, Charlie Outhwaite

Taxonomic Level		Population attribute (see Table 2.1)							
Level 1	Level 2	A	B	C	D	E	F	Sum as % of maximum score	Main person responsible for scoring
INSECTS	Psocoptera (outdoor species) / Barkflies	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Siphonaptera / Fleas	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Trichoptera / Caddisflies	10	10	18	24	10	10	28	Martin Harvey, Charlie Outhwaite
INSECTS	Tricladida (freshwater species) / Freshwater flatworms	10	10	10	10	10	10	21	Martin Harvey
INSECTS	Tricladida (terrestrial species) / Terrestrial flatworms	10	10	10	10	10	10	21	Martin Harvey
LICHENS	ALL SPECIES	12	10	22	19	N/A	N/A	30	Oli Pescott, Charlie Outhwaite
MAMMALS	Mammalia: ALL SPECIES	24	12	24	18	21	21	41	Colin Harrower
MAMMALS	Chiroptera: Bats (all species)	24	38	22	22	19	19	50	Colin Harrower
MAMMALS	Chiroptera: Bats (common and widespread species during breeding season)	18	50	24	25	19	13	51	Colin Harrower
MAMMALS	Chiroptera: Bats (rare or localised species)	20	18	24	17	16	16	38	Colin Harrower
MAMMALS	Chiroptera: Bats (wintering numbers)	10	30	25	25	12	12	39	Colin Harrower
MAMMALS	Specific species: Lagomorphs, all deer, squirrels, red fox, mustelids (excluding otter).	18	26	28	20	21	21	46	Colin Harrower
MAMMALS	Specific species: Other rodents, mole and hedgehog	17	15	19	16	15	15	33	Colin Harrower
MAMMALS	Specific species: otter, hazel and edible dormouse, water vole	33	35	26	20	21	21	54	Colin Harrower
MOSSES & LIVERWORTS	Bryophyta and Musci: ALL SPECIES	12	10	31	23	N/A	N/A	36	Oli Pescott, Charlie Outhwaite
SLIME MOULDS	Myxomycetes: ALL SPECIES	10	10	10	10	N/A	N/A	19	Oli Pescott
STONEWORTS	Charophyceae: ALL SPECIES	10	10	33	23	N/A	N/A	36	Oli Pescott

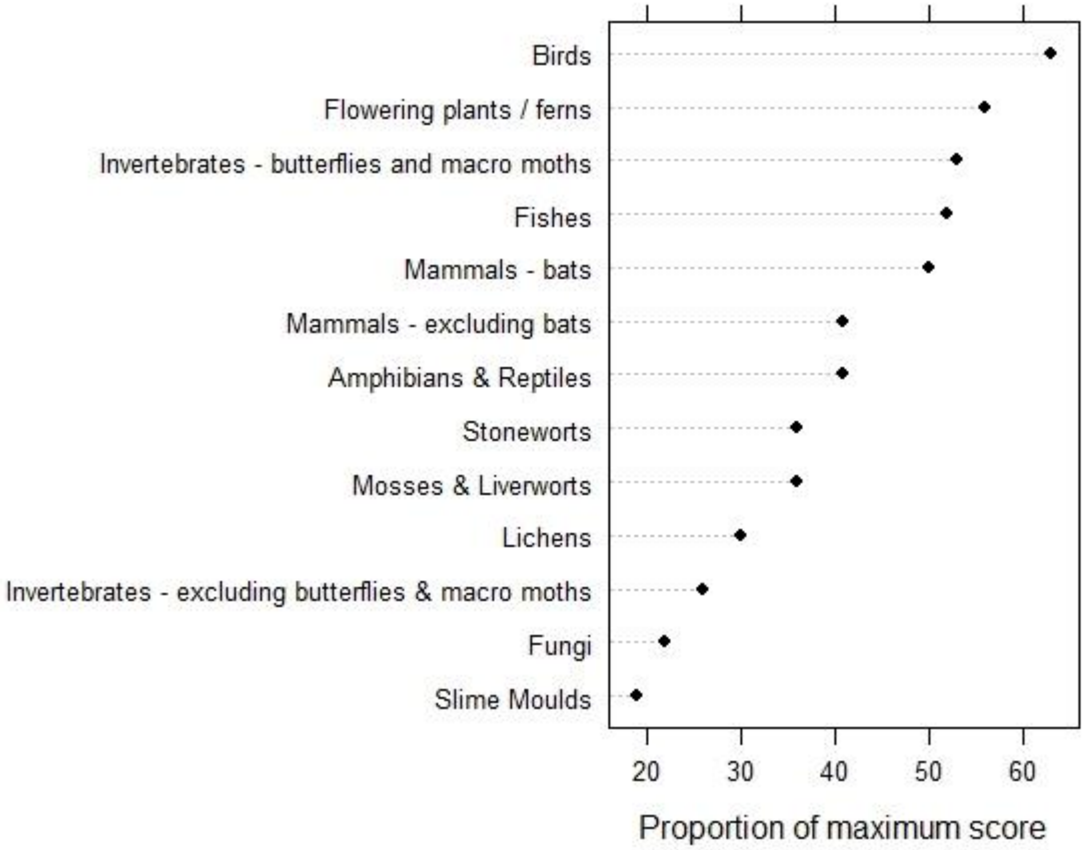

Figure 2.1. Ability to provide a precise and accurate assessment of the status and change in status of different species groups (all species) presented as the sum of score across six population measures, expressed as a proportion (%) of the maximum score.

Table 3.1. Summary of the causes for presence recording and non-detections in occurrence data.

Species recorded	**Species seen**	**Species present**	**Explanation**
Yes	Yes	Yes	Species present and recorded.
No	Yes	Yes	Bias in recording, e.g. due to the species being ubiquitous (and hence ignored) or unidentified. This is one form of false negative. This can be reduced by encouragement to record ‘complete lists’ of sightings (Szabo *et al.* 2010; Sullivan *et al.* 2014)
No	No	Yes	Imperfect detection. This can be species-specific, due to it being cryptic or rare. Another form of false negative.
No	No	No	Species absent and not recorded.
Yes	(Yes)	No	False positive due to mis-identification. For BRC data this is assumed to be negligible due to the careful verification.

Table 4.1. Descriptions of metrics that describe variation in presence/non-detection datasets across projects.

Metric	Level	Temporal aggregation	Definition	Interpretation
Median yearly number of records	Records of focal species	Annual	The median of the number of records of the focal species per year. Log-transformed.	This is the simplest measure of the amount of information on 'presences'. This is the overall level of recording for a typical year in the dataset
90th percentile yearly number of records	Records of focal species	Annual	As above, but the 90th percentile instead of the median.	As above, but for the best-recorded years in the dataset
Median yearly number of visits	Visits to sites known to be occupied (by focal species)	Annual	The median of the number of visits to a site each year, for sites where the species has been observed *in that year* (i.e. including visits where the focal species was not recorded), averaged (mean) across years. Log-transformed.	How well-recorded are the average sites where the species occurs? This helps us accurately assess how well the model is going to be able to estimate detectability. (Note that the metric is not the number of visits in which the species is recorded, it is the number of all visits to sites where the species had been recorded at least once.)
90th percentile yearly number of visits	Visits to sites known to be occupied (by focal species)	Annual	As above, but the 90th percentile instead of the median.	As above, but for the best-recorded years in the dataset
Proportion of repeat visits for the project	Site:year combinations	Total	The proportion of all site:year combinations, where the focal species is observed, that have > 1 visit.	This gives information on our ability to estimate detectability. It also captures information on overall sampling effort.
Proportion of successful visits with list length one	Visits when focal species was recorded	Total	Considering all the visits where the focal species was recorded, the proportion that had list length of 1 (i.e. records only of the focal species)	This is a proxy of quality of information on absences. It assumes that if the focal species is on lists of length one, then many other species will be too, and information on inferred absences is weak. This metric competes with the taxonomic group metrics
Proportion of visits with non-detection of focal species	Visits for the group	Total	Considering all visits for the group in the dataset, the proportion of all visits that did not record the focal species.	This gives the proportion of inferred absences. This is important because a rarely-recorded species will have occurrence estimated fairly well because of all the inferred 'absence' records. We expect that it will be important when considered in combination with the number of visits per year

Table 4.2. Number of species according to the individual projects. Taxonomic groups with the same superscript had data collated by the same recording scheme but are taxonomically sufficiently different from each other that they were treated as separate projects in this analysis. Full details of the recording schemes are at https://www.brc.ac.uk/recording-schemes.

Taxon	Number of species used in the decision tree analysis	% of species records removed (incidental records: due to sites being visited on a single year only)	Number of species used in the regional analysis (including those with 'no data')
Bryophytes	994	46.5	1002
Lichens	0*	n.a.	2143
Molluscs	255	47.8	266
Hypogean Crustaceans	9	7.2	0*
Ephemeroptera[1]	45	14.6	45
Plecoptera[1]	31	14.3	32
Trichoptera[1]	184	15.7	186
Odonata[1]	54	8.6	54
Orthoptera	27	33.4	28
ShieldBugs (various Heteroptera)	64	37.7	66
AquaticBugs (various Heteroptera)	88	42.3	92
PlantBugs (various Heteroptera)	403	31.1	413
FungusGnats (Diptera: Sciaroidea)	501	47.4	521
Craneflies (Diptera: Tipulidae)	332	46.4	344
Empidae and Dolichopodidae (Diptera)	634	36.3	650
Hoverflies (Diptera: Syrphidae)	270	18.8	272
Soldierflies and allies (Diptera)	147	35.1	148
Carabids (Coleoptera)	346	34.1	348
RoveBeetles (Coleoptera)	723	43.3	798
LeafSeedBeetles (Coleoptera: Bruchidae & Chrysomelidae)	267	31.2	273
Ladybirds (Coleoptera: Coccinelidae)	50	23.1	50
SoldierBeetles (Coleoptera: Cantharidae and allies)	56	49.7	59
Weevils (Coleoptera:	614	26.4	621
Neuropterida	77	51.5	82
Gelechiids (Lepidoptera)	152	10.6	152
Macro-moths (various Lepidoptera)	0*	n.a.	895
Butterflies (various Lepidoptera)	0*	n.a.	59
Bees[2] (Hymenoptera: Apoidea)	241	16.4	242
Ants[2] (Hymenoptera: Formicidae)	56	39.1	60
Wasps[2] (Hymenoptera: other Aculeata)	263	15.5	263

Centipedes[3]	48	65.0	51
Millipedes[3]	59	64.2	59
Spiders (Arachnida)	621	28.7	626
Fish	61	24.6	67

[1] Included in the Riverflies Recording Schemes
[2] Included in the Bees, Wasps and Ants Recording Society
[3] Included in the British Myriapod and Isopod Group
* Macro-moths and butterflies were not used in the classification analysis because the sheer size of the dataset made it difficult to extract all the metrics. Lichens appeared to be modelled poorly, and so were not included. Hypogean crustaceans were not used in the regional analysis because the small number of species made them uninformative.

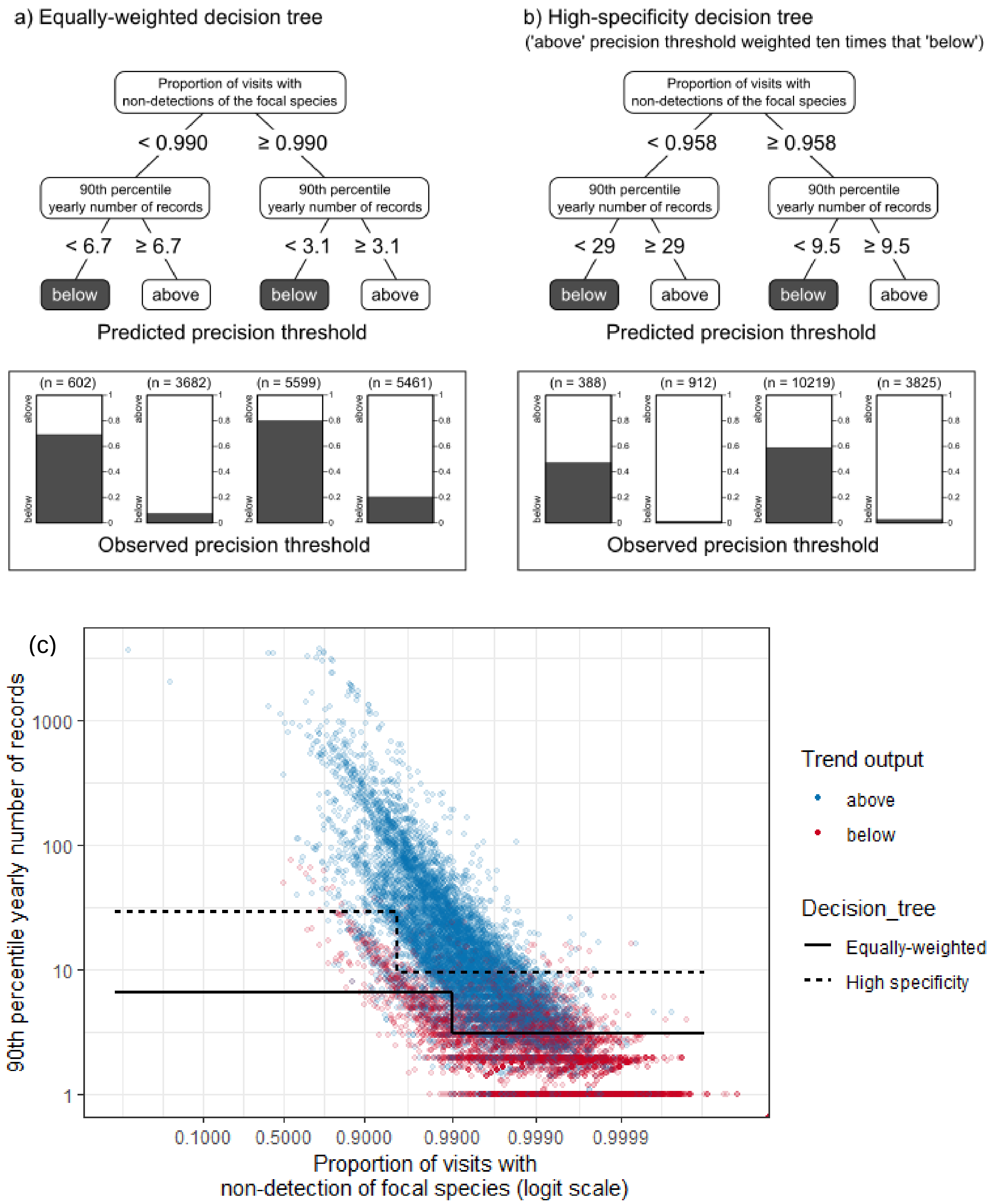

Figure 4.1. The classification trees using data metrics from 15 344 trend outputs (47-year trends and 10-year trends, treated separately for 7672 species from 31 taxonomic groups) that were (a) equally-weighted, i.e. specificity and sensitivity were balanced, and (b) high specificity, i.e. outputs above the precision threshold were weighted ten times those below. (c) This is presented graphically according to the two data metrics.

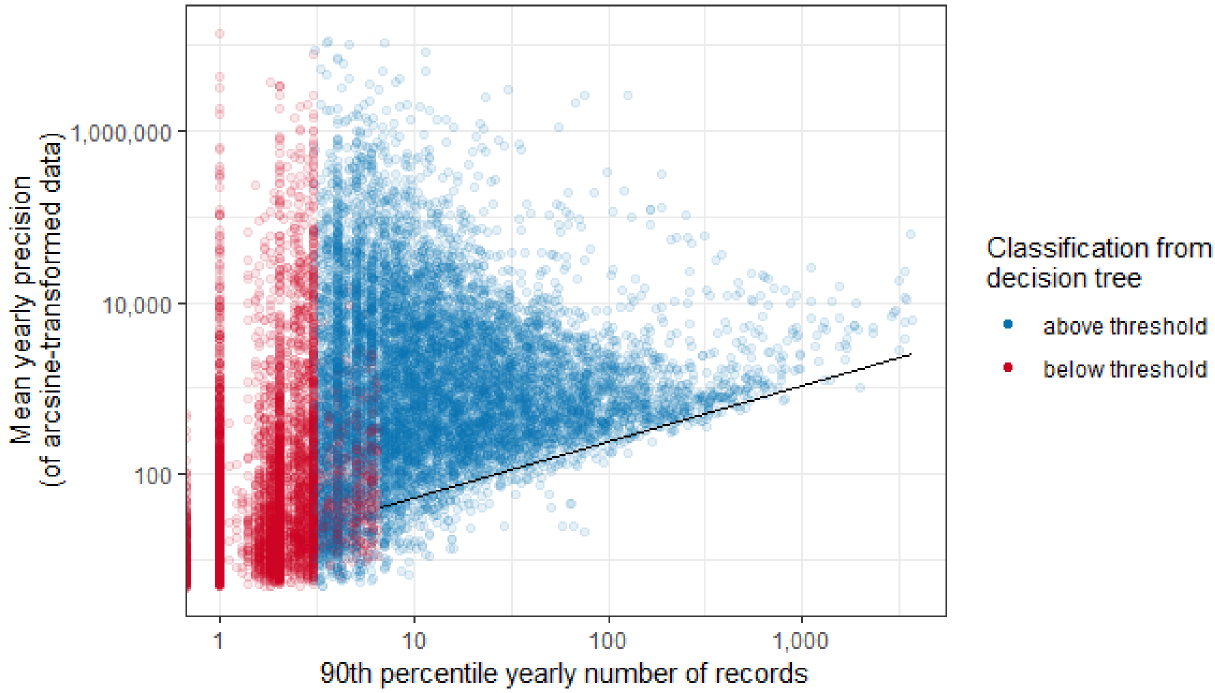

Figure 4.2 An increasing number of records in the 10% best recorded years leads to an increase in the minimum precision of occupancy estimates from analysis of occurrence data, as revealed with a quantile regression (of the 5th quantile of precision).

Table 5.1. The broad habitat type classifications from the 2015 Land Cover Map, the habitat codes used in graphs in this analysis and the median percent cover of each habitat type across all 1km squares in the UK.

Broad habitat type	**Aggregate class includes**	**Habitat code**	**Median coverage (%)**
Broadleaf Woodland	Broadleaved, Mixed and Yew Woodland	BLW	6
Coniferous Woodland	Coniferous Woodland	CWL	10
Arable	Arable and Horticultural	A	35
Improved Grassland	Improved Grassland	IG	35
Semi-natural Grassland	Neutral Grassland, Calcareous Grassland, Acid Grassland, Fen, Marsh and Swamp	SNG	15
Mountain, Heath, Bog	Heather, Heather Grassland, Bog, Inland Rock	MHB	33
Coastal	Supra-littoral Rock, Supra-littoral Sediment, Littoral Rock, Littoral Sediment, Saltmarsh	C	8
Freshwater	Freshwater	FW	3
Built-up Areas	Built-up Areas and Gardens	BU	4

Table 5.2 The number of 1km squares covered each year for each taxon. * sample sizes for 2017 were not available at time of analysis.

Year	Birds	Bats	Butterflies	Plants
2007	3,792	1,331	1,212	NA
2008	3,476	1,320	1,171	NA
2009	3,468	1,354	1,799	NA
2010	3,354	1,365	1,730	NA
2011	3,142	1,364	1,788	NA
2012	3,503	1,266	1,817	NA
2013	3,694	1,256	2,020	NA
2014	3,558	1,289	2,112	NA
2015	3,607	1,250	2,251	439
2016	3,588	1,199	2,259	470
2017	NA*	NA*	NA*	474
Total unique 1km squares	5,272	2,943	3,807	786

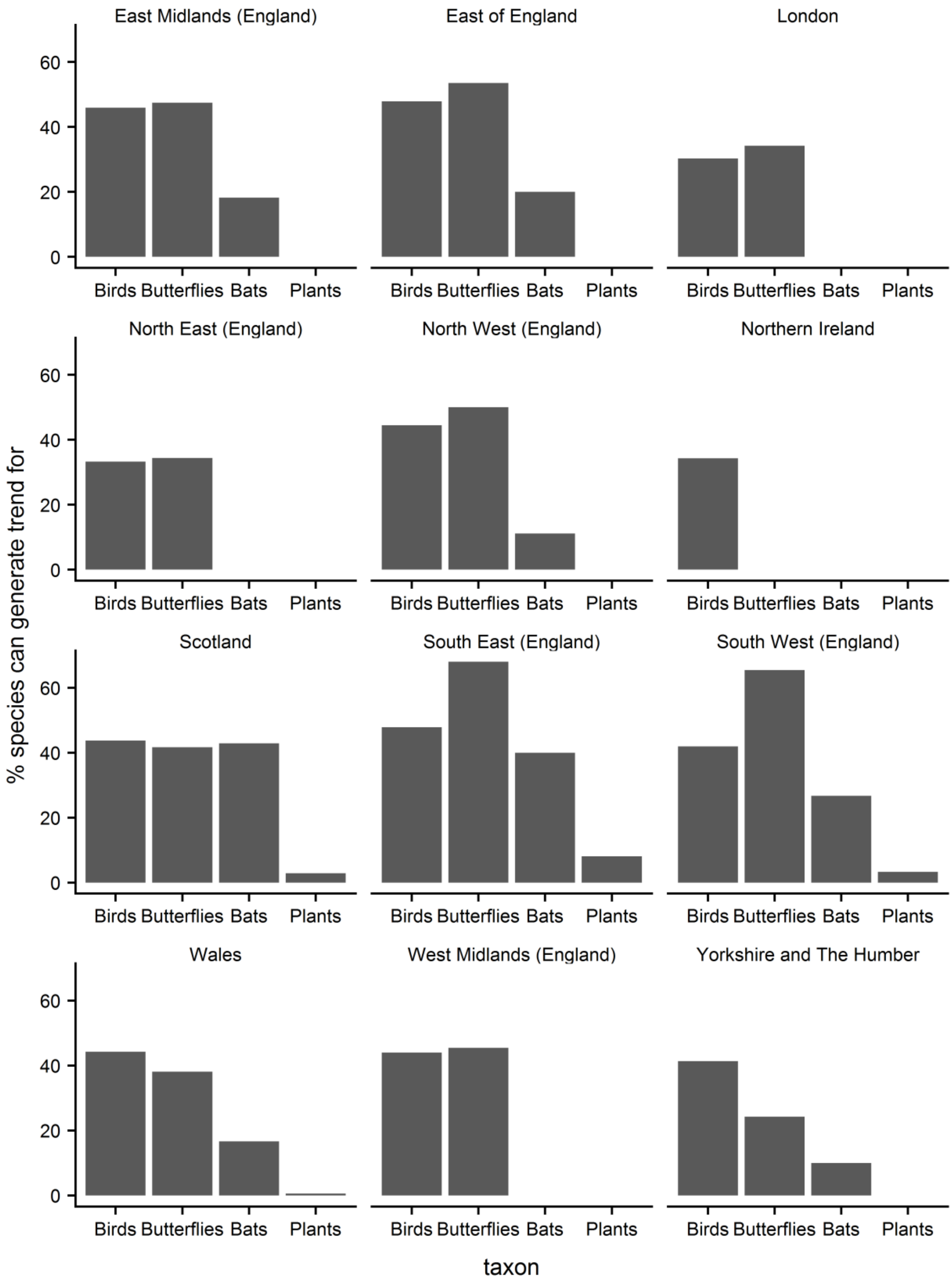

Figure 5.1. The % of species established in each region that we can generate a regional trend for (> = 30 squares on average a year).

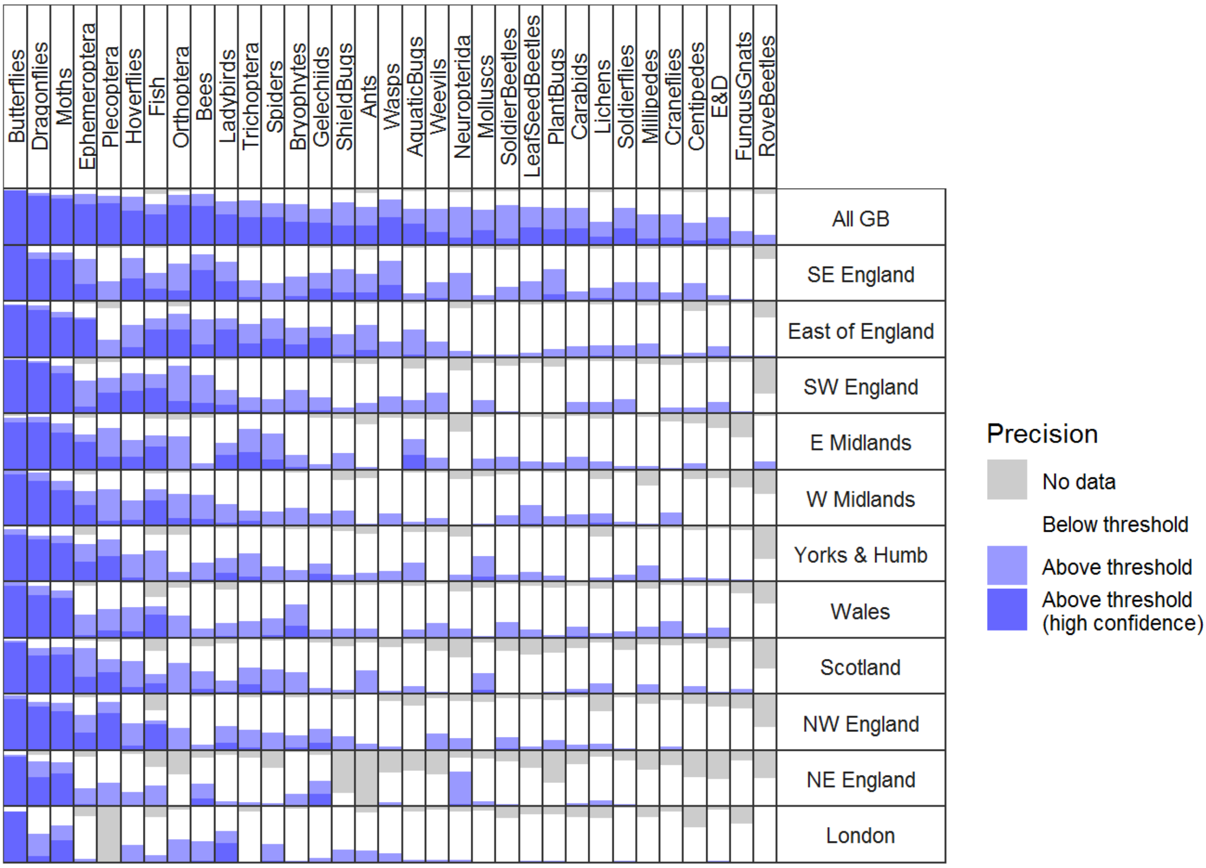

Figure 5.2. The percent of species recorded in each region that were predicted to be modelled, with occupancy analysis over the 40-year period, with acceptable or high precision. The rows and columns are ordered by decreasing coverage.

Table 5.3. The number of 1km squares containing more than the median cover of each habitat type (from Table 5.1), summarised for the whole of the UK and for squares surveyed for each taxon. Note: the number of squares is not directly comparable with the cover of habitat, some squares with higher than median covers of multiple habitats are included twice, and for fragmented habitats the number of squares will be higher than suggested by the simple area of the habitat.

Habitat	UK	Bird surveys	Bat surveys	Butterfly surveys	Plant surveys
Broadleaved Woodland	68,254	1,809	1,623	1,963	304
Coniferous Woodland	31,167	508	197	435	77
Arable	68,760	1,791	748	1,038	206
Improved Grassland	92,356	2,106	1,350	1,514	267
Freshwater	15,400	288	354	290	71
Semi-natural Grassland	37,473	583	158	385	112
Mountain, Heath,Bog	41,038	403	50	199	70
Coastal	10,914	150	107	196	94
Built up areas	53,478	1,698	1,564	1,356	248

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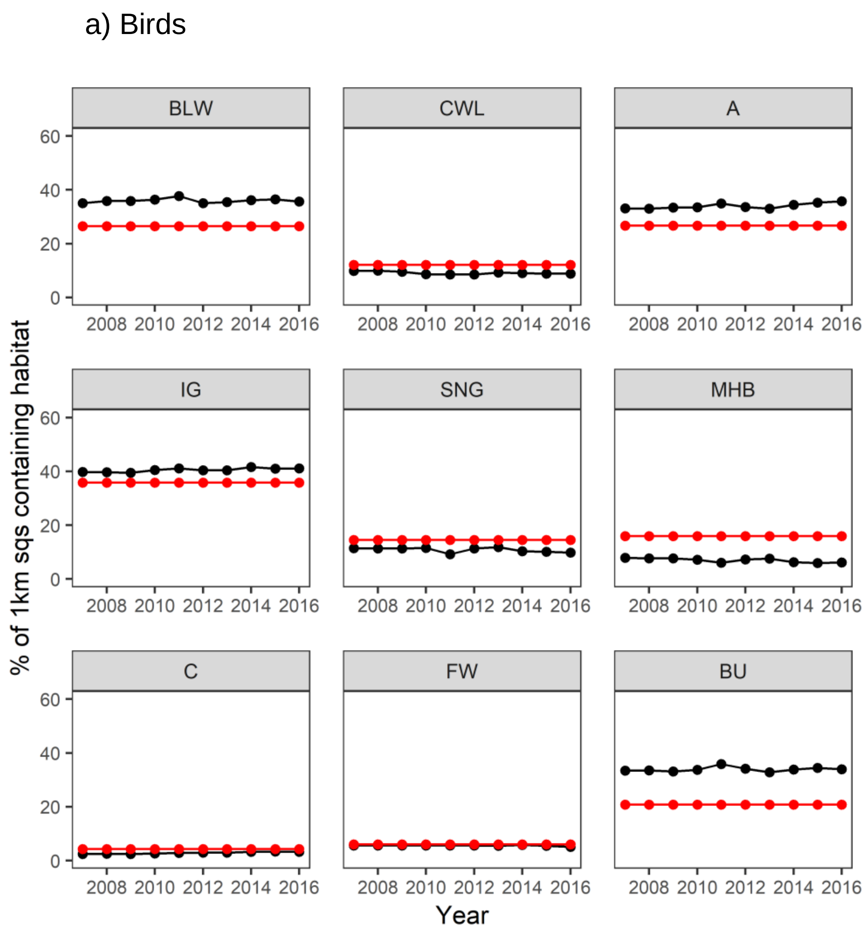
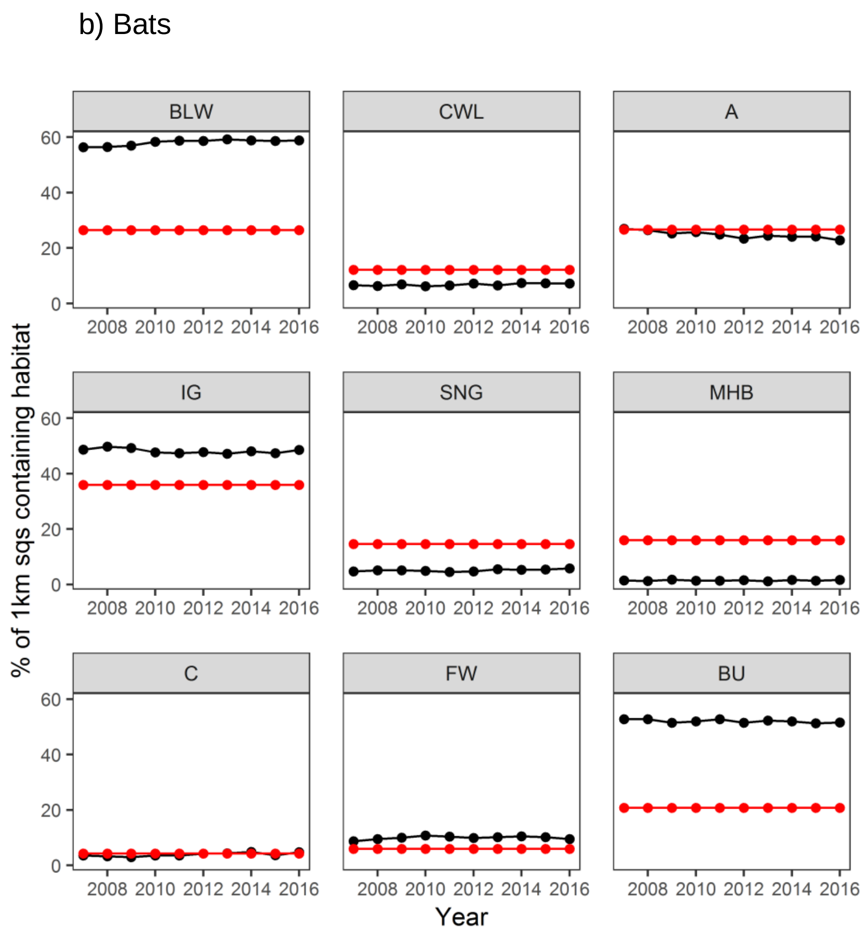

a) Birds
BLW
CWL
A
IG
SNG
MHB
C
FW
BU
% of 1km sqs containing habitat
0
20
40
60
2008 2010 2012 2014 2016
Year


b) Bats
BLW
CWL
A
IG
SNG
MHB
C
FW
BU
% of 1km sqs containing habitat
0
20
40
60
2008 2010 2012 2014 2016
Year

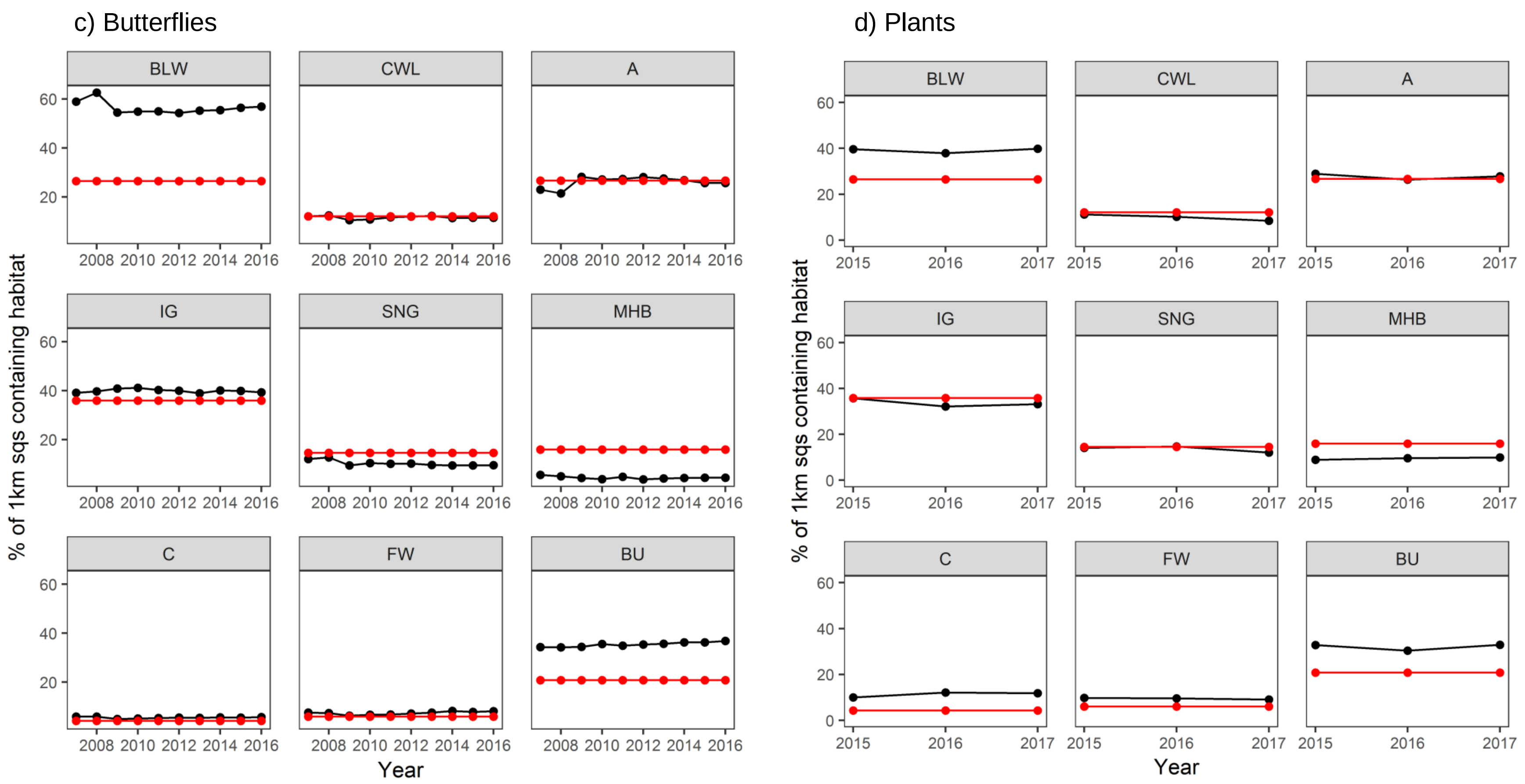


Figure 5.3. Percentage of 1km survey squares containing over the 0.50 quantile for occurrence of each habitat in the UK (black) and % of squares in the UK containing over the 0.50 quantile for occurrence of each habitat (red) for a) birds, b) bats, c) butterflies, d) plants.

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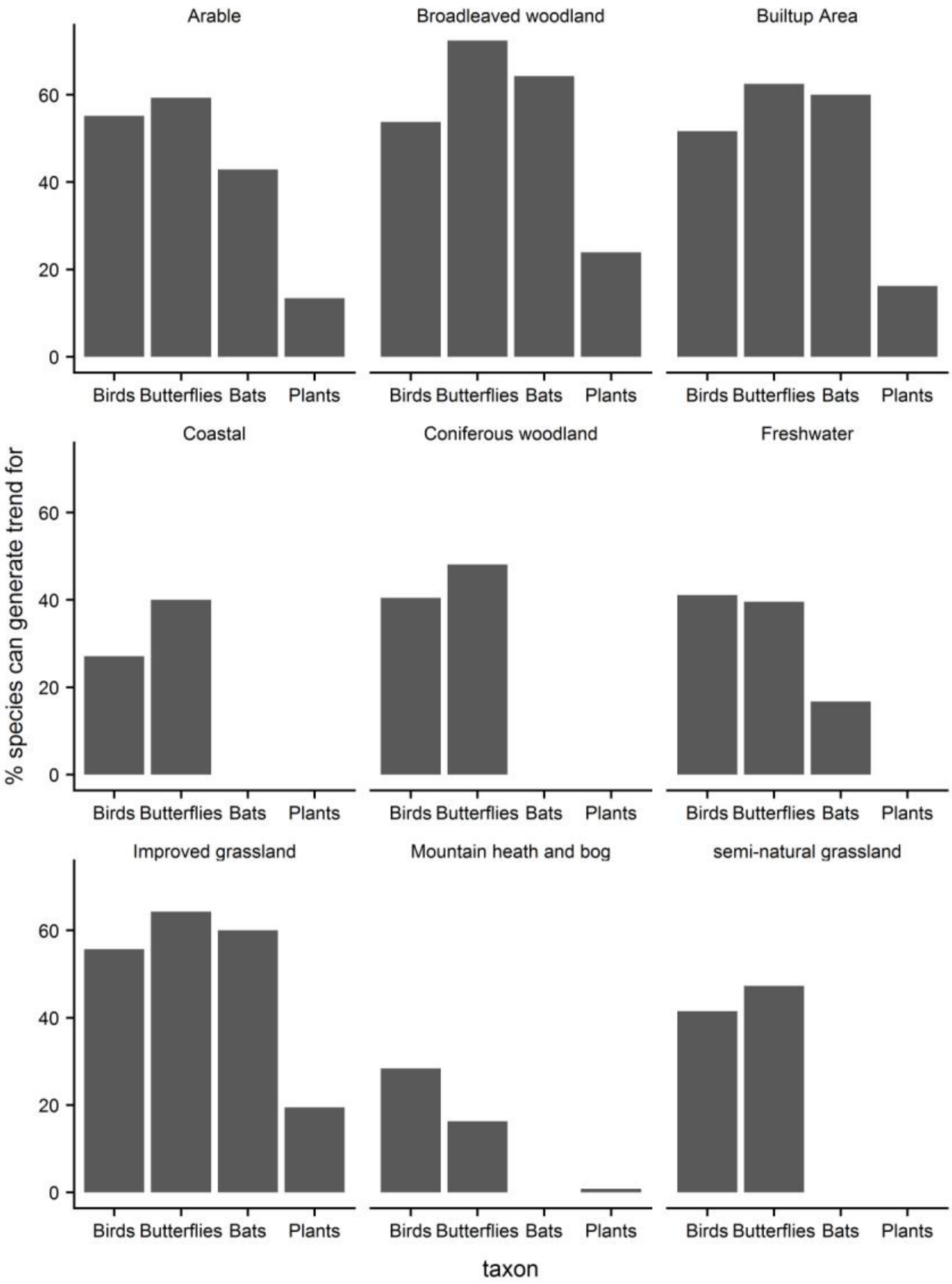


Figure 5.4. The % of species occurring in each habitat for which we can create habitat specific trends.

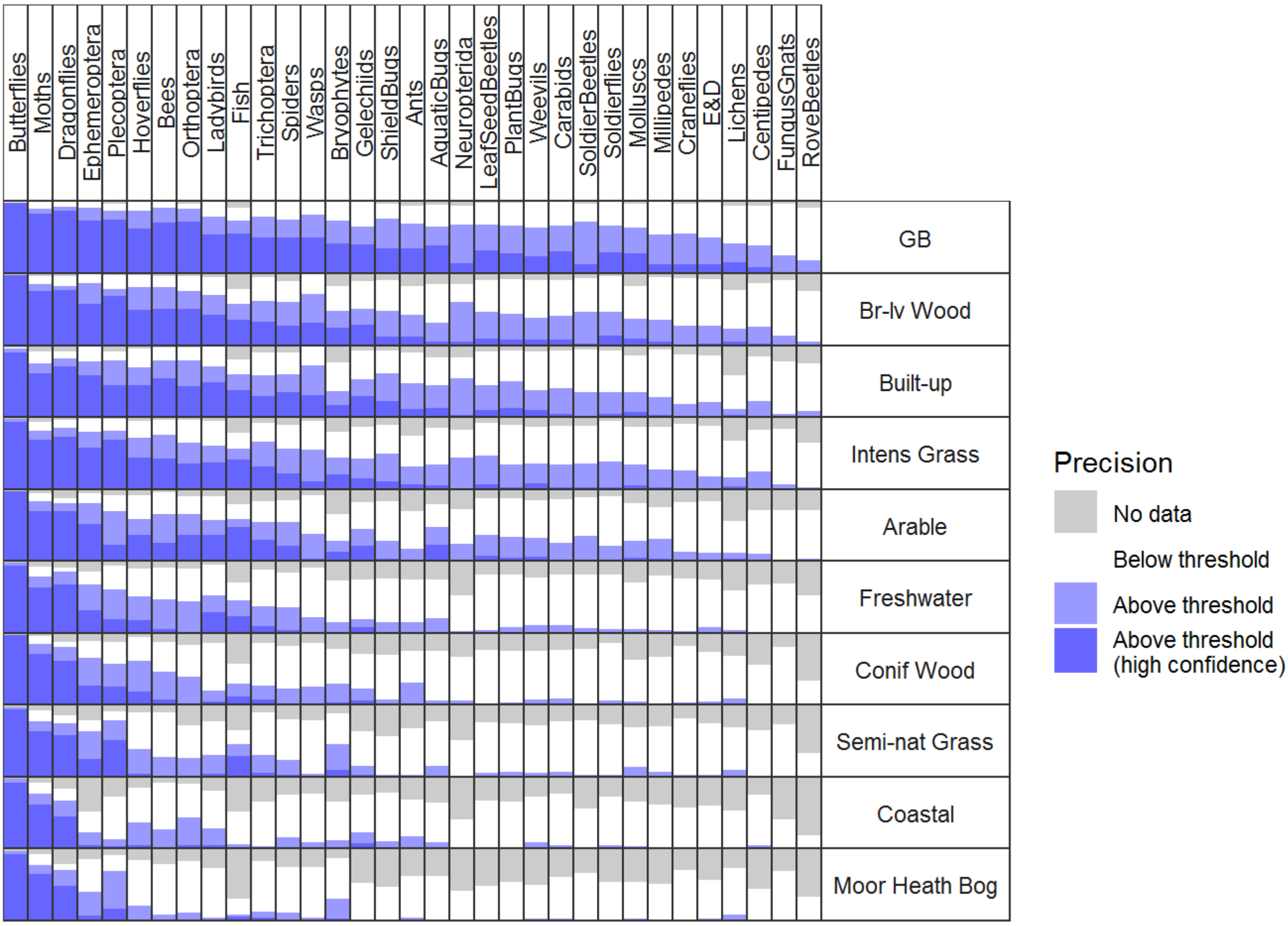


Figure 5.5. The percentage of species recorded in each habitat (strictly, the species occurring in squares which has greater than the median coverage of the habitat) that were predicted to be modelled, with occupancy analysis over the 40-year period, with acceptable or high precision. The rows and columns are ordered by decreasing coverage.

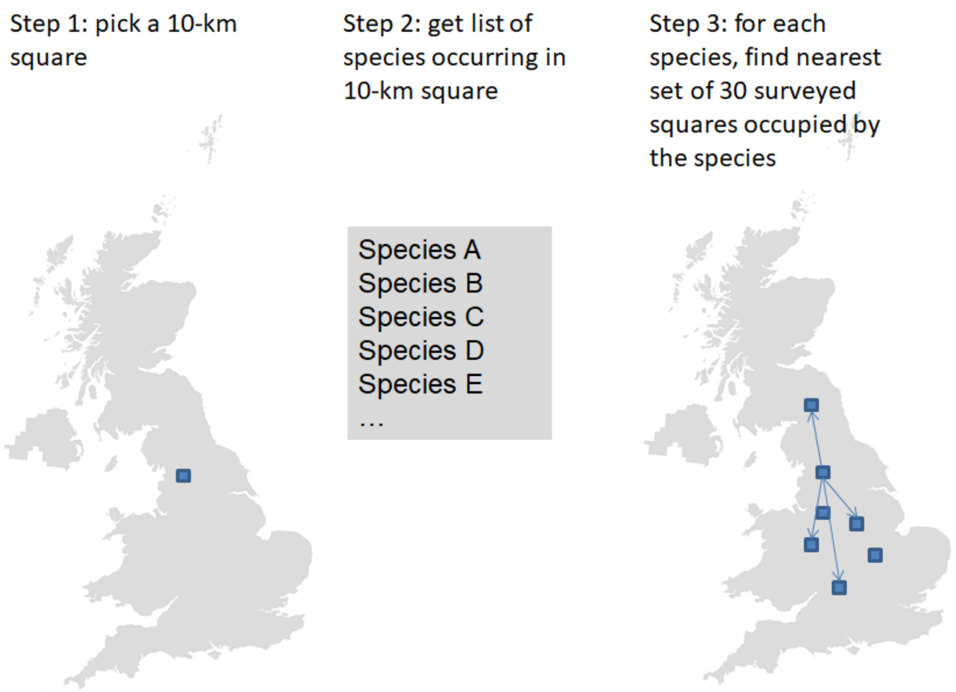


Figure 6.1. Schematic of the method used to assess spatial precision of individual species trends. Atlas and survey data were used to generate a list of species for each hectad. For each of the listed species within a hectad, the minimum distance from the hectad centroid needed to get 30 recorded occupied sites for each survey year was determined.

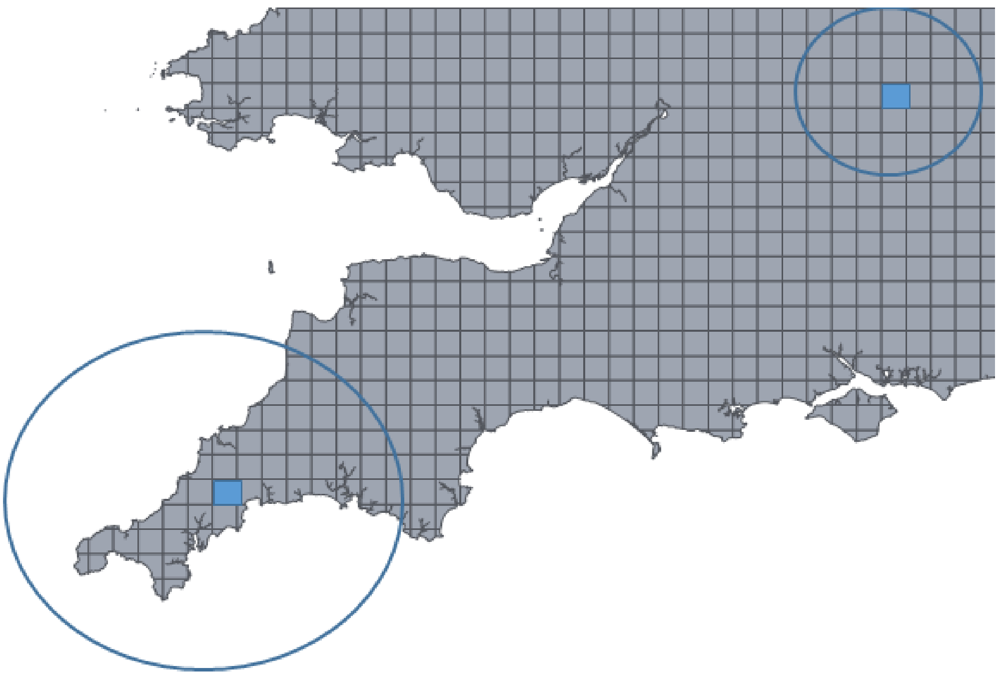
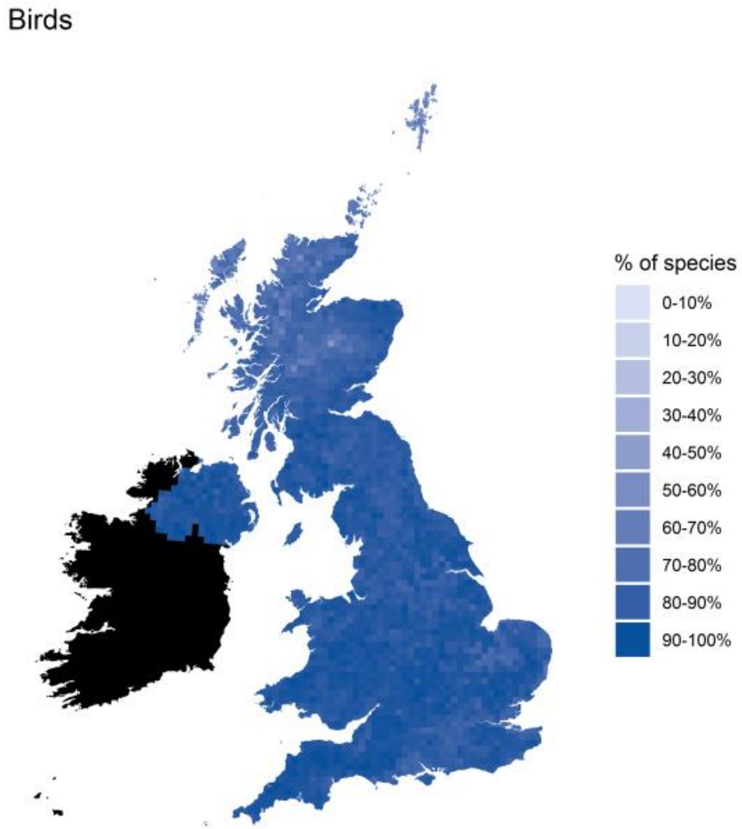
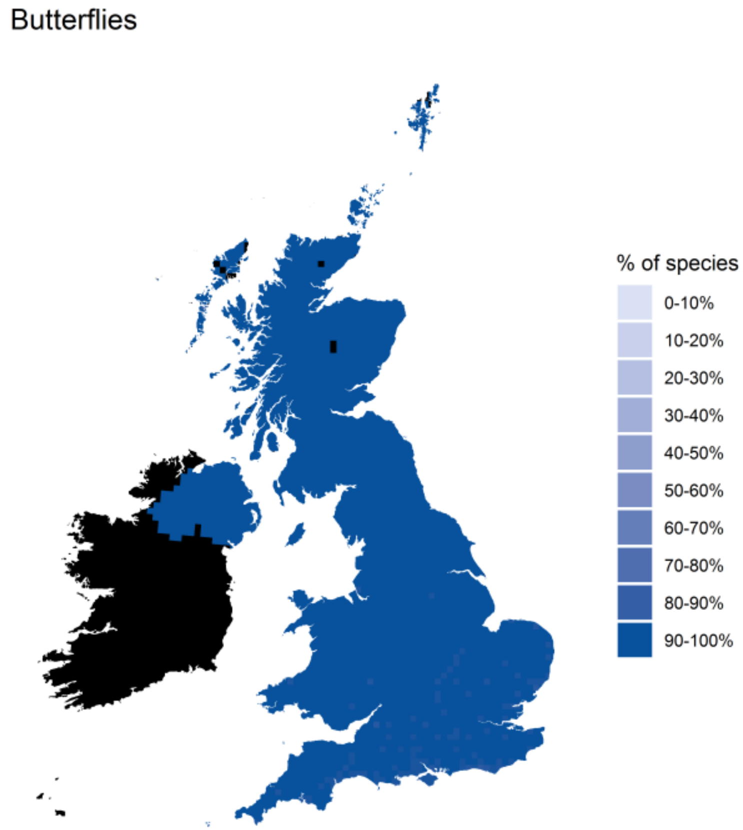


Figure 6.2. The distance value for each species in each hectad is standardized by dividing by the distance needed to find 30 hectads from the focal hectad if all squares in the UK were included. In this example you can see that a much larger area is needed to get 30 hectads if the focal hectad is coastal.


Birds
% of species
0-10%
10-20%
20-30%
30-40%
40-50%
50-60%
60-70%
70-80%
80-90%
90-100%


Butterflies
% of species
0-10%
10-20%
20-30%
30-40%
40-50%
50-60%
60-70%
70-80%
80-90%
90-100%

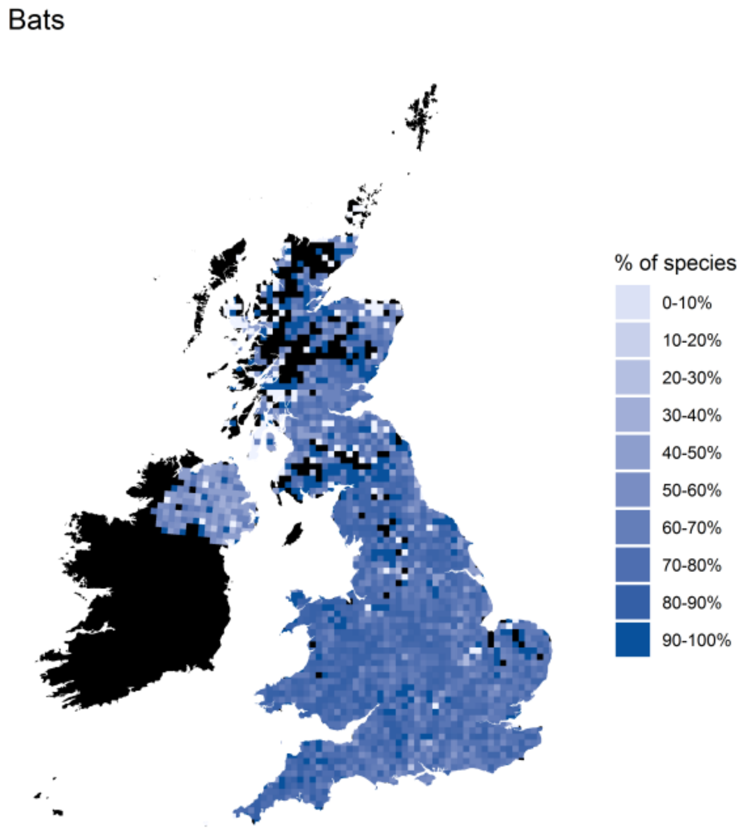
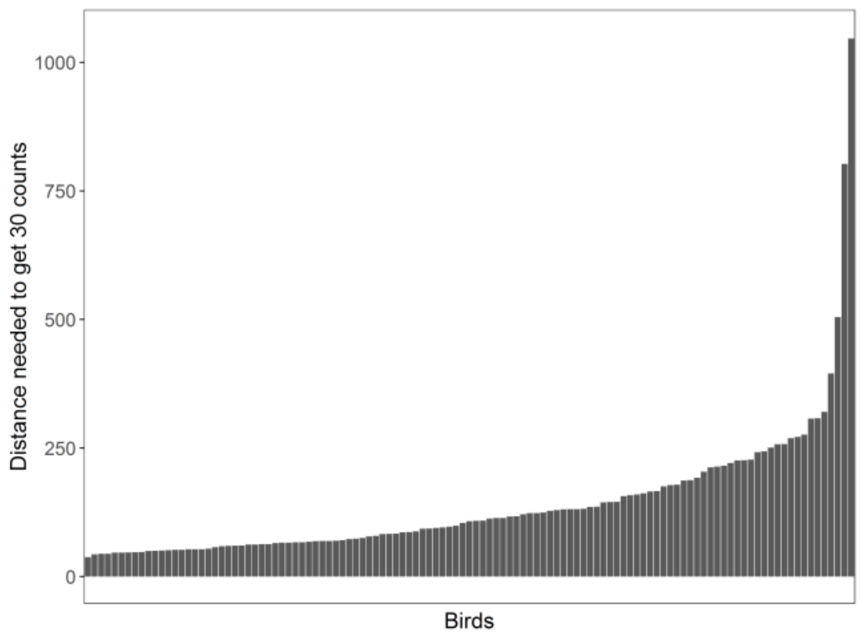
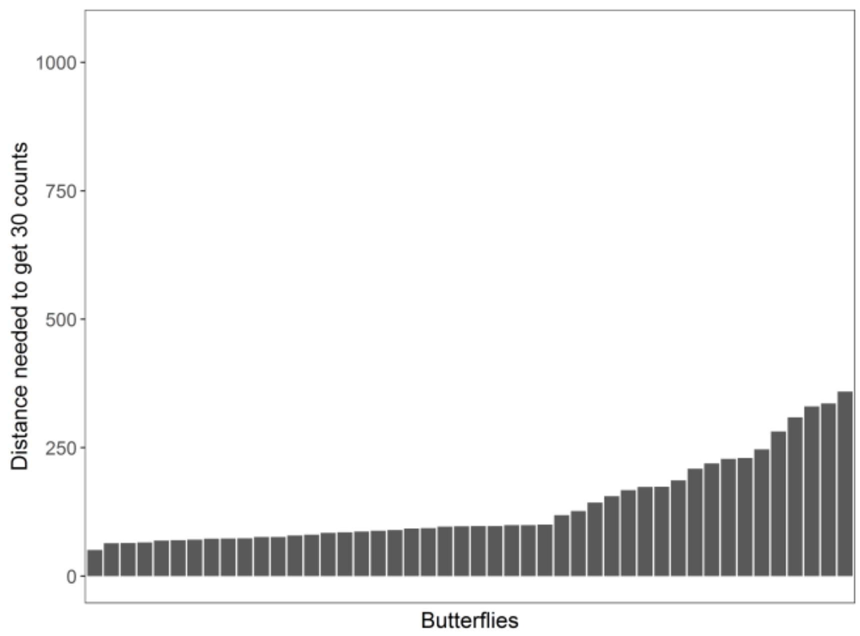


Figure 6.3. The percentage of species from each 10km square for which it is possible to generate a national trend. Black areas are areas where we have no atlas records for the taxa and we restricted bird species to those recorded as breeding.


1000
750
500
250
0
Distance needed to get 30 counts
Birds


1000
750
500
250
0
Distance needed to get 30 counts
Butterflies

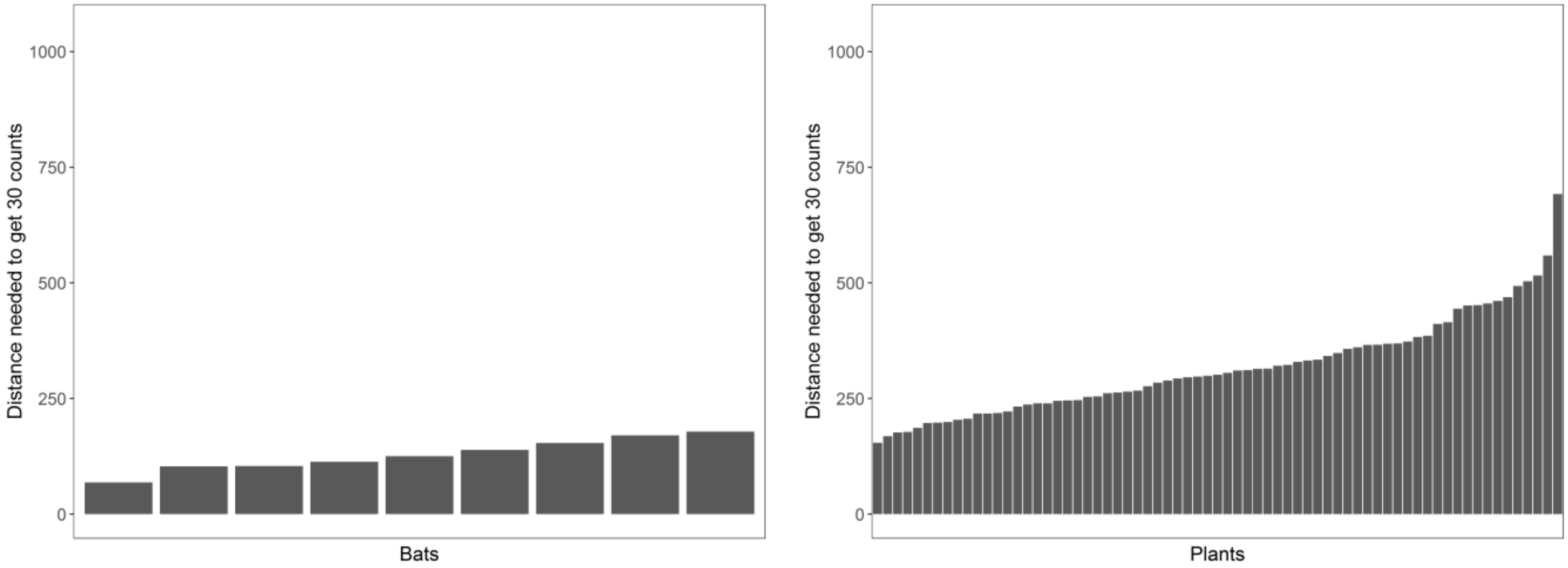
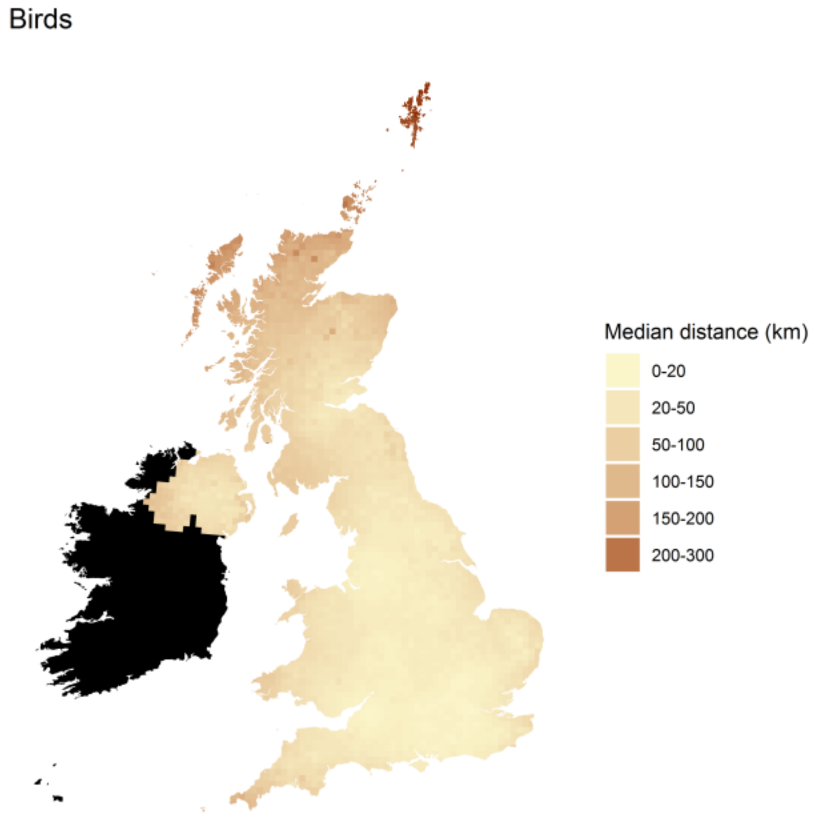
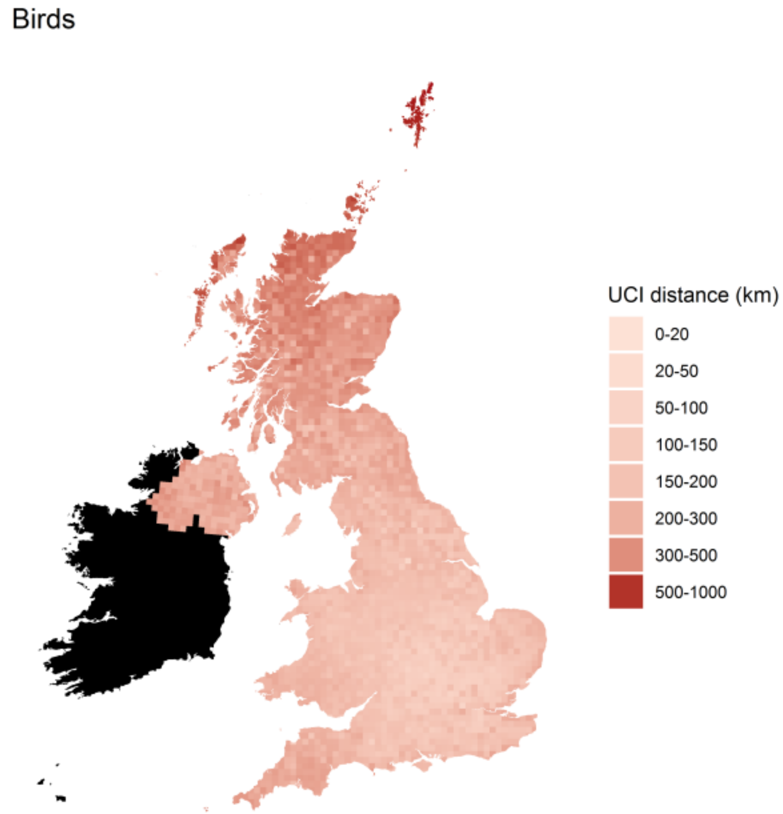
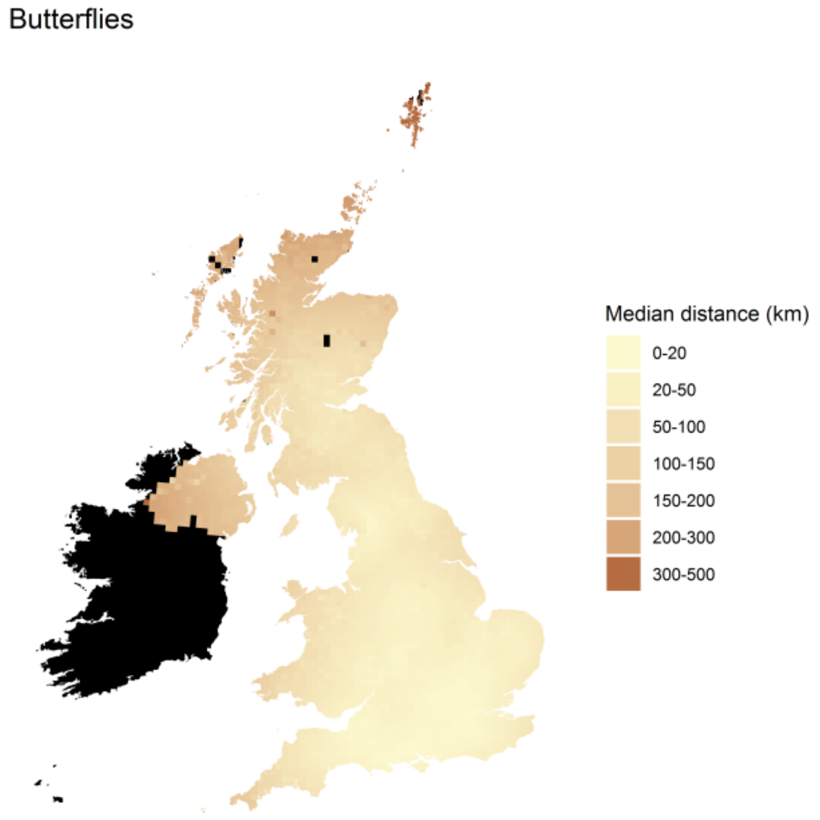
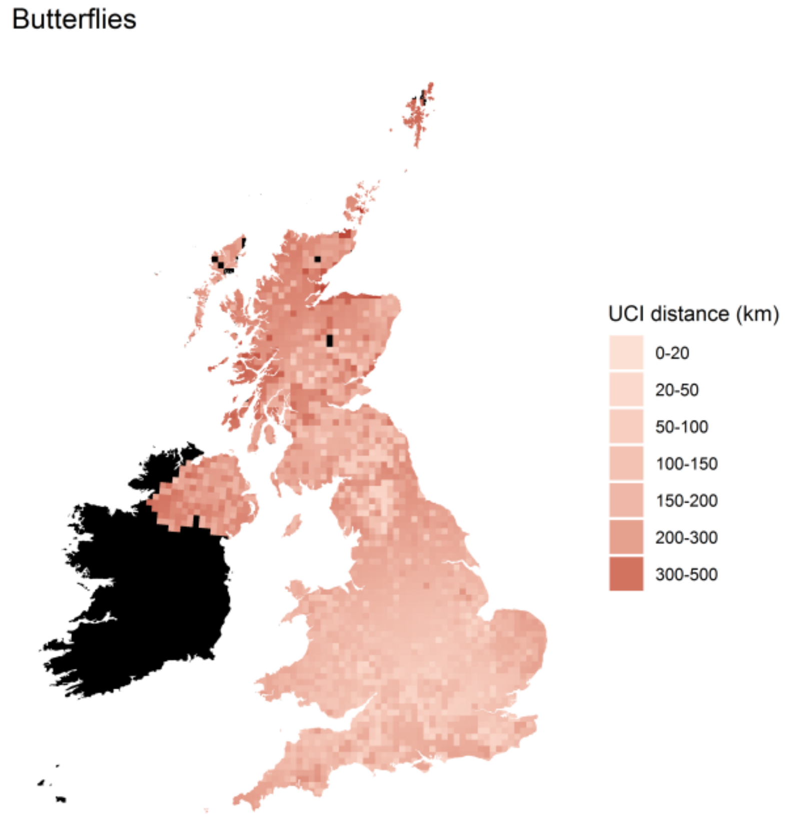


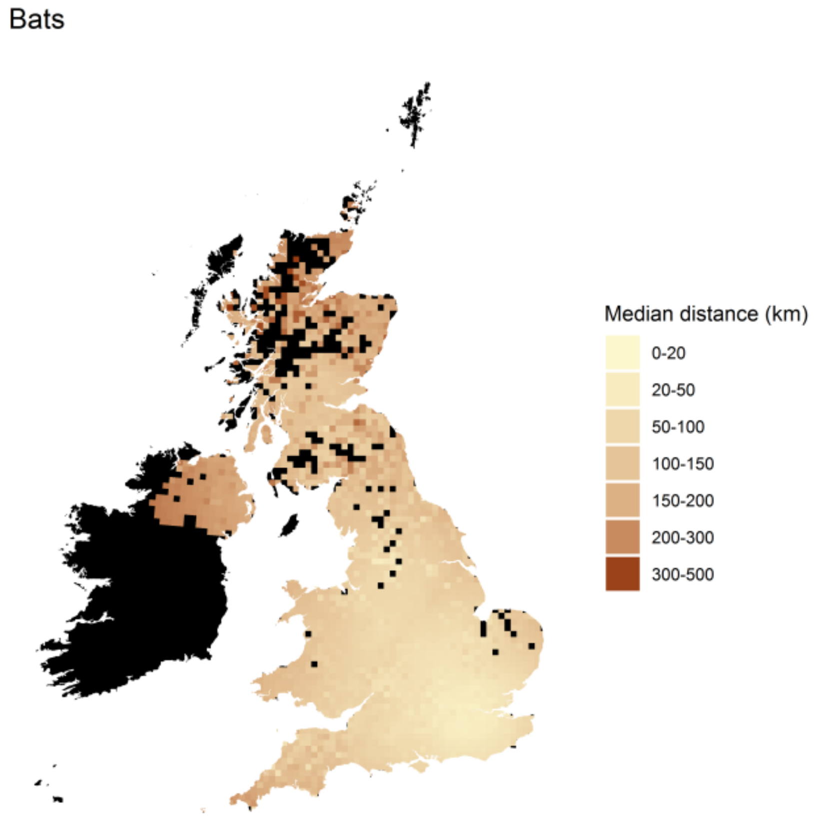
Figure 6.4. The mean distance needed to find 30 counts for each species averaged over all 10 km squares in the UK for a) birds b) butterflies, c) bats d) wildflowers. Each bar is a different species. Only the species for which it is possible to get 30 counts are included here (115 birds (58% of total recorded in atlas), 46 butterflies (71% of total recorded in atlas), 9 bats (53% of total recorded in atlas), 69 wildflowers (33% of total recorded in atlas)).

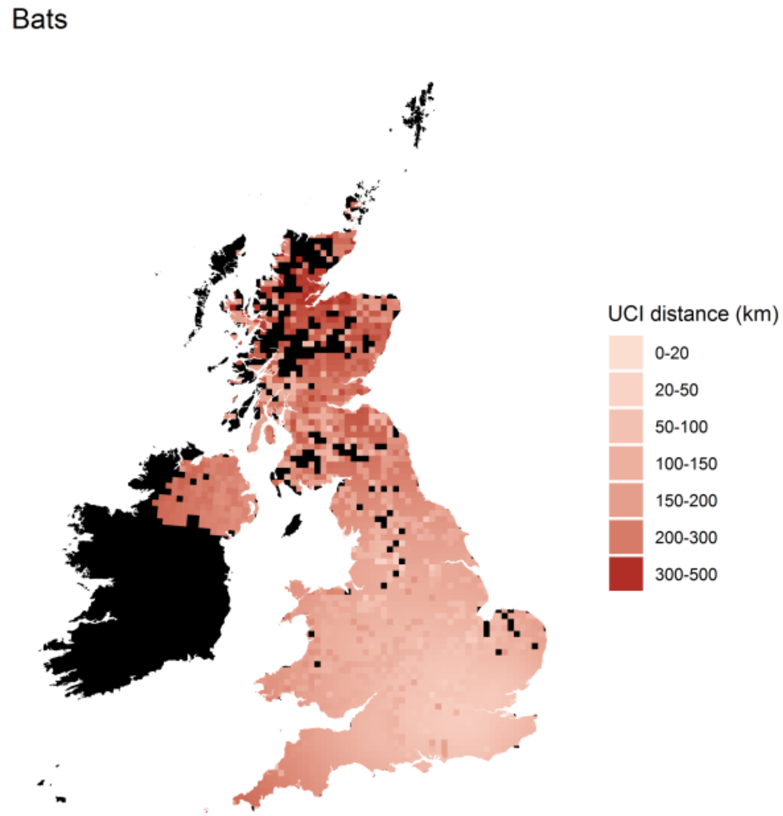

Birds
Median distance (km)
0-20
20-50
50-100
100-150
150-200
200-300


Birds
UCI distance (km)
0-20
20-50
50-100
100-150
150-200
200-300
300-500
500-1000


Butterflies
Median distance (km)
0-20
20-50
50-100
100-150
150-200
200-300
300-500


Butterflies
UCI distance (km)
0-20
20-50
50-100
100-150
150-200
200-300
300-500


Bats
Median distance (km)
0-20
20-50
50-100
100-150
150-200
200-300
300-500


Bats
UCI distance (km)
0-20
20-50
50-100
100-150
150-200
200-300
300-500

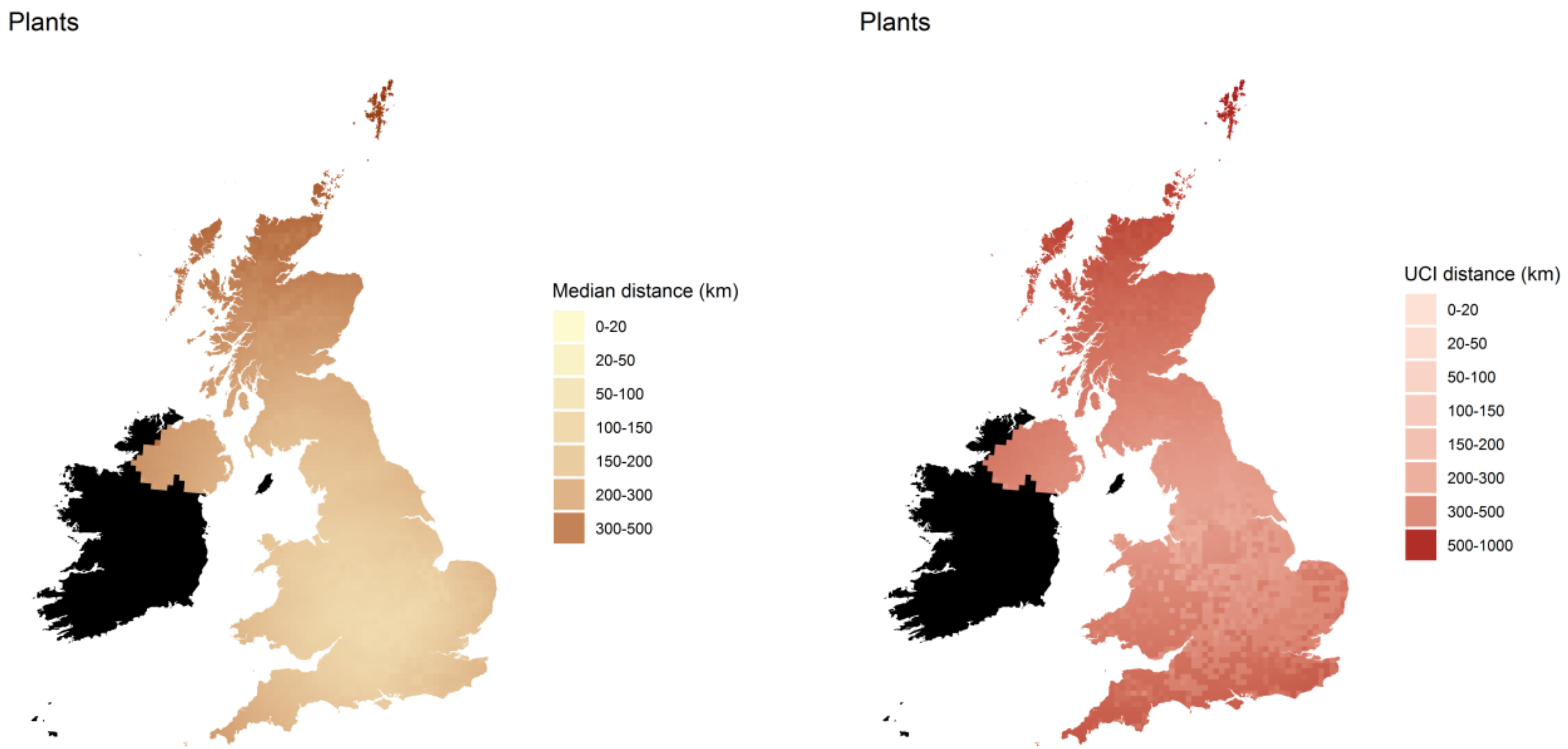
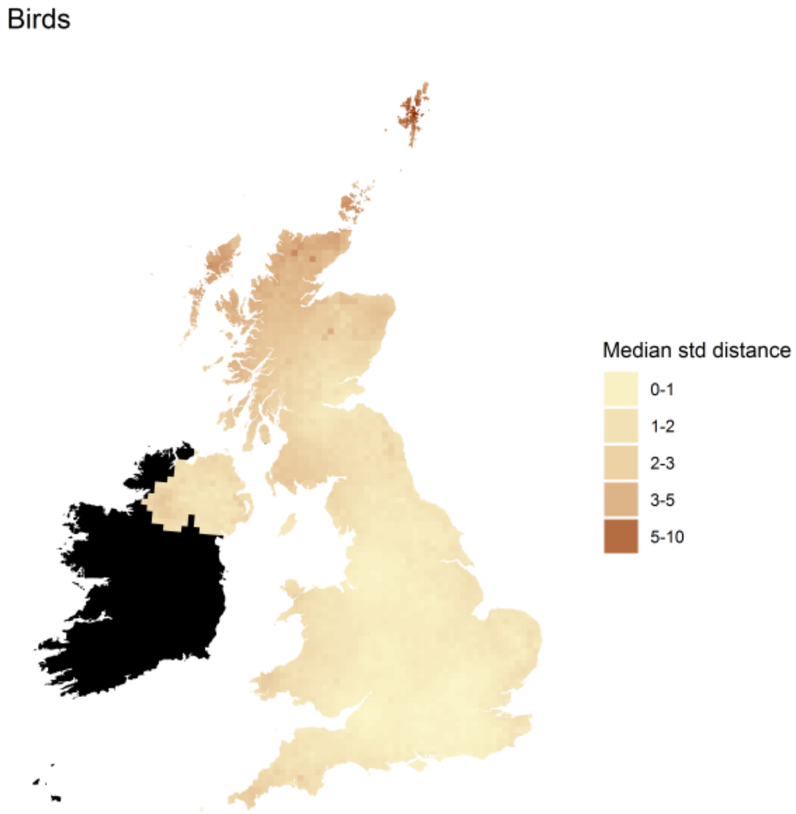
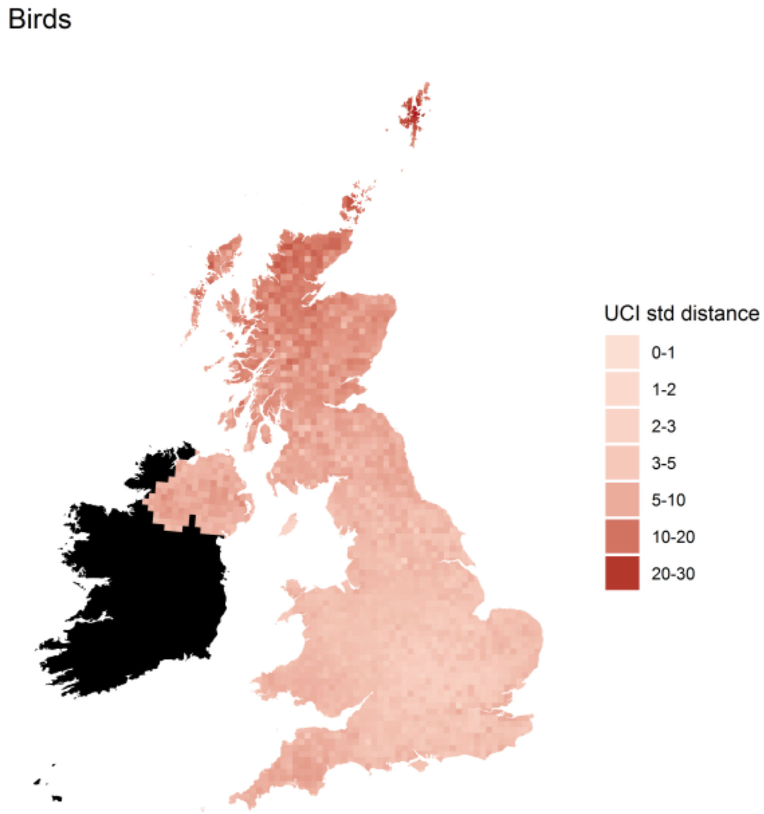
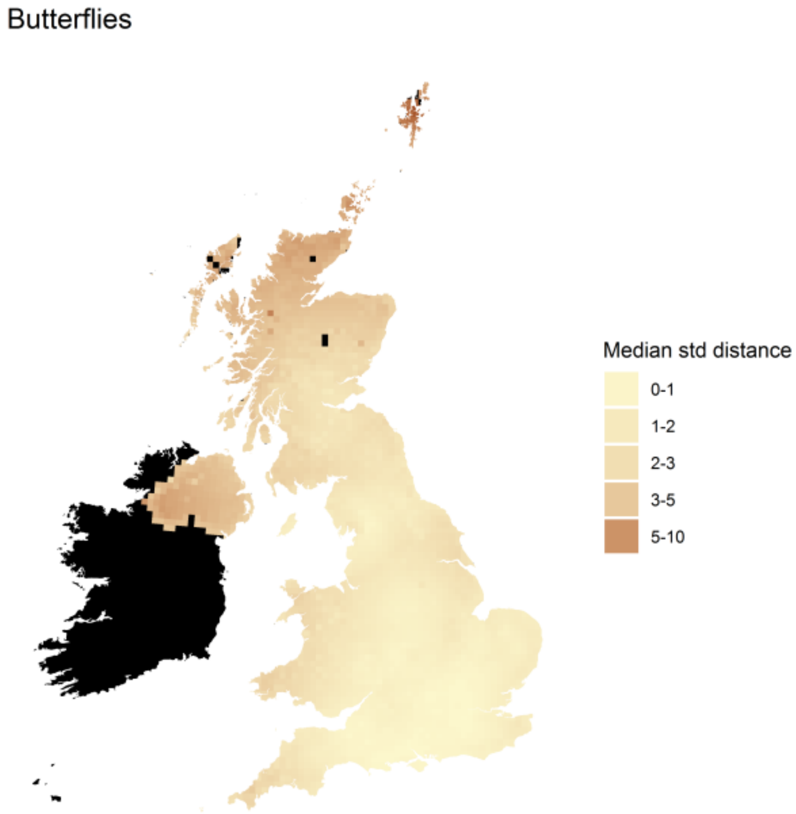
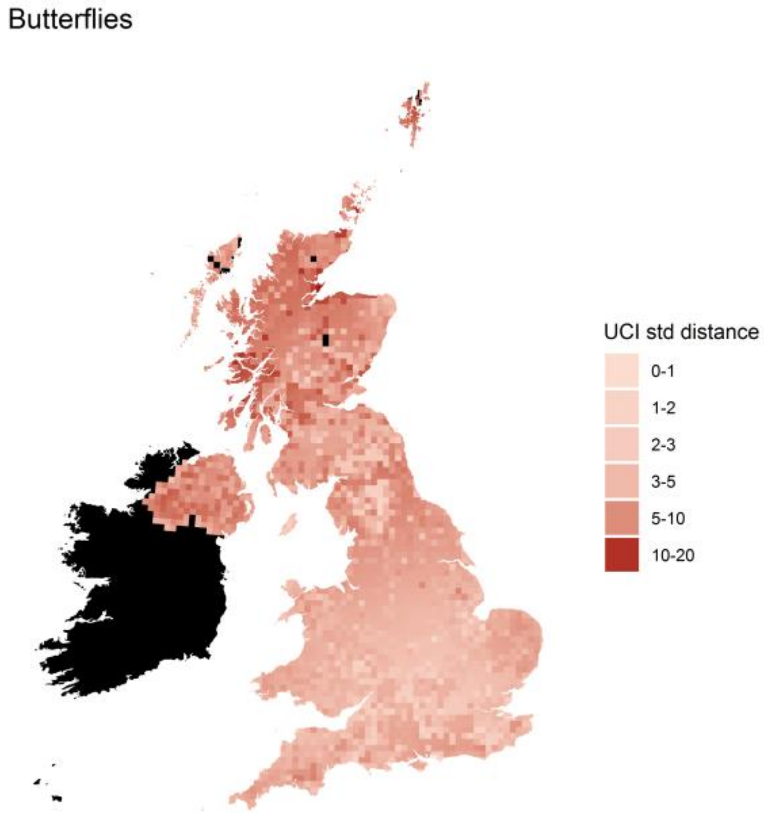


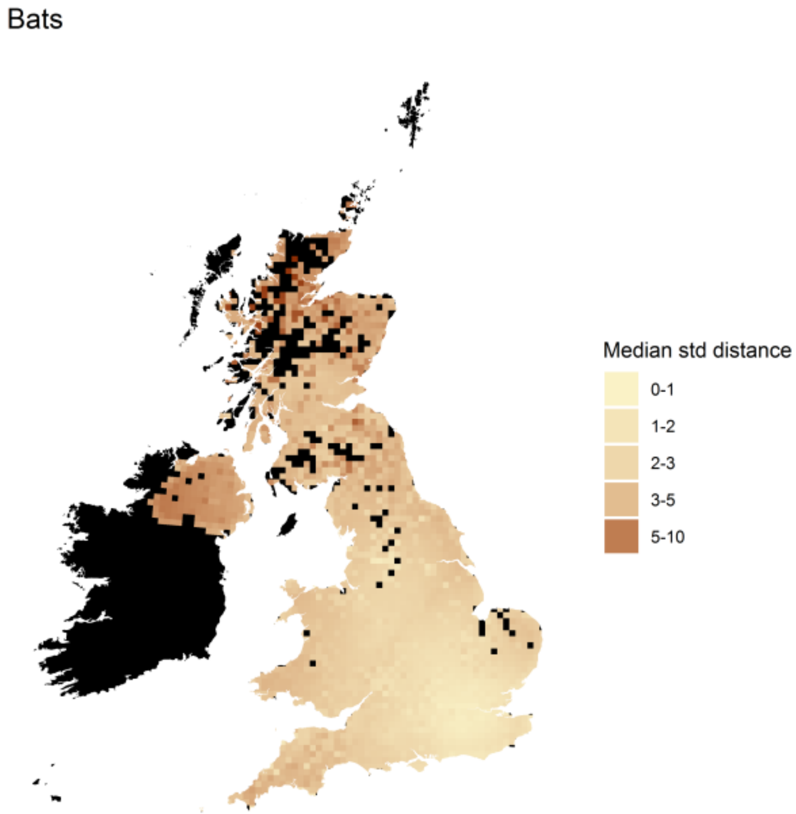
Figure 6.5. Maps of the median and upper 95% confidence intervals of the distance needed to reach the threshold level of 30 survey counts averaged over all the established species within each hectad. Black areas are areas where we have no atlas records for the taxon.

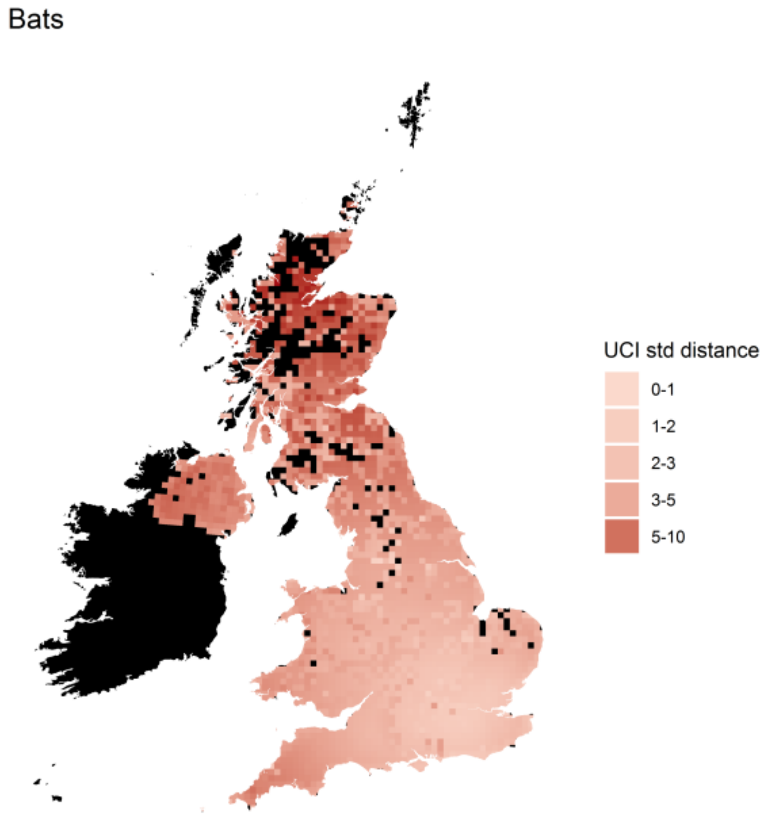

Birds
Median std distance
0-1
1-2
2-3
3-5
5-10


Birds
UCI std distance
0-1
1-2
2-3
3-5
5-10
10-20
20-30


Butterflies
Median std distance
0-1
1-2
2-3
3-5
5-10


Butterflies
UCI std distance
0-1
1-2
2-3
3-5
5-10
10-20


Bats
Median std distance
0-1
1-2
2-3
3-5
5-10


Bats
UCI std distance
0-1
1-2
2-3
3-5
5-10

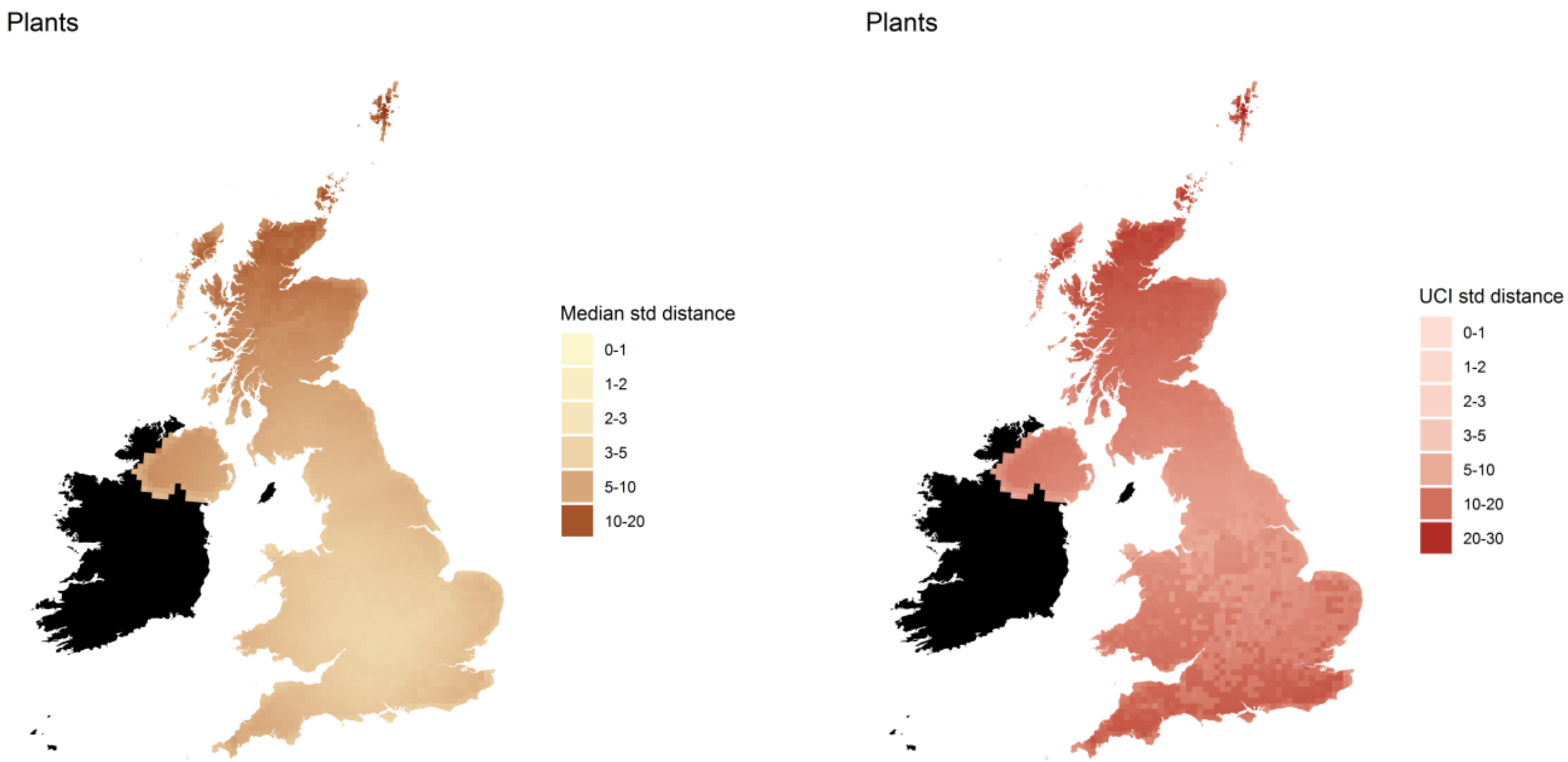


Figure 6.6. Maps of the median and upper 95% confidence intervals of the standardized distance to reach the threshold level of 30 survey counts averaged over all the established species within each hectad. Black areas are areas where we have no atlas records for the taxon. The standardised distance is the distance from the target hectad to get 30 samples divided by the distance needed to reach 30 hectads from the target hectad.

11 Annex 1

Raw scores of individual species groups in the taxonomic coverage analysis.

See separate Excel file: JNCC-Report-646-Annex-1-Taxonomic-coverage-table