A Review of the Population and Conservation Status of British Mammals

Mathews F.,*[1] Kubasiewicz L. M.,*[2] Gurnell J.,[3] Harrower C. A.,[4] McDonald R. A.,[5] Shore R. F.[6]

A report by the Mammal Society under contract to Natural England, Natural Resources Wales and Scottish Natural Heritage

[1] The Mammal Society and the University of Sussex, John Maynard Smith Building, Falmer, Brighton, BN1 9QG.

[2] The Mammal Society, 18 St John's Church Road, London, E9 6EJ.

[3] Queen Mary University of London, Mile End Road, London, E1 4NS.

[4] Centre for Ecology & Hydrology, MacLean Building, Crowmarsh Gifford, Wallingford, OX10 8BB.

[5] Environment and Sustainability Institute, University of Exeter, Penryn Campus, Penryn, Cornwall, TR10 9FE.

[6] NERC Centre for Ecology & Hydrology, Lancaster Environment Centre, Library Avenue, Bailrigg, Lancaster, LA1 4AP.

* Joint-first authors.

Author for correspondence: Fiona Mathews F.mathews@sussex.ac.uk; chair@themammalsociety.org

This publication should be cited as:

Mathews F, Kubasiewicz LM, Gurnell J, Harrower CA, McDonald RA, Shore RF. (2018) *A Review of the Population and Conservation Status of British Mammals. A report by the Mammal Society under contract to Natural England, Natural Resources Wales and Scottish Natural Heritage*. Natural England, Peterborough. ISBN 978-1-78354-494-3.

Natural England Access to Evidence Catalogue code JP025.

Contents

Executive summary

1. We present the first comprehensive review of the status of British mammal populations for over 20 years. The population size, range size, temporal trends and future prospects of Britain’s 58 terrestrial mammals are assessed. Island sub-species and feral mammals are excluded from the main review, but are considered in the appendix.

2. Britain has 44 native species that arrived before the formation of the English Channel. Previously extinct in the wild, the beaver has been reintroduced into Scotland and England in the last decade. The wild boar, also previously extinct, has been the subject of several illegal releases over recent years: the provenance of current populations is unknown. There are 7 species that, although introduced by human activities, are considered naturalised and have formed part of Britain’s fauna since at least Roman times. The remaining 7 species are more recent introductions.

3. The geographical ranges of 18 species have increased since 1995; 4 have declined; and 22 have remained stable. A lack of data prevented assessment of the remaining 14 species.

4. Population sizes have increased since 1995 in 15 species; 9 have declined; and 4 have remained stable. A lack of data prevented assessment of the remaining 30 species.

5. All of the species recently introduced to Britain show an increase in geographical range except the brown rat, which is stable, and the American mink, where there are differences between countries. Additionally, all show an increase in population size except the brown rat which appears — on the basis of very poor data — to be stable, and the American mink which appears to be in decline. There are important data deficiencies for all introduced species that need to be addressed urgently.

6. Among native and naturalised species where change could be assessed with reasonable confidence, there have been increases in the geographical range of the following animals:

- Otter, pine marten and polecat.

- Red, fallow and roe deer.
- Greater and lesser horseshoe bat.
- Beaver and wild boar (both of which have become established since the last review, following releases from unknown sources).

Population sizes have increased for the following species:

- Otter, pine marten, polecat and badger.
- Red and roe deer.
- Greater and lesser horseshoe bat.
- Beaver and wild boar.

7. Among native and naturalised species where change could be assessed with reasonable confidence, there have been decreases in the range of the following species:

- Red squirrel.
- Black rat.
- Wildcat.
- Grey long-eared bat.

8. Population sizes have declined for the following species:

- Hedgehog.
- Rabbit.
- Red squirrel.
- Hazel dormouse.
- Orkney vole.
- Water vole.
- Black rat.
- Wildcat.

9. Formally approved Regional Red List assessments, conducted for native species in Great Britain according to the International Union for Conservation of Nature (IUCN) criteria. Approved assessments were also made for the Orkney vole, which is naturalised, because it is officially recognised as an island sub-species; and for the

lesser white-toothed shrew because of uncertainty about whether it is naturalised or native. The assessments placed 26 species in the Least Concern category (meaning that the risk of extinction in the near future is low). Ten native species, plus the Orkney vole, were classified as Threatened (meaning that they face a high risk of extinction). Five native species, plus the lesser white-toothed shrew, were classified as Near Threatened (meaning that they were close to qualifying as Threatened, or are likely to qualify in the near future). Insufficient evidence was available to allow assessment of the other 4 species.

10. All species under review lacked some of the data required for robust estimation of population size. The most common issue was that no information was available on the percentage of potentially suitable habitat within the range that was actually occupied. In these cases, 100% occupancy was assumed, which will usually have led to overestimated population sizes. For example, the Bechstein's bat was assumed to be present in all deciduous woodland, and the red deer in all woodland, within their geographical range.

Robust population density data were lacking for all bats, with the exception of the greater and the lesser horseshoe. There were insufficient data to permit population size estimation at all for the whiskered, Brandt's and Alcathoe bats (cryptic species), barbastelle bat, Leisler's bat, and the potentially migratory Nathusius' pipistrelle bat. One other bat, the noctule, also had a score of zero for population estimate reliability. For this species, estimates could be computed, but they were based on very restricted data, resulting in correspondingly large confidence intervals.

Reliability scores of zero were also assigned to the population estimates for the water shrew, lesser white-toothed shrew, harvest mouse, and weasel, and it was not possible to compute a population estimate at all for the Orkney vole. Overall, 40% of the non-bat species, including all of the shrews, had very poor reliability scores (<=1).

11. Several drivers were associated with temporal changes in population size or range. Fifteen species are currently controlled to reduce their impact on the environment or on other species. Eighteen species have been affected by changes in habitat quality or availability since 1995.

12. The review presents the most up-to-date assessment of population size and status for the 58 terrestrial mammals in Britain. It highlights an urgent requirement for more

research to assess population densities in key habitats, and to assess the percentage of potentially suitable habitat where a given species actually occurs: at present, uncertainty levels are unacceptably high. It is possible that declines in many species are being overlooked because a lack of robust evidence precludes assessment. There is also an urgent need to quantify precisely the scale of declines in species such as the hedgehog, rabbit, water vole and grey long-eared bat, and, where necessary, to identify the causal factors. Finally, effective and evidence-based strategies for mammal conservation and management must be developed.

Acknowledgements

We would like to thank Katherine Walsh, Liz Halliwell, Jean Matthews, Rob Raynor, Rebecca Clews-Roberts and Sam Dyer from the Statutory Nature Conservation Bodies of England, Wales and Scotland for initiating the project, reviewing numerous interim drafts, and for providing ongoing support and feedback over the last two and a half years. Kate Morris provided valuable guidance on the interpretation and application of Red List criteria. We are grateful to Derek Crawley for coordinating data collection for the distribution maps, providing advice on species' ecology, and commenting on earlier drafts, and to Katherine Boughey for support in the expert consultation process and GIS work. Thanks go to Charlotte Marshall, Charlie Kendall, Emily Churchill, Frazer Coomber, Hannah Thorley, Luke Romaine, Thomas Bemment, Quinn Egner and Rina Quinlan for collating expert consultation data, editing and providing general project support. We are also very grateful to Tim Kendall for his weeks of careful proof-reading. Paul Lintott, Patrick Wright and Domhnall Finch helped to keep us smiling in the office.

We are grateful for the input we received during the expert consultation, for the provision of advice and guidance on specific species, and for comments on previous drafts of the review. We thank Alastair Ward, Amanda Lloyd, Andrew Crawford, Andrew Harrington, Andrew Kelly, Anna Champneys, Annette Faulkner, Anthony Caravaggi, Arnie Cooke, Brian Boag, Brian Ribbands, Catherine O'Reilly, Carol Williams, Chris Damant, Chris Sherlock, Colin Morris, Daniel Whitby, Daniel Hargreaves, Danielle Linton, David Noble, Dawn Scott, Debbie Alston, Deborah Wright, Derek Gow, Ed Wells, Elisa Fuentes Montemayor, Elizabeth Chadwick, El Kean, Emma Sheehy, Eric Palmer, Gareth Harris, Graham Scholey, Henry Schofield, Ian Bond, Ian White, Jimmy Irvine, Jochen Langbein, Johanna Judge, John Flowerdew, John Haddow, Johnny Birks, Kate Barlow, Kerry Kilshaw, Keith Cohen, Kirsty Park, Leo Gubert, Liz Halliwell, Lizzie Croose, Martin Gaywood, Matt Hayward, Mel Tonkin, Melanie Findlay, Neil Reid, Nicholas Aebischer, Nick Tomlinson, Pat Morris, Paul Chanin, Peter Lurz, Richard Yarnell, Richard Lawrence, Ro Scott, Rob Coope, Roger Trout, Roisin Campbell-Palmer, Roo Campbell, Sally Humphreys, Sam Berry, Scott Newey, Fiona Parker, Steve Adams, Steve Parker and Tom Fairfield. We would like to make special mention of Tony Mitchell-Jones, Henry Schofield, Anita Glover, Steve Carter and Paul Racey for their careful and constructive insights: they have considerably improved this document.

We would like to thank the very many volunteer surveyors and ecological consultants who contributed the data that allowed us to produce the distribution maps. We are also grateful for the support of local Biological Records Centres and other organisations in sharing their datasets; our task would have been very much harder without the tremendous contribution of county recorders and verifiers. Tom Hunt at the Association of Local Records Centres provided valuable assistance in ensuring the smooth transition of data, and we are also grateful to the National Biodiversity Network Trust. Specific records were supplied by The Bat Conservation Trust, Baseline Ecology, Bedfordshire and Luton Biodiversity Recording and Monitoring Centre, Biodiversity Gatwick Project, Biodiversity Information Service for Powys and Brecon Beacons National Park, Biological Records Centre, Boat of Garten Wildlife Group, Bristol Regional Environmental Records Centre, British Deer Society, British Trust for Ornithology, Buckinghamshire and Milton Keynes Environmental Records Centre, Cambridgeshire & Peterborough Environmental Records Centre, Canal & River, Central Scotland Green Network Trust, Cofnod (North Wales Environmental Information Service), Cumbria Biodiversity Data Centre, Derbyshire Biological Records Centre, Derbyshire Mammal Atlas, Derbyshire Mammal Group, Devon Biodiversity Records Centre, Doncaster MBC Biological Records Centre, Dorset Environmental Records Centre, Dorset Mammal Group, Dr Francis Rose Field Notebook Project, Dumfries and Galloway Environmental Resources Centre, East Ayrshire Countryside Ranger Service, EcoRecord, Environmental Agency, Environmental Records Centre for Cornwall and the Isles of Scilly, Environmental Records Information Centre North East, Essex Wildlife Trust, Fife Nature Records Centre, Focus Ecology Ltd, Furesfen Ecological Consultancy, Glasgow Museums BRC, Gloucestershire Centre for Environmental Records, Greater Lincolnshire Nature Partnership, Greater Manchester Ecology Unit, Greenspace Information for Greater London (GiGL), Hampshire Biodiversity Information Centre, Herefordshire Biological Records Centre, Hertfordshire Natural History Society, Herts Environmental Records Centre, Highland Biological Recording Group, Humber Environmental Data Centre, Inner Forth Nature Counts, iRecord, Isle of Wight Local Records Centre, iSpot, IW Council Parks and Countryside Section, John Muir Trust, Kent & Medway Biological Records Centre, Kent Biological Records Centre, Lancashire Environment Record Network, Leicestershire and Rutland Environmental Records Centre, Lorn Natural History Group, Lothian and Borders Mammal Group, Lymington Naturalists, Mammals of Suffolk, MAUWSE project, Merseyside BioBank, Ministry of Justice, MKA Ecology Ltd, National Trust for Scotland, National Trust, Natural Resources Wales, Nonsuch Watch, Norfolk Biodiversity Information Service, North & East Yorkshire Ecological Data Centre, North Ayrshire Countryside Ranger Service, North East Scotland Biological Records Centre, Northamptonshire Biodiversity Records Centre, Nottinghamshire Mammal Database, Outer Hebrides Biological Recording Project, People's

Trust for Endangered Species, Phlorum Consultancy, PJC Consultancy, Powys and BBNP Biodiversity Information Service, Riverbank Wildlife Area, Rotherham Biological Records Centre, Royal Horticultural Society, RPS, RSPCA, SCC Open Space, Scottish Natural Heritage, Scottish Wildlife Trust, Sheffield Biological Records Centre, Shetland Biological Records Centre, Shire Group of Internal Drainage Boards, Shropshire Ecological Data Network, Somerset Environmental Records Centre, Sorby Mammal Group, Sorby Natural History Society, South East Wales Biodiversity Records Centre, St Helens Wildlife Recording Group, Staffordshire Ecological Record, Suffolk Biological Records Centre, Surrey Biodiversity Information Centre, Surrey Biological Records Centre, Surrey Dormouse Group, Surrey Mammal Group, Sussex Biodiversity Record Centre, Sustrans, Tawny Croft Wildlife Consultants, Thames Valley Environmental Records Centre, The Biodiversity Information System for Cheshire, Halton, Warrington and the Wirral, The Ecology Consultancy, The Highland Biological Recording Group (HBRG), The Magnificent Science Company Limited, The Vincent Wildlife Trust, The Wildlife Information Centre, Unsted Wildlife Monitoring, Warwickshire Biological Records Centre, West Wales Biodiversity Information Centre, West Yorkshire Ecology Service, WildWatch Project Records, Wiltshire and Swindon Biological Records Centre, Worcestershire Biological Records Centre, Yorkshire Naturalists' Union, Yorkshire Wildlife Trust, Stephanie Tyler, Steve Lonsdale, Eve Mulholland, Derek Crawley, Helen Butler, A Blunden, Annie Haycock, Antony Witts, I Boyd, Brenda Mayles, Brian Ribbands, Anne Marston, Chris Matcham, Dave and Joyce Earl, David Jardine, Debbie Wallace, Declan Barraclough, Ed Pooley, Graham Scholey, Harriet James, Ian Bond, John Durkin, Jenny Jones, John Dobson, Johnny Birks, Ken Walton, Kirstie & Calum Ross, Penny Lewns, Preston Montford Field Studies Council Centre, C. R. Pope, Paul Chanin, Paul Seligman, Michael Walker, Peter Follett, Phil Richardson, Rob Spencer, Robert and Val Clinging, Robert Lamb, Rosy Jones, Tracy Underwood, Zoe Haysted, and G. Knass.

Finally, we would like to thank our families, friends, and numerous others for their encouragement during what — at times — felt like it would be a never-ending task. This project was only possible thanks to their support.

1 Introduction

The status of terrestrial mammal populations in Britain was last comprehensively reviewed over 20 years ago (Arnold, 1993; Harris et al., 1995). Yet mammals are key components of practically every terrestrial ecosystem in Britain. Small mammals are prey for mammalian predators and avian raptors (Norrdahl and Korpimäki, 1995) and their burrows create nesting sites for bumblebees (Kells and Goulson, 2003). Ungulates shape the landscape through grazing (Palmer et al., 2003), and bats and small mammals can indicate the health of an ecosystem (Pearce and Venier, 2005; Jones et al., 2009). An up-to-date review of the population and conservation status of British mammals, and an assessment of their likely future prospects under changing environmental conditions, is, therefore, long overdue. Whilst there have been, of course, focused studies on particular taxa conducted since the previous review, these do not deliver the broad overview of the relative status of different species necessary for the prioritisation of investment and practical action. This project not only summarises the available evidence derived from hundreds of different research projects, but it also uses — as far as possible — consistent methodologies across species, and is transparent about the approaches and assumptions that have been applied. Using more than 1.5 million individual biological records, this review presents the best available estimates of population size, geographical range, status trends and threats.

Many British mammals are intensively managed. The objective of this management may be to reduce damage to agricultural crops, forestry or other wildlife by species considered to be pests, such as rodents (Labuschagne et al., 2016) and deer (Trenkel et al., 1998), or to prevent the transmission of disease to livestock or humans (Gortázar et al., 2012). Conversely, conservation measures may be required to halt or reverse the decline of a threatened species. These may include protecting species such as bats and the water vole, *Arvicola amphibious,* from the impacts of built development and agricultural change (Roos et al., 2012); preserving the genetic integrity of native species threatened by hybridisation, such as the Wildcat *Felis silvestris* or the red deer *Cervus elaphus*; or helping to secure the long-term future of small populations of pine marten *Martes martes* or red squirrel *Sciurus vulgaris* through translocation. Unfortunately, the evidence base for management is frequently poor. This review highlights species of concern, identifies current and likely future threats, and also explicitly states where conclusions are limited by the lack of sound information.

On a global scale, the beaver is the only British mammal is considered at imminent threat of extinction ('Threatened' under IUCN Red List Criteria (IUCN, 2001)), though the otter *Lutra lutra*, Bechstein's bat *Myotis bechsteinii,* and barbastelle bat *Barbastella barbastellus*, are considered Near Threatened. Perhaps surprisingly, Great Britain is also a stronghold for the non-native Chinese water deer *Hydropotes inermis*, which is increasingly threatened in its native habitat. Despite this apparent lack of threat to the mammals found in Britain in a global context, the state of our wildlife is clearly important ecologically, culturally and morally, and responsibilities towards it are enshrined in national and international law. To inform future planning and conservation action, we have therefore produced a Regional Red List for British mammals using IUCN criteria. This work is presented separately, but the summary information is included within this review. Red Lists are designed to highlight imminent risks of extinction and short-term changes in conservation status ('short-term' being defined as 10 years or 3 generations – whichever is the greater), but they do not account for historical depletion of populations over the longer term. Yet, clearly, long-term change is also of concern, even where population sizes and geographical ranges are currently stable. This review therefore also assesses change since the last comprehensive assessment of geographical range (Arnold, 1993) and population status (Harris et al., 1995) more than 20 years ago.

Accurate information on the distribution and population density of mammal species, as well as insight into the temporal trends, is vital for the development of effective management (Gibbs et al., 1999; Collen et al., 2013). These data are also required to enable the UK to fulfil its international reporting obligations, for example under Article 17 of the Habitats Directive. Even post-Brexit, there will be obligations for monitoring and reporting under domestic law and international treaty (such as the Bern Convention). The amount of detail required depends on the intended scale at which the data will be used (Dickinson et al., 2010). Whilst species may show wide variation locally, a robust estimate of the national status of a species is important for conservation planning.

National surveys tend to be limited to the identification of a species' distribution or, sometimes, where effort is standardised, they identify the percentage of occupied habitat within a species' range. Surveys of this nature have been carried out for several mammalian species, including the otter (Strachan, 2007; Crawford, 2010; Strachan, 2015), the polecat *Mustela putorius* (Birks and Kitchener, 1999) and the pine marten (Croose et al., 2013; Croose et al., 2014). These surveys give a useful snapshot of the status of a species' distribution, and they also provide a baseline from which to measure change (Lindenmayer and Likens, 2010). When consecutive surveys are conducted, it is possible to measure temporal trends in a species' distribution or, where counts are taken, monitor changes in relative population size or occurrence. Schemes such as the National Dormouse Monitoring Programme (NDMP) have provided monitoring over considerable periods (the NDMP has run since 1988), and otters have been the subject of repeated country-wide surveys (e.g. Crawford, 2003; Strachan, 2007; Crawford, 2010; Strachan, 2015) that have allowed for the identification of temporal changes in distribution. Badgers *Meles meles* are monitored via counts of the number of badger setts (Cresswell et al., 1990; Wilson et al., 1997; Judge et al., 2014), and bat species via a combination of acoustic field survey data, hibernation surveys and roost counts (National Bat Monitoring Programme). These permit inferences to be made about relative changes in population size. However, absolute measures of population size would require additional data on the number of badgers per social group, and density of bat maternity roosts, respectively.

Absolute measures of population size rely on estimates of population density or total counts in a range of habitat types. The obtaining of density information requires a larger investment of time and effort than the determining of presence alone, and often necessitates repeated surveys. The effort needed to determine density is higher still where species are elusive or rare (Zylstra et al., 2010; Hui et al., 2011). These limitations mean that the scarce resources available for monitoring have tended to be directed at measuring changes in distribution or relative population size, rather than estimating absolute population sizes. However, there are purposes for which absolute population sizes rather than indices of relative change are extremely important. For example, determining whether wildlife fatalities (such as from collision with vehicles or wind turbines, or from culling) are likely to have a material impact on local populations depends on having reasonable estimates of population sizes. The prioritisation of conservation and management actions also often requires an understanding of population sizes: two of the three IUCN Red List Criteria, for instance, are based on knowing the number of mature individuals.

Where they occur, assessments of population size or density are often smaller in scale than distribution surveys, taking place at a single site or a small number of locations rather than at a regional, landscape or national scale. Limitations in the resources available, as well as the behaviour of the species under study, often dictate the methods employed. For easily observed animals, surveys can be carried out using direct observation. These surveys may take the form of direct attempts to count the local population (e.g. for deer species; Putman et al., 2011). Alternatively, inferences can be drawn from observing subsets of the population: refinements such as distance sampling — which adjusts for the declining probability of detection with increasing distance from the observer — help to improve the quality of such estimates (Borchers et al., 2015) (e.g. for mountain hare Lepus timidus; Knipe et al., 2013). The new Mammal Mapper app. for mobile phones, provided free of charge by the Mammal Society, is designed to encourage large-scale citizen science participation in distance sampling surveys and to improve the quality of data available on habitat-specific mammal densities.

For small mammals, measures of population density tend to involve live-trapping. The total number of animals trapped during a single trapping session can provide a proxy for population density in the form of the total of individuals trapped, or 'minimum number alive' (MNA), when divided by an estimate of the spatial extent of the trappable population (the 'effective trapping area'). This area is, however, difficult to define and leads to density estimates which are often considered unreliable (Efford, 2004). Where it is possible to conduct multiple trapping sessions, capture-mark-recapture models can be used to estimate density from the capture history of individuals within the trapped population, although multiple surveys inevitably require more time and effort, and certain assumptions about the population and capture process must be met (see Amstrup et al., 2005). Where species are rare or elusive, indirect methods of detection may be required (Pereira et al., 2010). Camera trapping (e.g. for wildcats; Hetherington and Campbell, 2012; Kilshaw et al., 2015), or counts of non-invasively collected samples such as faeces (Jarman and Caparano, 1997; Gormley et al., 2011) negate the need to observe or disturb the target animals. Aside from the notable exception where animal behaviour conforms to strict model assumptions (Rowcliffe et al., 2008), individual identification is required for absolute measures of population density. Recent advances in molecular techniques have enabled individual identification from non-invasive sources such as hair, faeces and feathers (Waits and Paetkau, 2005), although the costs required to process such samples are still relatively high and processing the samples is time-consuming. In summary, despite substantial effort to detect relative trends in distribution for some species such as otters (e.g. Crawford, 2003; Strachan, 2007; Crawford, 2010; Strachan, 2015), the investment of time, effort and funding for large-scale assessments of the status and trends for mammalian species means that such assessments are few and far between in Britain.

Battersby (2005) reviewed the monitoring schemes in place under the Tracking Mammals Partnership, as well as those proposed, and reported the available trends in relative abundance and distribution. That review did not, however, provide updated estimates of population size. Reliance has therefore been placed on the assessment of population size, status and trends undertaken in the mid-1990s by Harris et al. (1995). For that assessment, population density estimates for all terrestrial mammals in Britain were taken from peer-reviewed literature and expert opinion. Population sizes were then estimated by combining habitat-specific density estimates with the area of available habitat within the species' distribution. Habitat data were derived from the land classes devised by the Centre for Ecology and Hydrology (formerly the Institute of Terrestrial Ecology) (Bunce et al., 1981a; Bunce et al., 1981b; Bunce et al., 1996), and distributions were based on data deposited up to March 1992 which were reported in the Mammal Atlas (Arnold, 1993). Information on population trends for different mammal species were quoted where the information was available in the literature, or were derived through expert knowledge; current distribution and the legal status of each species were also described. We have followed broadly the same approach in the current review.

Recently, Croft et al. (2017) have tested an approach based on habitat suitability modelling to produce density estimates for mammals in Great Britain. In a first step, habitat suitability models were built using environmental data coupled with occurrence information derived from the National Biodiversity Network (NBN). These habitat suitability scores were then linked with habitat-specific density estimates to produce an abundance estimate. The current report and Croft et al. (2017) shared the requirement for habitat-specific density data, and therefore suffer the same constraint that for many species the required information is simply not available. This problem is particularly acute for bats, and so, in this review, an alternative approach that did not depend on habitat-specific information was deployed. Both studies also suffer from a lack of information about the level of occupancy in a given habitat across a species' range: for very widespread species such as the field vole *Microtus agrestis*, this may not be a major issue; for more patchily distributed species, such as the red deer, it could introduce important errors.

However, the approach used by Croft et al. (2017) differs from the present review in several important respects:

- We had access to data at a very much greater spatial resolution, whereas the data used to derive the habitat suitability models in Croft et al. (2017) were available only at 1km or 10km resolution.
- We were able to include datasets not available via the NBN.
- Our assessments were based on pre-breeding density data only, to ensure comparability with the Harris review.
- The present review used a rigorous process of data cleaning, based on the input of many recognised experts to generate a smoothed distribution map of distribution. The

habitat suitability models in Croft et al. (2017), in contrast, are based on the data as presented in NBN.

There are strengths and weaknesses to both approaches. The advantage of habitat suitability modelling is that it can allow extrapolation to poorly surveyed areas on the basis of other environmental variables (temperature, habitat, etc.). However, sensible models can only be built where there are sufficient data to parameterise them; and for many British mammals the evidence is lacking. Also, the spatial resolution of publicly available data is a major constraint, particularly where species are unlikely to be uniformly distributed across a grid square because of particular habitat requirements. The figures presented in the two reports are not directly comparable. Nevertheless, it is reassuring that for many species the population size estimates are within the same order of magnitude.

Population status is a dynamic quantity, and many mammals in Britain are thought to be in decline (e.g., the hedgehog *Erinaceus europeaus* and the red squirrel (Roos et al., 2012; Gurnell et al., 2015b)), whereas several others are apparently increasing in population size and range (e.g., the pine marten and the polecat (Croose et al., 2014; Croose, 2016)). Population assessments from Harris et al. (1995) are, therefore, unlikely to reflect the current status of British mammals, but are the most recent reference for most mammals and are still quoted in the current literature. In this review, we provide the following:

- A current distribution map for each species, using presence data from 1995 to 2016, smoothed using an alpha-hull approach.
- An assessment of current conservation status.
- Estimates of habitat-specific population density from the recent literature (1995-2016) or from expert opinion.
- Current estimates of population size for England, Scotland, and Wales, and the total for Great Britain.
- A critique of these estimates, with a review of data deficiencies.
- A review of the temporal trends in population size since the last review in 1995 (which was based on data collected up to 1992), and the drivers leading to the observed trends.
- A review of the future prospects of each species (see Appendix 7).
- Regional Red List statuses for Great Britain, England, Scotland and Wales that have been formally approved by the Inter-Agency IUCN Red Listing Group.

The review presents the most up-to-date assessment of population size and status for the 58 terrestrial mammal species resident in Britain. Accounts are provided in Appendix 6 for a further 9 species. These are either island sub-species (the Skomer vole), feral animals (the feral ferret, sheep and goat), vagrant visitors to Britain (the parti-coloured bat and Kuhl's pipistrelle), or are only present as managed populations or occasional individuals (the reindeer, wallaby and raccoon). Further, it identifies key areas for further research where the data required to assess population size accurately are lacking, and highlights the future prospects of each species and urgent requirements for conservation action.

2 Methods

2.1 Literature search

Literature was sourced using the databases ISI Web of Knowledge and Google Scholar. Search terms were the species' taxonomic name and/or the species' common name, as well as at least one of the terms (including wildcards) from two lists, where list one included the terms British, UK, England, Scotland and Wales, and list two included the terms 'population density', 'population estimate', 'abundance', 'population size', 'survey' or 'census'. As data published earlier than 1995 would have been incorporated into the review by Harris et al. (1995), the search was primarily limited to publications issued between 1995 and 2015, although references outside of these years were not excluded if the content was particularly relevant or recommended by an expert in the field. Peer-reviewed papers were screened by first reading the title, then the abstract where relevance was uncertain. The references and citations of each relevant paper were checked and sourced where applicable. Government and Non-Governmental Organisation (NGO) reports were sourced directly from the following organisations: Natural England, Natural Resources Wales (NRW), Scottish Natural Heritage (SNH), and the People's Trust for Endangered Species (PTES). Student postgraduate theses were used when identified via cross-referencing or recommended by an expert in the field, but were not specifically sought because of the difficulty of obtaining copies, particularly for older documents, and the inconsistency with which these documents are catalogued.

The following details were recorded from each paper/report: estimate type (i.e., minimum number alive, absolute population size/density); survey method; area or length (for linear features) of the study site; habitat type; start date; time of year; and duration of study.

2.2 Habitat data

To quantify habitat availability, data were taken from the 2007 Countryside Survey (CS2007; (Carey et al., 2008). The area of each broad habitat class (hereafter 'broad habitat') within each species' distribution in England, Scotland and Wales (see section 2.5) was extracted from the Land Cover Map (LCM2007) land-use layer (Morton et al., 2011) using ArcGIS (version 10.3).

The LCM2007 division of grasslands is difficult to use in the prediction of mammal densities, since it is the structure of the habitat, rather than species composition or underlying soil-type, that is a primary driver of its suitability as a habitat. Although 'Improved Grassland' is an LCM class, there is no LCM class of 'Semi-Improved' (i.e., mainly managed for pasture, silage or hay (Jackson, 2000)), and semi-improved grasslands will often be classified as 'Improved Grassland'. The divisions of 'Neutral', 'Calcareous' and 'Acid' grasslands are derived by re-assigning cells in the 'Rough Grassland' class on the basis of soil type, and include a continuum from unimproved to semi-improved grasslands. The grassland that retains the original classification of 'Rough Grassland' is therefore a mix of managed, low productivity grassland, plus some areas of semi-natural grassland, which could not be assigned 'Neutral', 'Calcareous' or 'Acid' grassland with confidence in the LCM2007. For the purpose of this report, 'Rough Grassland' is considered equivalent to unimproved grassland.

Given the widespread decline in rough grazing (Connors, 2016), and the rarity of unimproved neutral grassland in the landscape, it is considered appropriate to conclude that most 'Neutral', 'Calcareous' and 'Acid' grasslands are at least semi-improved. Indeed, these same category names are considered 'Semi-Improved' in the Countryside Survey 2007 (which uses botanical characteristics identified through field study rather than remotely-sensed data) (Carey et al., 2008). It is recognised that for 'Acid' and 'Calcareous' grasslands, there may be a larger component of unimproved land than for 'Neutral' grasslands (especially large areas of acid grassland in Scotland). Nevertheless, the dominant management strategies with relatively high stocking densities have resulted in 'smoothed grassland', lacking the structural complexity required to support high density mammal populations found in rough grassland. These categories have, therefore, been aggregated with 'Improved Grassland' on the basis of functional similarity to mammalian fauna.

The total length of hedgerows in each country was taken from the Countryside Survey 2007 linear features estimates (Bunce et al., 1996; Scott, 2007; Carey et al., 2008). Hedgerows are of variable value to mammals; for example, hedgerows under agri-environment scheme management (AES) contain higher densities of bank voles *Myodes glareolus* (Kotzageorgis and Mason, 1997) (Shore et al., 2005; Broughton et al., 2014) and field voles (Broughton et al., 2014), than those under non-AES management (see Tables 7.3a and 7.4a). The proportion of hedgerows under AES management (hereafter AES hedgerows) was therefore quantified for each country using data supplied by Natural England, Natural Resources Wales and Scottish Natural Heritage. As the length of AES hedgerows was available as a total value per country, and not a GIS layer, they were assumed to be evenly distributed throughout each country, and the proportion of AES hedgerows was used to divide the total length of hedgerows within each species' distribution into lengths for non-AES and AES hedgerows. In reality, AES hedgerows are unlikely to be evenly distributed throughout each country, so this assumption will probably have resulted in errors for species that do not occupy the whole country.

The total length of riparian habitats in each country was taken from Table 4 in Harris et al. (1995). The length of riparian habitats within each species' distribution was calculated by multiplying the total length by the percentage of the country included in the species' distribution. Waterways are not, however, evenly distributed throughout the country, and so this method is likely to have resulted in inaccurate lengths of riparian habitat for species that do not occupy the whole of Great Britain.

2.3 Habitat comparison

Harris et al. (1995) used two measures of habitat availability. These were: (a) the land classes devised by the Institute of Terrestrial Ecology (Bunce et al., 1981aa; Bunce et al., 1981bb); and (b) habitat types as described in Cresswell et al. (1990). As the Cresswell et al. (1990) habitat types are more comparable to the LCM2007 data used in this study than are current land classes, we have compared habitat availability using the data provided in table 3 of Harris et al. (1995) rather than the land class data. The Cresswell et al. (1990) habitat data used in Harris et al. (1995) were collected during field surveys of 2455 x 1km squares in the period 1985-1988 (Cresswell et al., 1990; see Harris et al. 1995 for further details).

In the current analysis, habitat types from Harris et al. (1995) were matched to each broad habitat in the LCM2007 dataset; land areas were summed where more than one habitat type fell within a broad habitat. Habitat sub-categories from LCM2007 were also matched to habitat types from Harris et al. (1995) for reference (see Appendix 1). The difference in the area of each habitat between the two datasets was then assessed as:

$$\textit{Change } (km^2) = \textit{LCM2007}(km^2) - \textit{Harris 1995 } (km^2)$$

and

$$\textit{\% Change} = (\textit{Change } (km^2)/\textit{Harris 1995 } (km^2))*100$$

Unfortunately, a direct comparison between habitat data from the LCM1990, 2000 and 2007 datasets to clarify the real changes in habitat availability is not advised because there are differences between the datasets as a result of differently-sourced satellite data and updated methodology (Clare Rowland, CEH, *pers. comm*.). Instead, to assess whether changes in the area of habitats between Harris et al. (1995) and LCM2007 (i.e., the current analysis) reflect real change over that time period, we assessed the changes between the Countryside Survey data for 1990 and 2007 (CS1990 and CS2007) (Carey et al., 2008). This dataset represents the most credible source of data to assess changes in habitat area across this time period, based on consistency of data collection methods between years (Lisa Norton, CEH, *pers. comm*.). It was not possible to use the CS2007 data directly in the estimates of population size for this review because the field survey data are derived from a sample of representative 1km^2 squares and spatial data were not available: therefore, information could not be matched to the entire range of any species. For the analysis of temporal trends in habitat, CS1990 was selected as the baseline year, being the closest to the field survey dates of 1985-1988 from Harris et al. (1995).

The LCM2007 and CS2007 datasets are considered to be approximately 80% accurate, although some discrepancies exist between the two. For a full assessment of correspondence between habitat classifications in the two datasets, see Morton et al. (2011).

2.4 Status

The conservation status of each species is presented within each species' account. In addition to the global conservation status provided by the IUCN Red List of Threatened Species, a Red List status at the British level and for constituent nations is provided for each species.

A national Red Data Book for mammals was first produced in 1993 (Morris, 1993). In addition to a statement about the conservation status of each species, the Red Data Book contained the legal status, distribution, population size, perceived threats and future actions for 18 British mammals, which provided a basis for setting conservation priorities. The statement on conservation status was based on expert opinion, as an appropriate classification system had not yet been developed. Since the publication of the Red Data Book, a more quantitative approach to threat assessment has been produced by the IUCN (2001), which has been used to help assess the current status of the 58 species in this report.

The status section of the current review provides the global and regional species listing on the IUCN Red List of Threatened Species. In addition, the national conservation status as assessed for Article 17 of the EU Habitats Directive, is shown where relevant. (For an overview of the assessment process, see http://jncc.defra.gov.uk/page-4096). Each species is indicated as being native, non-native or naturalised. Species are considered naturalised if they were introduced in or before the 12th century. If they have been present in Great Britain since before this time and their presence was not dependent on the actions of humans, then species are considered native.

Under the IUCN Red List criteria, each species is allocated to one of the following categories, relating to imminent risk of extinction:

- Critically Endangered (CR).
- Endangered (EN).
- Vulnerable (VU).
- Near Threatened (NT).
- Least Concern (LC).
- Data Deficient (DD).

The categories CR, EN and VU indicate an appreciable risk of extinction in the next decade, and are collectively described as ‘Threatened’: CR indicates the highest level of extinction risk in the wild, and EN and VU indicate progressively lower levels of risk. Near Threatened indicates that the species is close to qualifying as threatened, or is likely to qualify as threatened in the near future.

The IUCN classification system evaluates the risk of extinction against 5 different criteria. To ensure transparency and comparability across species, these criteria were assessed using standardised methods (see http://www.nationalredlist.org/home/about/ for further information on Regional Red Listing methodology). The 5 criteria are intended to be as independent as possible and to act cumulatively: it therefore follows that well-studied species, which can be assessed against all 5 criteria, are more likely to qualify as threatened than less well-studied ones which may only be assessed against one criterion (often, geographical range, which can be slow to show change even when there are significant population declines). This is likely to explain why a lower proportion of Britain’s native mammals (20%) are considered threatened compared with 39% of birds (Stanbury et al., 2017). A species may not be defined as Data Deficient unless there are no reasonable grounds for making an assessment against *any* of the 5 criteria. It should be noted that species cannot be considered extinct in the wild until exhaustive surveys have failed to reveal evidence of a single individual. Information on generation times (used in assessments under Criteria A and C) was based on standardised information for mammals (see http://www.iucnredlist.org/technical-documents/red-list-training/red-list-guidance-docs) provided to Natural England at an IUCN Red List Assessor Training Workshop in 2017.

The assessments of Regional Red List statuses for Great Britain and each country have been formally approved by the Inter-Agency IUCN Red Listing Group. Assessments conducted for non-native (naturalised) species also followed the same IUCN Regional Red List criteria, but there is no mechanism for these to be formally approved, except in the case of Ornkey vole (because it is officially considered a sub-species) and lesser white-toothed shrew (where there is uncertainty about whether it is naturalised or native). Country-level assessments are presented in square brackets in this report, and the assessments for non-native (naturalised) species are reported separately by the Mammal Society (see www.mammal.org.uk/science-research/population-review-red-list where full details of the rationale for each listing are also provided).

Information on legislation relating to each species is readily available elsewhere, and is not, therefore, outlined in this report. See the JNCC website for UK legislation (http://jncc.defra.gov.uk/page-1376), and for European legislation see http://jncc.defra.gov.uk/page-1372. The legal framework for Scotland can be found at http://www.snh.gov.uk/protecting-scotlands-nature/protected-species/legal-framework/. For further information on species legislation in Wales, see https://naturalresources.wales/conservation-biodiversity-and-wildlife/?lang=en, and for England, see https://www.gov.uk/topic/planning-development/protected-sites-species.

2.5 Species' distribution maps

Presence data collected between 1995 and 2016 at 10km resolution or higher were gathered from the NBN gateway, local record centres, national and local monitoring schemes and iRecord for each species (see Acknowledgements). In total, 1,678,548 records were included. The start date was chosen to provide continuity with Harris et al. (1995). For regions where recording effort is low (such as some parts of Scotland), or when the species is under-recorded (such as the house mouse *Mus musculus*), this approach may have generated some artefactual gaps in the distribution that could have been filled by including older records. However, given the variability in the spatial resolution and quality of some historical datasets, the difficulties in obtaining consistent access to older records across Great Britain, and the uncertainties in whether there have been true range shifts for many species, we have chosen to use a consistent approach across all species and to highlight potential difficulties where they arise.

Only data that had been verified by the source organisation were included in the distribution maps. The British Trust for Ornithology (BTO) provided a valuable additional source of unverified presence data. As mammal identification is not the primary objective of BTO surveys, data were included from this source, alongside the verified records, for those species which are unlikely to be misidentified, namely moles *Talpa europaea*, rabbits *Oryctolagus cuniculus*, badgers, foxes *Vulpes vulpes* and hedgehogs. Despite the reliance on verified data, erroneous records remained for some species, particularly those which are difficult to identify. Experts on each species were consulted to ensure that the maps represented current species' distributions as accurately as possible. Experts were presented with maps at a 10km resolution, and asked to remove any squares in which they were certain that the species had not been recorded since 1995. Deletions were only accepted when two or more experts agreed. Where experts had presence data from an unpublished field survey or could provide the source of such a record, they were asked to add these data to the map.

Smoothed distribution maps were created by fitting alpha hulls to the presence data using the Alphahull package in R (Pateiro-López and Rodrıguez-Casal, 2010). The area enclosed within the alpha hull (also known as the extent of occurrence, EOO (IUCN, 2001)) for each species is shown in Appendix 2, and the smoothed maps are presented within the species' reports. The alpha hull is an algorithmic method of assigning a boundary around a set of discrete points. The alpha hull algorithm contains a parameter (that is, alpha) that determines the extent to which the hull extends outwards from the area(s) with the highest densities of points. As the value of alpha increases, the hull will extend to encompass increasingly isolated points, and unoccupied areas between, until reaching a point where the hull encompasses all of the points and approximates the Minimum Convex Polygon (MCP).

When using alpha hulls to determine distributions, there is no correct or ideal value of alpha. Rather, the choice of alpha depends on the purpose for which the distribution hull is to be used, as well as the quality, quantity and spread of the data itself. In general, there is a trade-off between falsely including areas of unoccupied and/or unsuitable habitat and incorrectly excluding areas that are actually occupied but were not sufficiently represented in the data (i.e., areas with isolated records). For example, despite there being at least one verified polecat record in Cornwall, this area is not represented on the distribution map, as the record is too far from the nearest records in Devon (Figure 8.8a). In this study, an alpha value of 20km was determined by the authors based on a series of test maps, to represent the best balance between the inclusion of unoccupied sites (i.e., where records are sparse, but close enough for inclusion) and the exclusion of occupied areas owing to gaps in the data (i.e., where records exist, but are too isolated for inclusion). An additional 10km buffer was also added to the final hull polygon to provide smoothing to the hull and to ensure that the final distribution covered all parts of recording squares from which positive records had been received. Given that the coarsest resolution of data included in our analysis was hectads (10km x 10km squares), the 10km buffer ensured that the entirety of each positive hectad at the periphery of the range was included.

Our method differs slightly from that used to create surface area maps for the Article 17 EU Habitats Directive species' assessments ((Joint Nature Conservation Committee, 2007), with modifications in the most recent report outlined in Joint Nature Conservation Committee (2013c)) which employed alpha shapes rather than alpha hulls. Alpha hulls and alpha shapes are closely related, as they adopt the same underlying process to select the points that form the range edge, differing only in the type of line used to connect these edge points (straight lines in an alpha shapes and concave arcs in alpha hulls; see Figure 2.5a). A second difference is that the distance between points likely to be recognised as a gap in the distribution is approximately 40km in this report (i.e., the distance across a circle of radius 20km; 'approximately' because the precise size depends on the local distribution of points, and the impact of the 10km buffer depends on the size of the occupied shapes). However, these technical differences are unlikely to make a material difference to the areas calculated for most species.

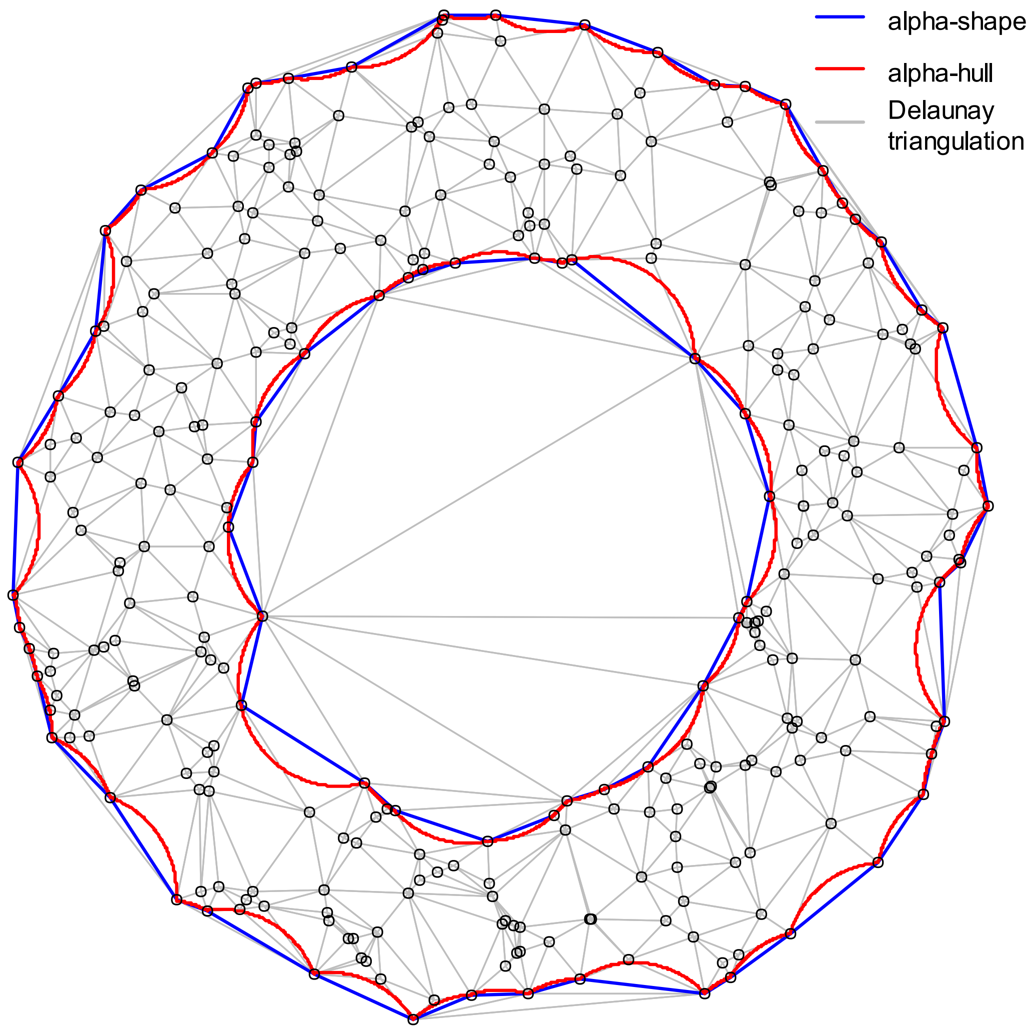


Figure 2.5a Schematic diagram comparing alpha-hull and alpha-shape approaches, with presence points shown as dots. With the alpha-shape method, the points determined to mark boundaries are joined by straight lines (blue line), whereas for the alpha-hull method the points are joined by a series of arcs (red line). It can be seen that the approaches will give a similar shape, with the alpha-shape being slightly flatter than the alpha-hull. The grey lines show the Delaunay triangulation which is the starting point of the alpha hull/shape algorithm. In general terms, gaps in the distribution using the alpha-hull approach will arise if a cluster of points is more than an unfilled circle away from the next set of points (c. 40km in this review because alpha (circle radius) is set to 20km; 45km in the JNCC reports (Joint Nature Conservation Committee, 2007)).

A more important difference is that the current review uses data at the finest spatial resolution available: for most records this was 100m x 100m or 1km x 1km, compared with a resolution of 10km x 10km (hectads) used in the Article 17 Reports (Joint Nature Conservation Committee, 2007). A key initial step in the algorithm to create alpha hulls/alpha shapes is to make a triangular tessellation of the data: adjacent points are connected to form ‘triangles’ so that there are no gaps or overlaps, in a process known as Delaunay triangulation (see Figure 2.5a). Lines in the Delaunay triangulation are then removed systematically where the radius of a circle that includes both the start and end points of the line on its circumference is larger than the given value of the parameter alpha. This means that the algorithm removes more of the lines, and becomes more restrictive, as the value of alpha decreases. Given the underlying triangular tessellation of the data in the algorithm, at least 3 points in proximity (depending on the value of alpha) are required for a hull, or hull fragment, to be produced.

It follows that when data are used at hectad-level resolution only, points towards the periphery of the range or in isolated areas are likely to be discarded because they will not form connections with two other points. In contrast, the use of high-resolution data means that if there are 3 or more records from an isolated area — provided that these are in close enough proximity (the distance depending on the value of alpha) — they will form a smaller isolated section of the hull. High-resolution data are also likely to result in a more realistic shape, since the alpha hull is generated from the original data points rather than from just the centre point of every occupied hectad. (The latter approach would mean that the same shape could be generated from datapoints 1km or 20km apart, depending on where the observations fall relative to the boundary grid.) The geographical ranges presented in this report are therefore slightly more accurate than the surface areas reported under Article 17.

Gaps in a species' distribution may occur because of a lack of submitted biological records, rather than because there is a true absence. This is particularly apparent in less densely populated areas and for species that, being reasonably common, do not generate sufficient interest to prompt recording. For example, there are often gaps in distributions in western and northern Scotland (see the distribution maps for stoat *Mustela erminea*, weasel *Mustela nivalis* and common shrew *Sorex araneus*).

The maps presented in this report should be viewed with the following limitations in mind:

- Areas that contain very isolated records may not have been included in the area of distribution.
- Gaps may represent low recorder effort rather than true absences.
- The maps do not reflect density: i.e., areas with dense records are not distinguished from areas with less dense records.
- All verified records, including those of occasional and transient individuals, are included. Therefore some areas may not represent an established, breeding population. This is a particular problem for more mobile species as ranges may be overestimated.

Where appropriate, the distribution maps have been presented alongside those published from other sources for comparison.

In each species' report, the area of suitable habitat is presented in a table, and the resulting population size is shown below it. The concept of suitability is very broadly applied: all habitats listed in the previous report (Harris et al., 1995) are included, together with any additional habitats highlighted by the literature review or expert opinion. In the case of species that use a mosaic of different habitat types, such as the red fox, all areas within the range (i.e., the extent of occurrence (EOO) as defined by the alpha hull) are treated as suitable. When species are reliant on a particular habitat, some other kinds of potentially suitable habitat are excluded in order to avoid double counting. For example, population sizes for many deer species are derived from densities in woodland even though the animals may temporarily also use surrounding habitats. There are some species where no information is available on a habitat that is known to be used, such as agricultural areas for the brown rat, so these are excluded from the suitable habitat calculation. In most cases, the values for suitable habitat and EOO are very similar, but for a few species there are important differences. The pine marten provides an extreme example: its EOO in Britain is 82,900km^2, whereas the area of suitable habitat is only 12,100km^2. For those species that have been assessed under the Article 17 Reports for the European Union (Joint Nature Conservation Committee, 2013b), the 'surface area' estimations —based on an alpha-shape method, as described above — from those reports are shown alongside the EOOs. The EOO for each species is provided in Appendix 2 to allow comparison.

2.6 Population size assessment

To enable a standardised assessment of population density per habitat type, the habitat type recorded in each study was matched to the most comparable broad habitat or linear feature. When the habitat did not match any broad habitat, the estimates were excluded.

Studies that reported population size were converted to population density by dividing by the study area. Where a study area was not provided, the estimate was excluded. When a study contained more than one estimate, for example by including replicate sites, all estimates were recorded separately. All density estimates were standardised to the number of animals per unit area. For reasons of presentation, the denominators for density vary between taxa. For smaller taxa (rodents and soricomorphs) these are per hectare and per 100m for linear features; and per square kilometre for more mobile taxa (the bats, lagomorphs, carnivores, ungulates, erinaceomorphs) and per 1km for linear features.[1]

For rodents and soricomorphs, when studies provided a capture rate per trap night (e.g. Kotzageorgis and Mason, 1997; Marsh and Harris, 2000; Shore et al., 2005; Moro and Gadal, 2007; Broughton et al., 2014), the number of animals per minimum of 80 trap-nights was taken as the 'minimum number alive' (MNA) and scaled to density per hectare using 'Estimate/study area (ha)'. As these studies do not identify an 'effective trapping area' (i.e., the area containing the home range of all trapped animals), the resulting estimates may be inflated.

To calculate the total population size for each broad habitat, the median value of population density per habitat type was used. Percentile bootstrapping with 10,000 resamples was conducted in order to calculate 95% confidence intervals for the median values using the 'boot' package (Canty and Ripley, 2012) in R v3.2.2 (R Core Team, 2015). Where only a single estimate was available, the confidence intervals from the original publication were used when available. Where no data were found in the literature from 1995 to 2015 for a particular broad habitat, estimates from the expert opinion assessment were used. If neither the literature nor expert opinion provided an estimate for a particular habitat, the estimate from Harris et al. (1995) was used (see Appendix 1 for habitat matching). These earlier estimates were frequently based on expert opinion, and cannot therefore be assumed to have been a good estimate of true density. However, it was considered preferable to use these values rather than to score them as missing, which would have meant that the habitats were excluded altogether from the population estimates. The median density estimates and 95% confidence intervals per broad habitat were then multiplied by the area of each respective broad habitat within the species' distribution, per country, then summed to provide a total estimate for each country and for Great Britain as a whole. Population size estimates for each species are provided within each species' report, as well as a summary table in Appendix 3.

For bats, a slightly different approach was required for most species because habitat-specific densities are not meaningful for animals that use the landscape on a broad scale. Instead, densities (bats km^{-2}) were generally computed by multiplying the typical maternity roost density in an average quality landscape by twice the typical number of adult females per roost. Lower plausible intervals (PIs) — which can be thought of as roughly equivalent to

---

[1] 1ha = 0.01km^2

lower 95% confidence intervals, though without the same statistical foundations — were derived by multiplying the plausible maternity roost density for poor habitat by twice the lowest plausible estimate of adult females per maternity roost in poor habitat. The upper plausible limit was calculated similarly, but this time employing the highest plausible estimate of bats per roost, proportion of females, and typical roost density in good habitat. The population size and plausible limits were then obtained by multiplying the density estimate by the area within the range. For the two horseshoe bat species, direct count data were available for maternity colonies. An estimate (with upper and lower plausible limits) was made of the number of females, based on plausible sex ratios, and this value was multiplied by two to give the total population size. Full details are provided within each species' account.

Where possible, population sizes were adjusted to account for the percentage of occupied habitat within the species' range. Occupancy data were only included where studies used standardised surveys and reported both presence and absence. Knowledge of the percentage of occupied habitat can have a significant impact on estimates of population size, yet the evidence is not available for most species under review. In the absence of data on percentage occupancy, 100% was assumed. Clearly, this value is not realistic for many species, and therefore the use of 100% occupancy will have resulted in an overestimation of population sizes. For example, the population of the hazel dormouse *Muscardinus avellanarius* in Britain is estimated to be 930,000 (95%CI = 389,000-2,640,000; see Table 7.4b). Population sizes are adjusted to reflect the area of occupied woodlands (34%) and hedgerows (35.5%). The occupancy value for woodlands was originally calculated from surveys of hazel scrub only, rather than all types of woodland, but in the absence of more thorough surveys, it was applied to all woodlands within the species' range. If population size had not been adjusted for occupancy, the population estimate would have been almost doubled and the confidence intervals would have been much wider (2.4 million (95%CI = 829,000-6,500,000)). Although it was potentially possible to estimate occupancy from expert opinion for a small number of species-habitat combinations, in most cases experts were unable to provide the relevant information. Therefore, rather than arbitrarily imposing different values for species with data gaps, the study opted for consistency and transparency by applying an assumption of 100% occupancy unless contrary evidence was available. Where applicable, this decision is highlighted by the reliability scores and the data deficiency section in the species' reports.

2.7 Expert opinion assessment

For some species, limited data exist in the recent literature on habitat-specific population density. To make use of unpublished data and the experience of experts in the field, a survey was developed and sent to a list of people considered experts on each species (see Appendix 4 for survey questions). Experts were provided with the median habitat-specific density estimates from the literature wherever they were available. They were asked to provide, with justification, alternative estimates if they disagreed with those supplied, and to provide estimates for any habitats with no available data. The likely upper and lower limits of density ranges, taking into account variation within habitat types, were also collected. The median values were then computed across all expert responses for each of these parameters (central estimate, upper limit, and lower limit).

Where density estimates were not available in the literature, those calculated from expert opinion were used. Where density estimates were not available from expert opinion either, those applied by Harris et al. (1995) were used. The source of the density estimate for each habitat is provided in the case of each species.

2.8 Reliability assessment and identification of temporal trends

For poorly-studied species, the population size estimate can be strongly influenced by a single density estimate if it is particularly extreme, or if the relevant habitat accounts for a high percentage of the total species' distribution. For example, the population density of common shrews in bog is estimated to be $12ha^{-1}$ ($0\text{-}35ha^{-1}$; see Table 5.2a). This density is based on data from one study and is at least twice the estimated density in other habitats. The use of the median density would imply that 32% of the estimated population is derived from this habitat type. However, given the very wide confidence intervals around the density estimate, there are also wide confidence intervals around the population estimate. It is therefore plausible that between 0% and 58% of the population is found in this habitat. To identify which data have the strongest influence on population size for each species, we carried out two assessments. First, we calculated the percentage of the total population found in each habitat, and then we identified which habitat-specific population sizes accounted for more than 25% of the total population size. We went on to assess whether these habitat types formed a high proportion of the geographical range (>25% total area), and whether they supported high population densities (which would mean that the habitat was important to the estimate even where it comprised <25% of the geographical range).

We performed a sensitivity analysis by re-calculating population size with stepwise deletion of individual density estimates from habitats which met the following conditions:

- The habitat contains >25% of the estimated population.
- Median population density is supported by fewer than 10 individual density estimates.

These revised population estimates are reported where they fall outside the confidence limits of the original estimate. When there is only one density estimate for an influential habitat type, this habitat has been flagged as a priority for further data collection.

Where density estimates were found in the literature and also provided by experts, a comparison was made between the population size calculated using the standard method (i.e., using density estimates from the literature) and a population size re-calculated using median expert opinion values in place of those from the literature. This comparison was made only under the following conditions:

- Confidence limits for median density estimates from the literature do not overlap with the upper and lower ranges provided by experts.
- Fewer than 10 separate density estimates were obtained from the literature.

A reliability score has been calculated for each habitat containing more than 25% of the species' distribution, or accounting for more than 25% of the total population size. These scores are based on the number of locations in which individual assessments of population density were conducted, on the sample size (number of individual density estimates contributing to the median), and on whether data on the percentage of occupied habitat were available (see Table 4.1d for an example). A higher score indicates a more reliable estimate. The values across each of these criteria were summed to give a score per habitat; and where more than one habitat was assessed, the mean of the different scores is presented. The choice of values given to each component in the scoring system, and the decision of how to combine these values, are to some extent arbitrary: the absolute value of the score therefore has no inherent meaning. In addition, no consideration is given to the differing scientific quality or precision of the estimates provided in the original studies. Nevertheless, the scores can be used as a rough index for ranking reliability across different species, and are also helpful in highlighting data deficiencies. Final scores are colour-coded under a 'traffic light' system to indicate reliability as follows: 0 to 1 = red; 2 to 3 = orange; >3 = green, where a 'green' score is the most reliable. Harris et al. (1995) used a reliability scoring system of 1-5, where 1 was considered the most reliable, but the basis for the reliability assessment was not explicitly described. It is important to recognise that in the case of some species, data were completely lacking for habitats known to be used. For example, estimates for brown rats were based on dwellings and farms only because no evidence was available on riparian habitats, sewers, farm ditches, etc.; similarly it is known that the highest pine marten densities occur in landscapes with 20%-35% forest cover, but no information was available for habitats other than woodland. Given that it was impossible to know the extent to which these habitats contributed to the population size or distribution, they could not be included in the reliability scores. However, if these habitats do contribute substantially to the population, as is likely, then this will be an important source of error.

For bats, different methods were used to estimate population sizes, which are outlined in each species' account. Generally, the estimate of population size is based on the multiplication of the following three parameters: i) the median maternity roost size; ii) the ratio of males to females within maternity roosts; and iii) the number of maternity roosts per unit area of average-quality habitat (note that there was frequently only a single estimate of roost density available — species-specific details are provided within the individual reports). Unlike other taxa where habitat-specific density estimates could be used in the derivation of population estimates, for bats this approach was not possible except for a few species with high woodland dependency (see individual species' reports for details). This is partly because of a lack of data for most habitat-species combinations, but also because the importance of a given habitat (such as built environments) can be highly dependent on the composition of the surrounding mosaic of habitats. Therefore, maternity roost density estimates were instead made for $10km^2$ blocks of 'typical' landscape encompassing multiple habitats, with experts providing estimates for different parts of the country and for areas they considered to be of 'average', 'poor' and 'good' quality for bats. Reliability scores for bats are based on the number of maternity roosts for which a count was available, the number of roost density estimates available (including those from consultation with experts), and the availability of sex-ratio data. Scores were colour-coded in the same way as outlined for other taxa, but differences in the criteria used to score reliability mean that these scores are not directly comparable.

The population estimate is shown in brackets where the reliability score was <=1, where the upper confidence limit for the British population was more than 5 times larger than the central estimate, or where it was not possible to compute confidence intervals (except for the beaver, where total counts are assumed to account for most of the population), to highlight the uncertainty.

To prioritise for future research, data deficiencies were highlighted in the case of each species (see summary in Appendix 5) using the following categories:

- Density estimates do not represent within-habitat variability. A species was classified as data deficient if only one density estimate, from either the literature or expert opinion, existed for an occupied habitat and/or no density range or confidence intervals were given. Moreover, in cases where species' calculations were made across multiple habitats and it was not possible to take into account differing densities, the species was also classed as data deficient.
- The most recent density estimates for a particular habitat are more than 10 years old.
- Only limited density estimates are available for a key habitat (i.e., fewer than 10).
- Populations are managed and this is not taken into account in the population size estimate.
- Populations experience multi-annual cycles which make a single population size estimate uninformative.
- No density estimates are available for the specified habitat.
- No occupancy data are available for the specified habitat.
- Population sizes are based on total counts in some locations rather than density estimates.

To identify changes in population size over time, our estimates were compared to those from Harris et al. (1995), and to any others reported in the literature, where the estimation methods were comparable. Trends in range size were identified by changes in the number of occupied hectads in the new Mammal Atlas period (1995-2016)[2], with those in the last Mammal Atlas period (1960-1992; (Arnold, 1993)). No smoothed ranges were available from the previous review, and small changes in the numbers of hectads occupied can readily be generated by differing survey effort over time. An range change was therefore only noted where the number of hectads was >20% higher or lower, respectively, except for i) bats, where the radical change in survey methodologies over this time invalidates comparisons (an exception was made for the horseshoe bats as the methods used for these did not vary temporally; and ii) species where there were very few records in the first Atlas period, which would mean that small changes in observer effort could have a substantial increase on the percentage change observed. Given that the 1995 *Review of British Mammals* (Harris et al.,

[2] Later start dates were used for red squirrel, grey squirrel and water vole because of their rapid recent changes in range.

1995) relied heavily on mammal distribution data presented in the 1960-92 Mammal Atlas (Arnold, 1993)), the period of comparison lies somewhere between 20 and 24 years. For simplicity, this document refers hereon to a time-frame of 20 years.

For consistency, trends from BTO Breeding Bird Survey (BBS) are provided for those species where presence data from this source were included in the distribution maps (i.e., moles, rabbits, badgers, foxes and hedgehogs; see 2.5 Species' distribution maps). Data from the Game and Wildlife Conservancy Trust's (GWCT) National Gamebag (NGB) survey and the Bat Conservation Trust's National Bat Monitoring Programme (NBMP) are also provided where relevant. Potential issues that must be considered when interpreting these surveys are noted within the species' reports.

2.9 Future prospects

An assessment of the future prospects, in terms of the likely changes in population size, range size and habitat quality, was carried out for each species. The assessment is based on several factors:

- Changes in population size and range over the last 20+ years by comparison of the current estimated population size and range to those in Harris et al. (1995 (population size)) and Arnold (1993 (range size)). The records used in those reports extended to 1991-1992, and those in the current review extended to early 2016. Where relevant, reference was made to other data on trends found in the recent literature.
- Direct drivers of change (e.g., predation, persecution) in range and population size, as identified for each species in the individual species' accounts.
- Indirect drivers of change (factors affecting habitat availability, connectivity or quality) as identified in each species' account, as well as general habitat changes which may affect the species in the future.
- Drivers related to climate change. These are most likely to affect the species via changes in suitable habitat or climatic conditions.

Assessments of future prospects were based on a combination of empirical evidence and expert opinion. A summary table of results is presented in each species' account. A more detailed assessment can be found in Appendix 7. Populations have been ranked as 'stable' unless there is evidence of previous population declines or reductions in range or habitat.

However, for a high proportion of species, the absence of evidence of a previous population decline reflects an absence of evidence on which to make a judgement, rather than positive evidence that the population is stable. Great care should therefore be exercised when interpreting the future population prospects.

3 Habitat comparison results

The most abundant habitats in the LCM2007 dataset are improved grassland (74,239km^2), and arable and horticulture (hereafter ‘arable land’; 62,985km^2; Table 3.1a). Errors in the estimated change in extent of these habitats are, therefore, likely to have the biggest implications for the assessment of population size changes for many species.

The Countryside Survey, which uses consistent survey methodology over time, suggests an increase of 1.6% in improved grassland between 1990 and 2007 when improved, neutral, calcareous and acid grassland are combined. In contrast, the present analysis (i.e., comparison between Harris et al. (1995) and the LCM2007) suggests a 40% increase. If we accept that the Countryside Survey is the most accurate assessment available, then our analysis overestimates the extent of change in this habitat by about 38%. The estimates of population change over time compared with Harris et al. (1995) may, therefore, also be overestimated by up to 38% for improved grassland habitat. The effect on temporal trends for these species is described further in the respective species’ accounts where comparisons to Harris et al. (1995) are possible.

The extent of arable land is estimated to have decreased by 8.3% between 1990 and 2007 according to the Countryside Survey, whereas the current analysis suggests a 3% increase (1974km^2), implying that our assessments of change in population size over time may have underestimated declines in arable habitats by 11%. Nine species under review are found in arable land, with >25% of the population estimate being derived from arable land for the brown hare *Lepus europaeus*, Chinese water deer, rabbit, stoat and harvest mouse *Micromys minutus*.

Coniferous and broadleaved woodland are the habitats used by more species than any other kinds of habitat (44 and 26 species respectively), and many species make use of both types. All 18 species of bat make some use of broadleaved woodland, and 4 are regularly recorded in coniferous woodland, especially in association with bat box schemes. The Countryside Survey suggests that broadleaved woodland increased by 4.7% between 1990 and 2007, whereas the comparison between the current review and Harris et al. (1995) would suggest an increase of 15%. For coniferous woodland, the Countryside Survey shows an increase of 6.4%, whereas the comparison between the current review and Harris et al. (1995) suggests a decrease of 4%. Changes in population size between the two review periods may therefore be overestimated by about 10% for broadleaved woodland habitats, and underestimated by 10% in coniferous woodland. If populations are evenly distributed between the two types of woodland, these differences will largely cancel each other out, but further assessment is given where relevant in individual species' reports. Note that for most species of bat, the population estimates were not derived using habitat-specific data and so will not be influenced by the issues outlined above.

The area of urban and garden habitat is estimated by the Countryside Survey to have increased by 4.5% between 1990 and 2007, whereas the comparison between the current review and Harris et al. (1995) indicates a decrease of 39%. This apparent decline in the area of urban environment is likely to be an artefact of overestimation in Harris et al. (1995), and contradicts evidence from other sources such as the Ordnance Survey. Changes in population size because of urban expansion are therefore likely to be 45% greater than suggested by a simple comparison between the current report and that of Harris et al. (1995). These effects are assessed in individual species' reports.

The habitat estimated to have changed by the highest percentage between Harris et al. (1995) and the LCM2007 is supra-littoral sediment (-72%). However, this habitat only covers 470km^2 in Britain and is occupied by just one species in this review: the rabbit. Given that it contributes a very small proportion of the total population estimate for this species, it is highly unlikely to have had a material impact on the assessment of change over time.

Table 3.1a Area of each broad habitat type from the LCM2007 dataset. Linear features (hedgerows) are estimated from the Countryside Survey 2007. Areas are given for Britain and per country in km^2 and km (hedgerows).

Habitat	Britain	England	Scotland	Wales
Broadleaved woodland	13,333	9,375	2,700	1,257
Coniferous woodland	14,592	3,073	10,075	1,444
Arable and horticulture	62,985	53,761	7,445	1,779
Improved grassland	74,239	40,761	22,073	11,405
Unimproved grassland	13,025	5,035	5,785	2,206
Dwarf shrub heath	20,727	3,681	15,914	1,132
Fen, marsh and swamp	101	69	26	6
Bog	10,281	2,040	7,822	419
Freshwater	2,665	811	1,737	117
Salt water	1,558	868	550	140
Montane habitats	4,991	370	4,604	17
Inland rock	1,227	427	712	89
Littoral rock	497	114	352	30
Littoral sediment	2,533	1,607	610	317
Supra-littoral rock	79	10	60	8
Supra-littoral sediment	480	185	231	64
Urban and gardens	14,086	11,744	1,443	899
Hedgerows	477,000	402,000	21,000	54,000

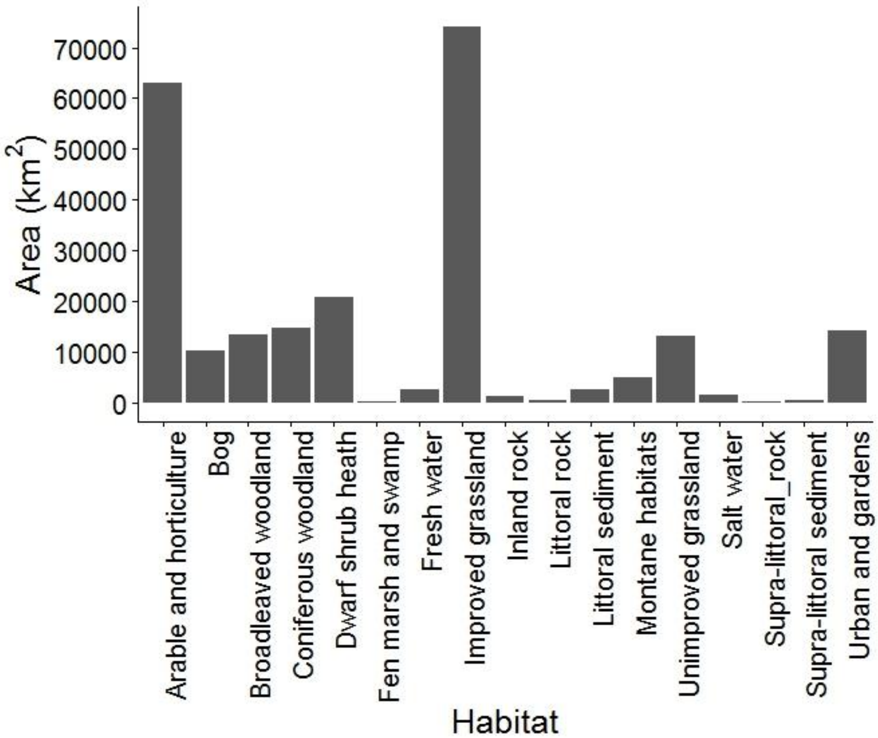


Figure 3.1a Area of each broad habitat class in the LCM2007 land use layer (Table 3.1a). Acid, neutral, and calcareous grassland have been included in the 'Improved grassland' class.

Table 3.1b Comparison of the area of each habitat type in the LCM2007 dataset (Countryside Survey 2007 linear features estimates for hedgerows) and in Harris et al. (1995). Habitat types from Harris et al. (1995) were matched to an LCM broad habitat type; land areas were summed where more than one habitat type fell within an LCM broad habitat (for details, see Appendix 1). Areas are given in km^2 (km for hedgerows). Differences in area (or length) per habitat between the two datasets are given using LCM2007-Harris1995. Percentage differences are: ((LCM2007-Harris1995)/Harris1995) *100.

Habitat	Harris 1995 (km^2)	LCM2007 (km^2)	Difference (km^2)	Difference (%)
Broadleaved woodland	11,581	13,333	1,752	15
Coniferous woodland	15,175	14,592	-583	-4
Arable and horticulture	61,011	62,985	1,974	3
Improved grassland	53,201	74,239	21,038	40
Unimproved grassland	25,013	13,025	-11,988	-48
Dwarf shrub heath	17,736	20,727	2,991	17
Bog	12,360	10,281	-2,079	-17
Freshwater	3,491	2,665	-826	-24
Inland rock	1,519	1,227	-292	-19
Littoral sediment	1,416	2,533	1,117	79
Supra-littoral sediment	1,702	480	-1,222	-72
Urban and gardens	23,280	14,086	-9,194	-39
Hedgerows	527,616	477,000	-50,616	-10

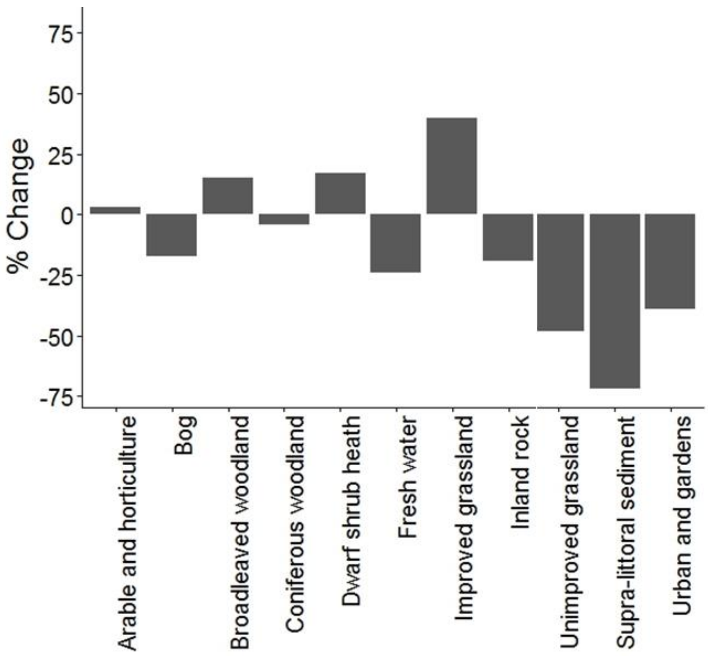


Figure 3.1b Percentage change in the area of each habitat type between Harris et al. (1995) and the LCM2007 land use layer (Table 3.1b). Habitat types from Harris et al. (1995) were matched to an LCM broad habitat type; land areas were summed where more than one habitat type fell within an LCM broad habitat (for details, see Appendix 1). Percentage differences are given as: ((LCM2007-Harris1995)/Harris1995) *100.

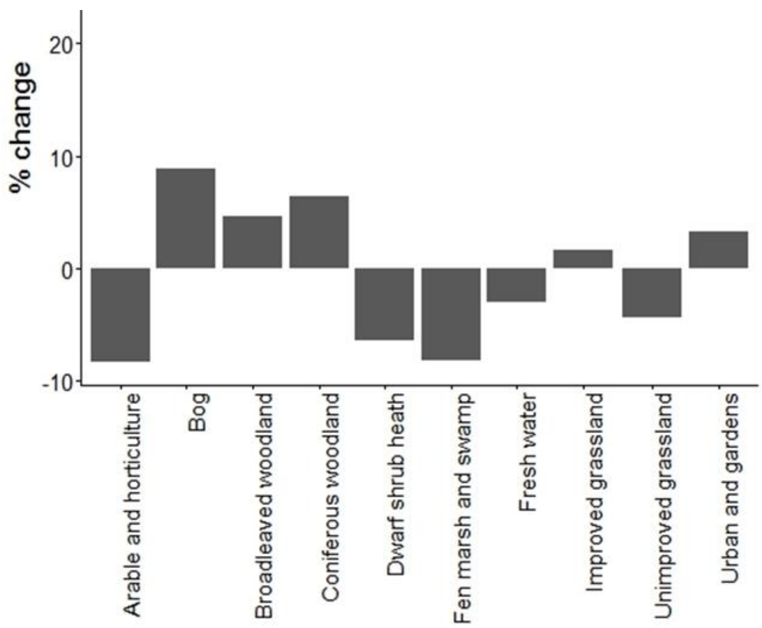


Figure 3.1c Percentage change in the area of each habitat type from Countryside Surveys 1990 and 2007. Habitats are presented only where they appear in both datasets. Data were reported in Table 2.2 of Carey et al. (2008).

4 ERINACEOMORPHA

4.1 Hedgehog *Erinaceus europaeus*

Habitat preferences

The hedgehog is found in most habitats, although it is increasingly associated with urban areas, and is often observed in gardens and amenity grasslands. Because it is a mobile, generalist species, road fatalities are fairly common, but these have declined over recent decades, providing some indication of a declining population (Wembridge et al., 2016b). Hedgehogs in rural villages have recently been shown to have small home ranges compared with other habitats, presumably because of greater foraging resource and nest site availability (Pettett et al., 2017). Density is higher in areas with amenity grassland compared with pasture (Micol et al., 1994; Young et al., 2006; Parrott et al., 2014); key prey items, including earthworms, ground beetles and tipulid larvae, are important in determining their distribution. The presence, and abundance, of badgers — one of the few natural predators of hedgehogs — is inversely linked with hedgehog distribution patterns (Doncaster, 1994; Young et al., 2006; Parrott et al., 2014; Trewby et al., 2014).

Status

Native.

Conservation Status

- IUCN Red List (GB: VU; England: [VU]; Scotland: [VU]; Wales: [VU]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 4.1a. Gaps in the species' distribution in Scotland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

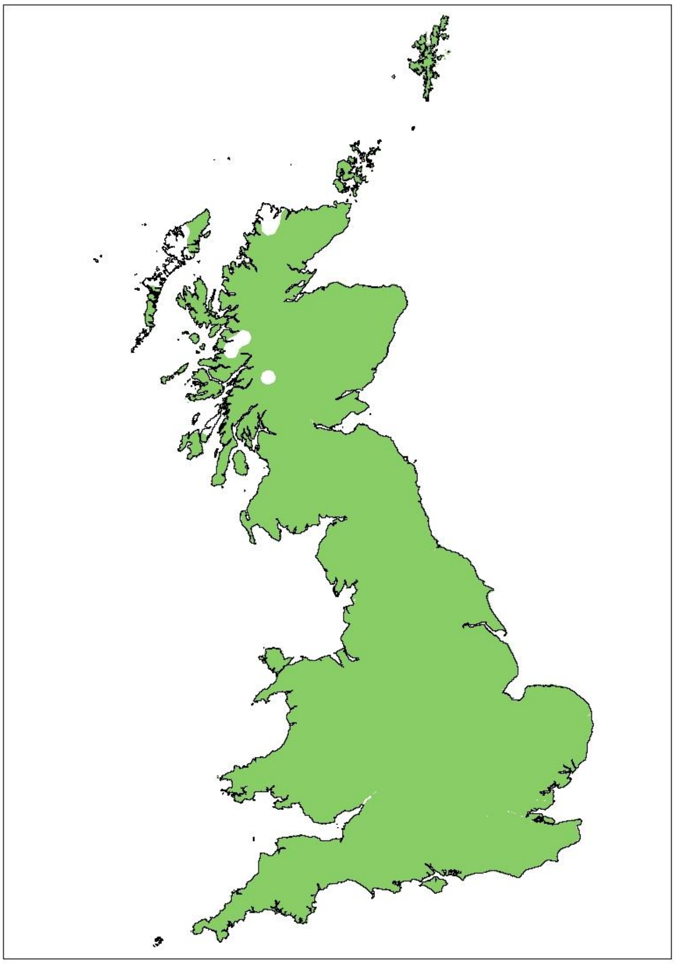


Figure 4.1a Current range of the hedgehog in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Hedgehog occurrence has been monitored by a number of nationwide surveys, where percentage occupancy is the percentage of unique locations where hedgehogs or field signs were observed. Several of these studies have measured hedgehog occurrence in gardens and amenity grassland (Micol et al., 1994; Roos et al., 2012; Parrott et al., 2014). The results of Living with Mammals (PTES), Garden Birdwatch (BTO), Make Your Nature Count (The Royal Society for the Protection of Birds) and HogWatch (PTES) are summarised by Roos et al. (2012). The mean percentage occupancy for the most recent year (per country where available) from these surveys, and from Micol et al. (1994) and Parrott et al. (2014), was used as percentage occupancy for urban and gardens. The HogWatch data came from 2006, as very few responses were received in subsequent years.

Percentage occupancy was also available for some other habitats, including pastoral land, grassland, woodland, arable land, roads and along waterways. However, most of the surveys incorporate a variety of habitats, rather than providing habitat-specific occupancy values. Data from a range of studies conducted since the Harris et al. (1995) review were summarised by Roos et al. (2012). Mean percentage occupancy for the most recent year (per country where available) from these surveys, and from Hof and Bright (2012) and Parrott et al. (2014), was used as percentage occupancy for all other habitats.

Results

Ten papers were returned from the literature search in total. One paper contained pre-breeding population density estimates but was not included because they were only for blackland (peatland) and machair habitats that are specific to the north west of Scotland and the offshore islands (Jackson, 2007). Three papers contained measures of percentage occupancy, two contained post-breeding density estimates, one contained relative abundance measures, and two presented habitat suitability measures. Adjustments were made the density estimates for Urban and gardens, and for Improved grassland, to account for the fact that the reported densities contained a significant proportion of juveniles. The median proportion of adults observed by Parrott et al. 2014 in amenity grassland in 4 regions was 76.75% (samples were too small to provide independent estimates for pasture in this study, and the same was true for the study by Young et al. (2006)).

Table 4.1a Median density estimates for hedgehogs with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})*	-95%CI	+95% CI	Source**	n†	%Occ††
Urban and gardens	13,800	41	25	118	Parrott et al. (2014) Young et al. (2006)	4 1	49%
Improved grassland	72,800	3	<0.01	8	Parrott et al. (2014) Young et al. (2006)	4 1	37%
Arable and horticulture	62,600	5	-	-	Harris et al. (1995)	1	37%
Broadleaved woodland	13,100	40	-	-	Harris et al. (1995)	1	37%
Coniferous woodland	14,400	5	-	-	Harris et al. (1995)	1	37%
Unimproved grassland	12,200	40	-	-	Harris et al. (1995)	1	37%

* Density reported in literature multiplied by median proportion of adults observed in population at same time of year (76.75%) to estimate to estimate pre-breeding densities
Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 4.1b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates. Values were obtained by multiplying population density estimates in Table 4.1a with the area of habitat within the species' distribution, and adjusting for occupancy. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	123,000	[597,000]	-	-
Scotland	47,100	[196,000]	-	-
Wales	18,800	[87,000]	-	-
Britain	**189,000**	**[879,000]**	**-**	**-**

Critique

Very few recent density estimates were found for hedgehogs: data were available only for improved grasslands and urban areas (Parrott et al., 2014; Young et al. 2006) and neither study assessed pre-breeding densities. These were converted to pre-breeding densities by multiplying the observed values by the fraction of adults captured in one study (Parrott et al. 2014). Nevertheless, the adult population in these habitats may be over-estimated by 27-37%, given other information on population demographics which suggests only approximately 40-50% of animals observed in late summer and autumn are adults (Harris and Yalden, 2008). Urban and garden (including amenity grassland) contribute 32% of the total population size, but, in addition to the above uncertainties, it should be noted that no account could be taken of the likely variations with urban density. Improved grassland and arable land are the most abundant habitats across the species' distribution (39% and 33% respectively; Figure 4.1b), but they contribute only 9% and 13% to the population size. Unimproved grassland and broadleaved woodland contribute 20% and 18% respectively to the population size, but the density estimates are based on Harris et al. (1995), and were largely based on the expert opinion expressed in Burton (1969). The population densities in Table 4.1a, and population sizes in Table 4.1b, are therefore likely to be over-estimated.

As the current estimate largely uses density estimates from Harris et al. (1995), the reason for a reduction in population size (from 1,555,000 in 1995) is likely to be the use of percentage occupancy data, rather than differences in population density, as occupancy data were not used in Harris et al. (1995). Percentage occupancy was taken from a large number of sources (see Micol et al., 1994; Hof and Bright, 2012; Roos et al., 2012; Parrott et al., 2014), although a mean value was used for most habitats. Percentage occupancy ranged from 0% to 81% across all studies, in all parts of Great Britain, so stratification by area, with habitat-specific occupancy data, would significantly improvement the analysis.

If we assume the estimate provided in Harris et al. (1995) to be the best estimate available for that time period, then applying the decline in relative abundance of hedgehogs estimated by Roos et al. (2012) from citizen-science surveys (40% every 10 years) would result in a total population size of 560,000 in Britain, which is lower than our estimate. This extrapolation is, however, subject to uncertainty in the original population size as well as in the trend data. The species' range has remained relatively stable since 1993 (Arnold, 1993), suggesting that declines in population size are owing to reduced density or occupancy. The density data are comparable with those from studies in continental Europe, which are between 2km^{-2} and 300km^{-2}, depending on habitat type (Huijser and Bergers, 2000). As well as suggesting that hedgehog density is highly variable, these figures highlight the need for more empirical data on the population density and occupancy of hedgehogs to improve confidence in the current population density and subsequent size estimates. The reliability of the population density estimates is given in Table 4.1c. To reflect the uncertainty arising from the use of expert opinion in Harris et al. (1995), the density estimates classified as having been derived from a restricted range.

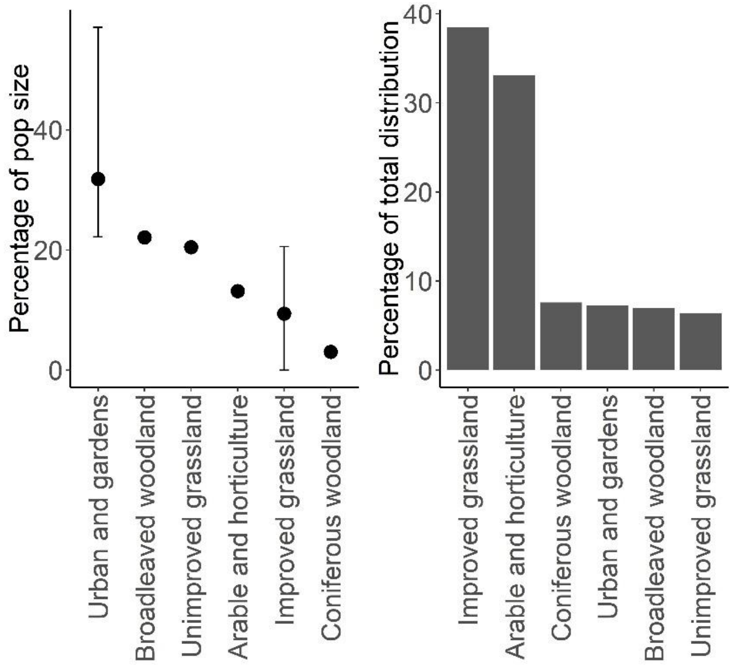


Figure 4.1b Left: The percentage of the total population of hedgehogs derived from each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 4.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat			
			Broadleaved woodland	Unimproved grassland	Improved grassland	Arable and horticulture
Location of study sites	0	Estimates from one location				
	1	Estimates restricted	1	1	1	1
	2	Estimates widespread				
Sample size	0	<10 density estimates	0	0	0	0
	1	10-30 density estimates				
	2	>30 density estimates				
Occupancy data available?	0	No				
	1	Yes	1	1	1	1
		Habitat score	**2**	**2**	**2**	**2**
Overall reliability score			**2**			

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) provided an estimate of 1,555,000 hedgehogs in Britain. The current estimate is substantially lower (-43%). Nationally, there are changes between the two reviews in the estimated availability of key habitats (broadleaved woodland, improved and unimproved grassland, and arable land), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and Table 3.1b for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape, and the probable over-estimation of the extent of urban cover in Harris et al (1995) — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a population size of 944,471 which is a 39% decrease.

It is possible that population size has declined further than estimated here, but all current estimates are very uncertain.

Other evidence of changes through time

The relative abundance of hedgehogs has been monitored by several organisations in the UK over the last 25 years. These studies, which used different methodologies and were of varying duration, were reviewed by Roos et al. (2012). There was considerable inter-annual variation within each study, and also variation between them — annual declines ranged from a mean of 1.8% to 10.7% — but there was consistency in the direction of the effect. The authors inferred a decline in relative abundance of 40% in 10 years. However, the scale of this decline contrasts with another study which used non-systematic occupancy records from Biological Records Centres and adjusted for survey effort (Hof and Bright, 2016). Here, a decline of between 5.0% and 7.5% was found for England over a 40- year period, which would mean a maximum decline of 1.9% over 10 years, though the reliability of the subsampling approach as a method of detecting trends has been questioned (Calcutt et al. 2018).

Table 4.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995), and trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		All countries*		
	Data deficient				

*Roos et al. (2012).

Drivers of change

Table 4.1e Drivers of population change for hedgehogs between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Agricultural intensification results in a reduction in invertebrate biodiversity and loss of hunting opportunities. There may also be an impact of pesticide use, including in gardens, on prey abundance.	Hof and Bright (2010)	Negative
Habitat availability.	Agricultural intensification results in the loss of hedgerows and field margins, with the effect of reducing habitat connectivity and availability of refugia.	Hof and Bright (2010)	Negative
Vehicle collisions	Road casualties are likely to have had an important effect on local populations.	Huijser and Bergers (2000) Wembridge et al. (2016b)	Negative
Predation/ competition.	Predation and possible competitive exclusion by badgers.	Parrott et al. (2014) Trewby et al. (2014)	Negative

Data deficiencies

Table 4.1f Areas where further research is required to improve the reliability of population size estimates for the hedgehog.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Arable and horticulture Broadleaved and coniferous woodland, unimproved grassland	Density estimates were taken from Harris et al. (1995).
Density estimates do not represent within-habitat variability.	All habitats	Densities range from $2km^{-2}$ to $300km^{-2}$ in continental Europe (Huijser and Bergers, 2000).

Future prospects

Table 4.1g An assessment of the future prospects of the hedgehog, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline*

*Causes of historic declines are poorly understood. There is no evidence that the trend is likely to change.

5 SORICOMORPHA

5.1 European mole *Talpa europaea*

Habitat preferences

The mole is a highly adaptable species, found in most habitats where invertebrate prey is present and the soil is sufficiently deep to allow tunnel construction. In low-lying areas and regions prone to flooding, it can build more permanent ‘fortresses’ which contain a nest chamber above the level of the surrounding land. These are often provisioned with stores of decapitated worms for consumption when the surrounding area is flooded or frozen. Originally an inhabitant of broadleaved woodlands, the mole thrives in pastures and on arable land. It lives at low densities in coniferous forests, on moorland and in sand-dune systems, probably because of the paucity of suitable prey (Harris and Yalden, 2008). Home ranges are small — around 0.2ha for females and 0.3ha for males — and adults rarely disperse once a territory is established (Stone and Gorman, 1985). Although it is aggressive towards intruders, agonistic encounters are very rare (Gorman and Stone, 1990). Earthworms are the most important prey item, particularly in winter, whereas in summer up to 50% of the diet is formed of insects (adult and larvae) (Funmilayo, 1979).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

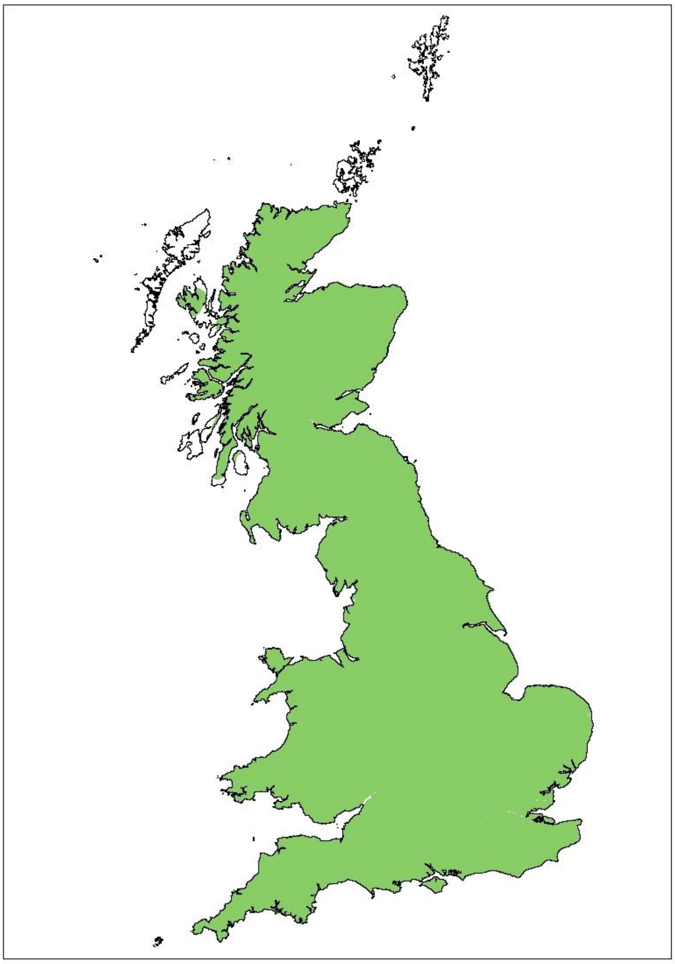


Figure 5.1a Current range of the mole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Two density estimates from Harris et al. (1995) fell into the LCM2007 'Improved Grassland' category; these were 1.3ha^{-1} for lowland improved grassland, and 4ha^{-1} for semi-improved grassland. For the current analysis, a mean value of 2.65ha^{-1} was used.

Results

No papers were identified with population size estimates for moles, nor were any estimates obtained from expert opinion. The population density estimates (Table 5.1a) are therefore taken from Harris et al. (1995). These were based on expert opinion, where each habitat was deemed ‘poor’ or ‘good’, and assigned a density of 1.3ha^{-1} or 4ha^{-1}, respectively. Population size estimates are provided in Table 5.1b.

Table 5.1a Median density estimates for moles, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

	Area within range (ha)	**Density (ha^{-1})**	**-95%CI**	**+95%CI**	**Source***	**n****	**%Occ†**
Arable and horticulture	6,250,000	1.30	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	1,310,000	4.0	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	1,430,000	1.30	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	1,820,000	1.30	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	7,250,000	4.0	-	-	Harris et al. (1995)	1	n/a
Improved grassland	1,150,000	2.65	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 5.1b Area of suitable habitat (not adjusted for occupancy) within the species’ range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 5.1a with the area of habitat within the species’ distribution. It was not possible to calculate confidence limits as a measure of variance was not available from Harris et al (1995).

Country	**Area of suitable habitat (ha)**	**Population size**	**-95%CI**	**+95%CI**
England	11,500,000	[24,300,000]	-	-
Scotland	5,800,000	[12,200,000]	-	-
Wales	1,910,000	[4,900,000]	-	-
Britain	**19,210,000**	**[41,400,000]**	**-**	**-**

Critique

No percentage occupancy data were available; therefore, the population size is overestimated for this species. 46% of the estimated population size for moles was derived from improved grassland habitat, with a further 19% from arable and horticulture. These habitats represent 38% and 33% of the species' range, respectively (Figure 5.1b). As the density estimates for these habitats were derived from Harris et al. (1995), a sensitivity analysis was not possible. To assess reliability, we have considered the population density estimates from Harris et al. (1995) to be the expert opinion of the authors and, therefore, to represent a restricted area of the species' range (Table 5.1c).

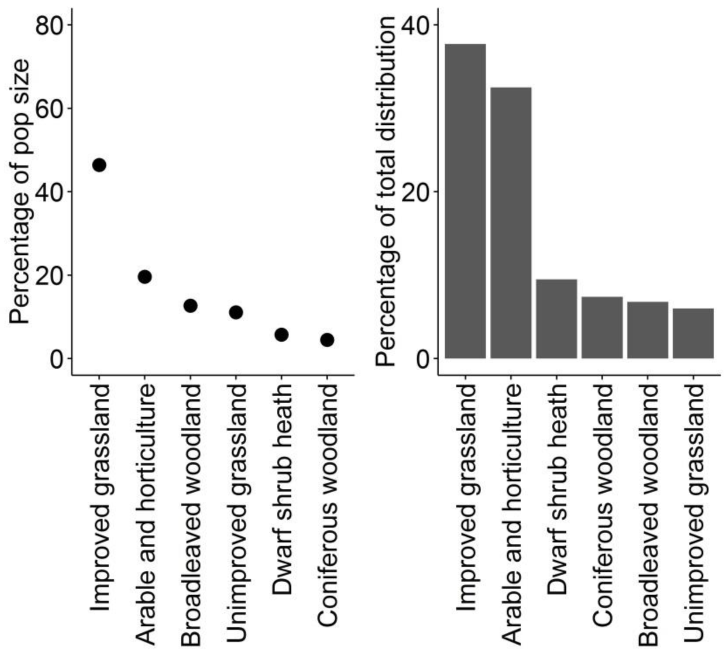


Figure 5.1b Left: The percentage of the total population of moles derived from each habitat type. It was not possible to compute error bars for this species. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 5.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Improved grassland	Arable and horticulture
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

* Populations may be unstable owing to inter-annual cycles, documented fluctuations in population size, or as a result of management.

Changes through time

Comparison to Harris et al. (1995)

Total population size was estimated to be 41,400,000 in Britain, with 24,300,000 in England, 12,200,000 in Scotland and 4,900,000 in Wales. The density estimates used in the current analysis are taken from Harris et al. (1995), so any differences are entirely owing to changes in the species' distribution and land classification. Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a population size of 38,400,000, and a 23% increase in population size since 1995. However, a lack of confidence intervals means that it was not possible to assess whether the difference across time is significant.

Other evidence of changes through time

Mole signs have been recorded as part of the BTO Breeding Bird Survey since 1995. The number of 1km survey squares with signs of moles was 7% in 1995, 32% in 2003, and 18% in 2015. The extent to which differences between survey years reflects variation in recorder effort or true biological variation is not known. Nor is it possible to relate presence of signs to population estimates. A summary of trends in population size and range is provided in Table 5.1d.

Table 5.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 5.1e Drivers of population change for moles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Habitat quality.	Intensification of agricultural practices (ploughing/re-seeding) may reduce food (earthworm) density.	Edwards and Lofty (1972)	Negative
Habitat availability.	The loss of hedgerows, through neglect or removal, and reduction in unimproved grassland may reduce availability of refugia.	Harris et al. (1995)	Negative

Data deficiencies

Table 5.1f Areas where further research is required to improve the reliability of population size estimates for the mole.

Data deficiencies	Habitat	Details
Density estimates more than 10 years old.	All habitats	All population density estimates are taken from Harris et al. (1995).
Density estimates do not reflect within-habitat variability.	All habitats	No ranges or confidence limits were available for the density estimates.
No occupancy data.	All habitats	

Future prospects

Table 5.1g An assessment of the future prospects of the mole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Stable/Decline

5.2 Common shrew *Sorex araneus*

Habitat preferences

The common shrew is found in most terrestrial habitats, providing that some low vegetation cover is available. It is most abundant in thick grass, bushy scrub, hedgerows and broadleaved woodland. Fallow land, roadside verges and urban habitats are colonised rapidly. At high altitudes, it is occasionally found among heather and more frequently in stable scree (Harris et al., 1995). Its very high energy requirements, the result of its high surface area:volume ratio, means that it requires habitats with high invertebrate abundance. It may therefore be negatively affected by changes to agricultural practice and/or pesticide use which reduce its prey availability. There is very little research in the recent literature: most records are reported as part of multi-species small mammal studies.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales; [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 5.2a. Gaps in the species' distribution throughout the mainland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

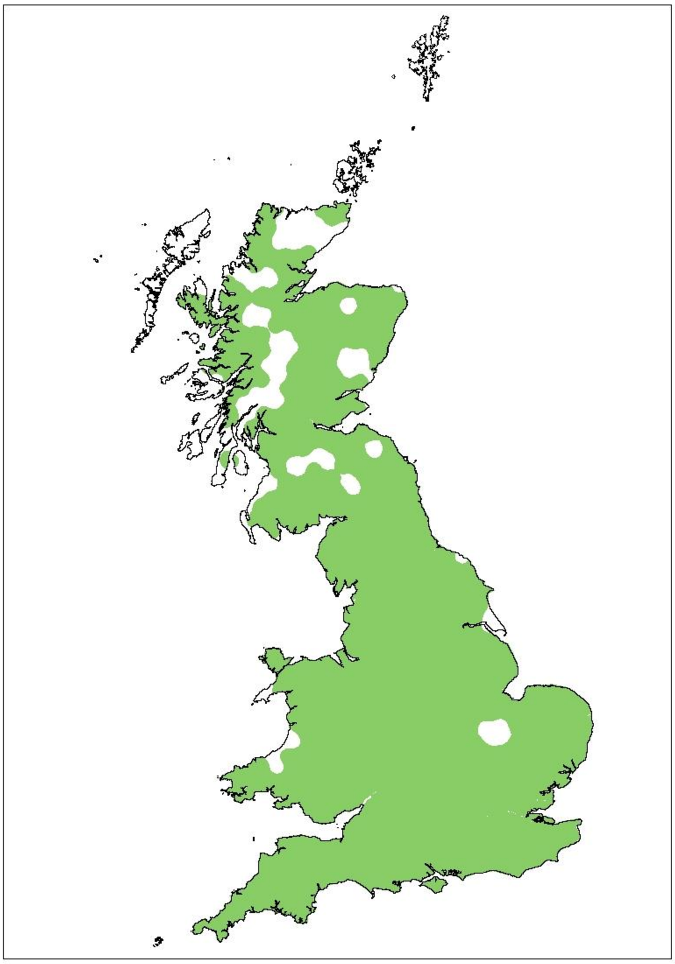


Figure 5.2a Current range of the common shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Specific methods

As many small mammals in arable land are known to use field margins as primary habitats, with cropped areas incorporated into home ranges only before the harvest (Tattersall et al., 2001), we assume that the population within fields will be captured in the estimate for hedgerows. Arable land is therefore excluded from the analysis.

Results

The literature search identified eight relevant papers, four of which contained pre-breeding density estimates, three contained post-breeding estimates of density, one contained percentage occupancy for hedgerows (Gelling et al., 2007), and one containing estimates sourced from papers already included in the current analysis. Population density estimates are provided in Table 5.2a, and population size estimates in Table 5.2b.

Table 5.2a Median density estimates of common shrews with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerow length and density within hedgerows are presented as km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,260,000	2.0	0.50	7.25	White and Searle (2007)	5	n/a
Coniferous woodland	1,210,000	1.25	0.25	2.75	White and Searle (2007)	5	n/a
Dwarf shrub heath	1,480,000	0.50	0.25	3.75	White and Searle (2007)	5	n/a
Unimproved grassland	1,040,000	4.80	4.0	9.50	Pernetta (1977) Churchfield et al. (1995) White and Searle (2007)	9 8 10	n/a
Bog	555,000	12.0	0	35.0	Shore and Mackenzie (1993) ††	1	n/a
Urban and gardens	1,350,000	1.95	0.4	3.95	expert opinion	2	n/a
Fen, marsh and swamp	8,000	0.50	0.05	1.0	expert opinion	1	n/a
Improved grassland	6,660,000	0.10	0	0.50	expert opinion	1	n/a
Hedgerows	460,000 (km)	7.70	2.07	13.9	Kotzageorgis and Mason (1997) Flowerdew et al. (2004)	9 9	37.8%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
†† Median density and confidence limits using data from this paper were provided by the authors.

Table 5.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 5.2a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha) *	Population size	-95%CI	+95%CI
England	7,480,000	[11,000,000]	3,520,000	29,500,000
Scotland	4,340,000	[7,690,000]	1,980,000	22,900,000
Wales	1,740,000	[2,330,000]	1,010,000	6,120,000
Britain	**13,600,000**	**[21,100,000]**	**6,520,000**	**58,500,000**

* The lengths of hedgerows are 394,000km in England, 17,100km in Scotland, and 48,600km in Wales.

Critique

No percentage occupancy data were available for most habitats; therefore, the population size estimate for this species is an overestimate. Although most of the land cover within the species' range is improved grassland (48%; Figure 5.2b), population density in this habitat is thought to be small (0.1ha^{-1}, 95%CI = 0-0.5; Table 5.2a). This density estimate is based on the opinion of one expert; any uncertainty in this estimate will affect the population size estimate. In addition, at least some of the animals included within density estimates for improved grassland are likely to be individuals that also live in hedgerows, potentially introducing an element of double counting. Deriving robust estimates for improved grassland is therefore considered a research priority. Most of the estimated population is derived from unimproved grassland habitat (24%), for which the average density estimate is supported by 18 estimates; and from bog (32%), for which values were based on data from Shore and Mackenzie (1993) provided by the authors. The population density for bog habitat reported in Harris et al. (1995) was markedly lower, at 0.5ha^{-1}, although this value was based on expert opinion rather than empirical data. An increase in population density of this magnitude in the last 20 years is unlikely, and so further data are urgently required to increase confidence in the density estimate for this habitat. Reliability scores for unimproved grassland and bog are provided in Table 5.2c.

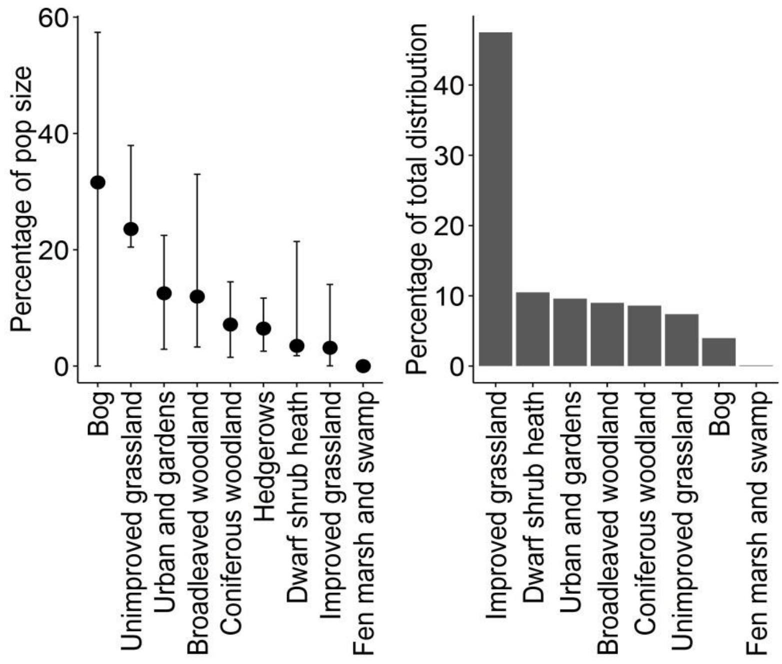


Figure 5.2b Left: The percentage of the total population of common shrews derived from each habitat type. Error bars are obtained by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 5.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**		
			Unimproved grassland	**Bog**	**Improved grassland**
Location of study sites	0	Estimates from one location		0	
	1	Estimates restricted	1		1
	2	Estimates widespread			
Sample size	0	<10 density estimates		0	0
	1	10-30 density estimates	1		
	2	>30 density estimates			
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**2**	**0**	**1**
		Overall reliability score	**1**		

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated population sizes as 41,700,000 in total, with 26,000,000 in England, 11,500,000 in Scotland and 4,200,000 in Wales. These estimates were made using sparse data on population density, which can be highly variable within each habitat. Similar uncertainties persist in the current estimates, where the confidence limits range from 6 million to 58 million individuals. This level of uncertainty means that no inference can be made about temporal trends.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature search.

Table 5.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995), and trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

* Differences in range are likely to be the result of variable recorder effort.

Drivers of change

Table 5.2e Drivers of population change for the common shrew between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.2f Areas where further research is required to improve the reliability of population size estimates for the common shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Urban and gardens Improved grassland Fen, marsh and swamp	Density estimates are based on expert opinion.
Density estimates are more than 10 years old.	Hedgerows Bog	Density estimates are from 1997 and 2003, respectively.
No occupancy data.	All habitats except hedgerows	

Future prospects

Table 5.2g An assessment of the future prospects of the common shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Decline

5.3 Pygmy shrew *Sorex minutus*

Habitat preferences

The pygmy shrew is found in all terrestrial habitats and shows a preference for areas with dense ground cover, particularly unimproved grasslands (O'Keeffe and Fairley, 1981). It frequently uses surface burrows through thick vegetation or exploit the abandoned burrows of other species. Abundance is high in wet habitats, such as moorland and blanket bog (Croin-Michielsen, 1966), and also in ditches and on the edges of riparian habitats. Its very high energy requirements, the result of its high surface area:volume ratio, means that it requires habitats with high invertebrate abundance. It may therefore be negatively affected by changes to agricultural practice and/or pesticide use which reduce its prey availability. There is very little research in the recent literature: most records are reported as part of multi-species small mammal studies.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 5.3a. Gaps in the distribution in England and Wales are likely to result from a lack of survey effort, rather than true absences. It is less clear whether larger gaps in Scotland represent true gaps in distribution or are influenced by survey effort. Further surveys are recommended in these areas to increase confidence in the current distribution.

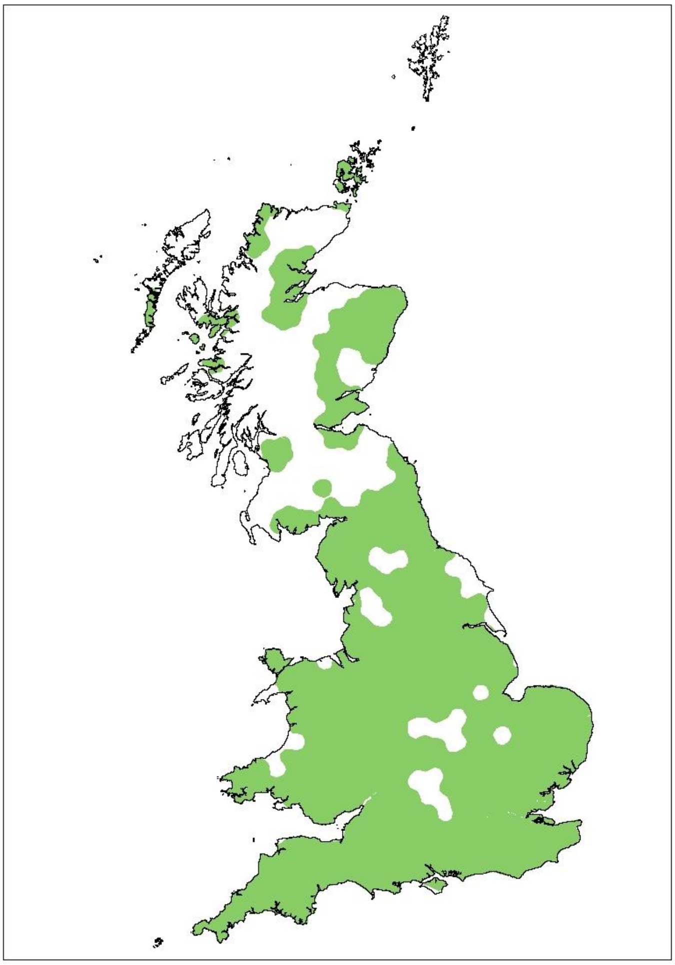


Figure 5.3a Current range of the pygmy shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

As the same animals are likely to use the cropped area and the field margins (hedgerows and buffer strips), we have excluded the estimate for arable fields to avoid counting the same animals twice.

Results

Two papers were identified by the literature search: one reported pre-breeding density estimates, and one contained post-breeding density estimates. Population density estimates are shown in Table 5.3a, and population size estimates are provided in Table 5.3b.

Table 5.3a Median density estimates for pygmy shrews, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerow length and density within hedgerows are presented as km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Unimproved grassland	823,000	4.80	1.20	4.80	Pernetta (1977)	9	n/a
Improved grassland	5,440,000	0	0	1.0	expert opinion	1	n/a
Bog	333,000	3.0	0	8.50	expert opinion	2	n/a
Broadleaved woodland	1,100,000	1.0	0	3.0	expert opinion	2	n/a
Urban and gardens	1,250,000	0.10	0	5.0	expert opinion	1	n/a
Dwarf shrub heath	870,000	0.10	0	5.0	expert opinion	1	n/a
Fen, marsh and swamp	7,190	0.02	0	2.0	expert opinion	1	n/a
Hedgerows	427,000 (km)	0.1	0.01	30.0	expert opinion	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 5.3b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 5.3a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha) *	Population size	-95%CI	+95%CI
England	6,600,000	[3,690,000]	552,000	27,900,000
Scotland	1,660,000	[1,430,000]	217,000	6,040,000
Wales	1,550,000	[1,170,000]	231,000	4,970,000
Britain	**9,810,000**	**[6,300,000]**	**999,000**	**38,900,000**

* The lengths of hedgerows are 371,000km in England, 9,700km in Scotland and 46,600km in Wales.

Critique

No percentage occupancy data were available; the population size for this species is therefore overestimated. All of the population density estimates except those for unimproved grassland are based on the opinion of two to four experts, depending on habitat. 63% of the population resides in unimproved grasslands, which is supported by nine individual density estimates from one paper (Pernetta, 1977). Stepwise deletion and replacement of each of these nine density estimates reduced population size by 8% in four of the nine instances, but all alternative population sizes fell within the confidence limits of the original. Most of the species' distribution consists of urban areas and gardens (29%), despite population densities being relatively low in this habitat (Table 5.3a). It is likely that at least some of the population found in improved grassland lives primarily within hedgerows. The inclusion of both of these habitat types may have introduced an element of double counting, although only the upper confidence limit would be affected since the median density for improved grassland was estimated to be zero. A reliability assessment is provided in Table 5.3c.

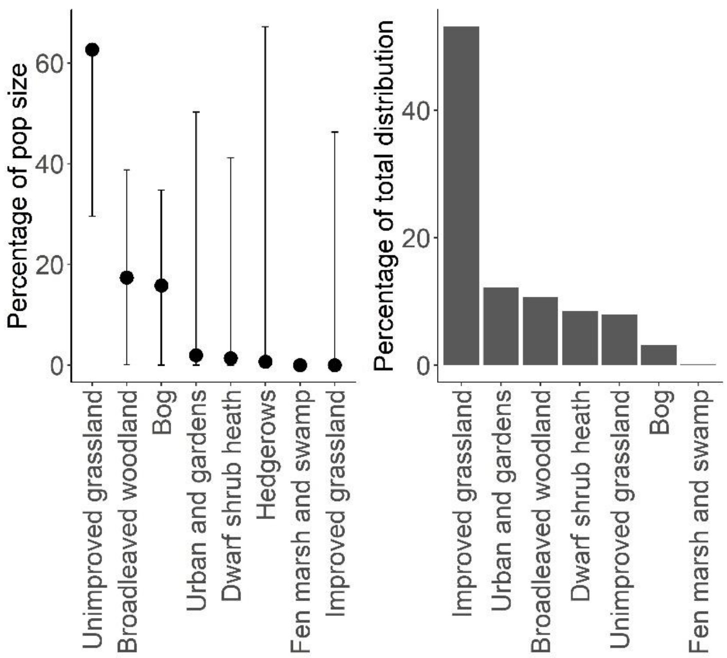


Figure 5.3b Left: The percentage of the total population of pygmy shrews accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 5.3c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**	
			Unimproved grassland	**Urban and gardens**
Location of study sites	0	Estimates from one location	0	
	1	Estimates restricted		1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**0**	**1**
		Overall reliability score	**0.5**	

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total of 8,600,000 pygmy shrews in Britain, with 4,800,000 in England, 2,300,000 in Scotland and 1,500,000 in Wales. These figures are based on the ratio of pygmy shrews to common shrews, which itself was derived from very sparse population density data. Our estimate is based on expert opinion with empirical data for one habitat type only, and so is also highly uncertain. The difference in population size estimates is, therefore, likely to be caused by methodological differences, rather than any true change in population size.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is provided in Table 5.3d.

Table 5.3d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

* Differences in range size are likely to be because of variable recorder effort.

Drivers of change

Table 5.3e Drivers of population change for the pygmy shrew between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.3f Areas where further research is required to improve the reliability of population size estimates for the pygmy shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Bog Broadleaved woodland Urban and gardens Dwarf shrub heath Fen, marsh and swamp Hedgerows Improved grassland	Density estimates are based on expert opinion.
Limited density estimates for the key habitat.	Unimproved grassland	Median density is based on 9 density estimates.
Density estimates are more than 10 years old.	Unimproved grassland	Density estimates are taken from Pernetta (1977).
No occupancy data.	All habitats	

Future prospects

Table 5.3g An assessment of the future prospects of the pygmy shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline
Range	Stable
Habitat	Decline

5.4 Water shrew *Neomys fodiens*

Habitat preferences

The water shrew is usually associated with aquatic habitats, including rivers, streams, marshes, fens, reed beds, watercress beds and, occasionally, garden ponds (Greenwood et al., 2002). However, it is also found regularly — but infrequently — in non-riparian habitats, including hedgerows, especially those associated with ditches, regardless of whether the features are currently wet (Fiona Mathews, *pers. obs.*). Research on the specific habitat preferences of the species is limited. Greenwood et al. (2002) found that rivers with steep, high banks (45° incline, 1.5m banks) have a higher chance of occupancy, although these characteristics are not essential. A complex relationship was identified between occurrence and vegetation cover: water shrews are found only at sites with trees, and have a higher occurrence where tree cover is sparse. The decrease in occurrence with dense tree cover is thought to be because of the correlated decrease in ground vegetation. High water quality is thought to be an important determinant of occurrence, through both the direct ingestion of pollutants via grooming, and the reduction of prey diversity and abundance. It is currently unclear whether water shrews have been negatively affected by organochlorine pesticides in water. Further testing is required to confirm all habitat associations, which were identified in a single study in the south east of England (Greenwood et al., 2002), and may not represent habitat requirement throughout Britain. Loss of connectivity between habitat patches is likely to have occurred as a consequence of physical features such as dams, weirs, and embankments, as well as human activities such as land draining and the deepening, straightening and widening of channels.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC)
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 5.4a. Gaps in the species' distribution in England and Wales are likely to represent areas lacking survey effort, rather than true absences. It is unclear whether larger gaps in Scotland are true gaps in the distribution or reflect a lack of survey effort.

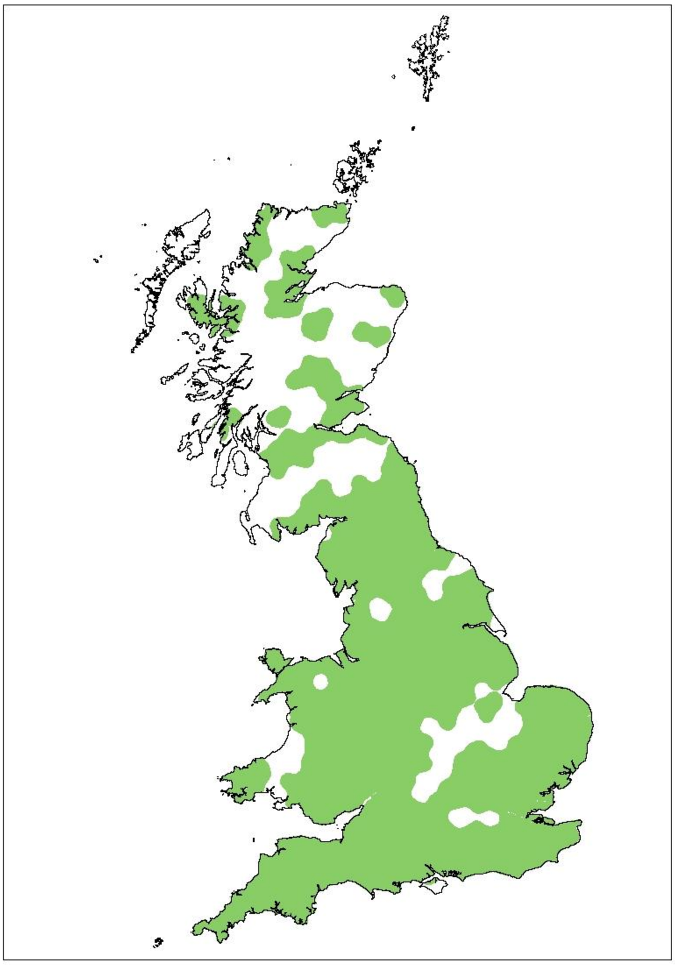


Figure 5.4a Current range of the water shrew in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Harris et al. (1995) estimated population size based on the ratio of common shrews to water shrews, stating that water shrews are known to use a variety of habitats. In the absence of any recent data on population size or density for water shrews, the same method and ratios were used here, although population sizes were adjusted to reflect range size.

First, the density of common shrews in their total range was calculated by dividing the total population size (see Table 5.2b) by area (total range area, not the area of suitable habitat). For the water shrew, density was calculated by dividing the density of common shrews by the ratio of common shrews to water shrews. The density of water shrews was then multiplied by the total range area to give an estimate of total population size for each country.

Results

No papers with pre-breeding population density estimates or trends were identified by the literature search. The density estimate for unimproved grassland was derived from expert opinion ($0.5ha^{-1}$; 95%CI = $0\text{-}1ha^{-1}$). One reference to occupancy for waterways was identified, derived from a national survey carried out in 2004-2005 (Carter and Churchfield, 2006). Surveys took place at 2159 sites across Britain, with signs of water shrews found at 17.4% of sites. As the population size estimate is not habitat-based, occupancy values could not be applied.

Table 5.4a Density of common shrews and water shrews per hectare of their total range. For common shrews, density was calculated by dividing total population size (see Table 5.2b) by area. For water shrews, density was calculated by dividing the density of common shrews by the ratio of common shrews to water shrews.

Country	Common shrew					Water shrew		
	Area within species' range (ha)	Estimate (ha^{-1})	-95CI	+95CI	Ratio*	Estimate (ha^{-1})	-95CI	+95CI
England	1,280,000	8.59	2.75	23.1	22.1 : 1	0.39	0.12	1.04
Scotland	529,000	14.5	3.74	43.3	31.7 : 1	0.46	0.12	1.36
Wales	194,000	12.0	5.20	31.5	15.3 : 1	0.78	0.34	2.06

*Ratio of common shrew to water shrew as derived from Harris et al (1995).

Table 5.4b Total population size estimates, with 95% confidence intervals, for England, Scotland, Wales, and the whole of Britain. Values were obtained by multiplying the range area with the density estimates for water shrews in Table 5.4a. The length of waterways within the species' range is provided, but was not used in the current analysis.

Country	Area within species' range (ha)	Population size	-95%CI	+95%CI
England	11,780,000	[458,000]	147,000	1,228,000
Scotland	2,580,000	[118,000]	30,000	353,000
Wales	1,750,000	[137,000]	60,000	361,000
Britain	**16,115,000**	**[714,000]**	**237,000**	**1,942,000**

Critique

The population size estimate for the water shrew is based on the ratio with the common shrew. It is therefore subject to the same errors as the common shrew population size estimate, in addition to uncertainties in the ratio. The use of a ratio as a method is questionable because of the differing habitat preferences of the species.

Fifty-six percent of the population size for common shrews (on which the water shrew estimate is based) is derived from unimproved grassland and bog. The average density estimate for common shrews in unimproved grassland is supported by 27 replicate density estimates, and the estimate for bog is based on data from Shore and Mackenzie (1993), which was provided by the authors. Reliability scores of two and zero were applied to the population size estimates for these habitats, respectively, and the overall confidence limits for common shrews are wide (6 million to 58 million), reflecting the uncertainties associated with all stages of the analysis. The ratio of the common shrew to the water shrew was based on bird of prey pellet analysis, and bottle and trap samples from England, Scotland and Wales (see Tables 5-8 in Harris et al., 1995). Although the sample size for these ratios is high (42 papers and one value derived from expert opinion), they do not take into account the high variation in population density and patchy distribution of water shrews though to occur in the British population (Harris et al., 1995).

Harris et al. (1995) estimated the population size of water shrews to be 1,900,000, with 1,200,000 in England, 400,000 in Scotland and 300,000 in Wales. These estimates, however, are not adjusted to take into account the smaller distribution of the water shrew compared to the common shrew. Instead, in the current review, the population size in Britain was computed by dividing the population size of the common shrew by the ratio of the common shrews to the water shrew per country. Reassessment of the data from Harris et al. (1995) using the method presented here suggests a total population size in Britain of 1,500,000 ((common shrew population size/current common shrew distribution area/ratio of common shrew to water shrew) * area of water shrew distribution), although this method assumes that the species' range has remained constant since 1995.

Table 5.4c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All habitats
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**0**
		Overall reliability score	**0**

Changes through time

Comparison to Harris et al. (1995)

Population sizes were estimated by Harris et al. (1995) to be 1,900,000 in Great Britain, comprising 1,200,000 in England, 400,000 in Scotland and 300,000 in Wales. The current review applied the same approach as Harris et al. (1995), using the ratio of the common shrew to the water shrew. Both the current and previous estimates for the common shrew are uncertain, being based on few density estimates, and share the same sources of error. The population size of 1,500,000 estimated by Harris et al. (1995), however, falls within the confidence limits of the current estimate. Further surveys are needed to increase confidence in the estimated population sizes and allow for an assessment of trends.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature search. A summary of trends in population size and range is provided in Table 5.4d.

Table 5.4d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	England Wales*	Scotland		

* Increase in range may be the result of increased survey effort.

Drivers of change

Table 5.4e Drivers of population change for the water shrew between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Habitat availability.	Bank clearance and modification may destroy burrows and alter water supplies. Effects of wide-scale alterations unknown.	Howie and Stokes (2003)	Negative
None quantified, though there are potential impacts of declining invertebrate abundance.	Alterations in farming practice and potentially also use of pesticides.	None	Negative

Data deficiencies

Table 5.4f Areas where further research is required to improve the reliability of population size estimates for the water shrew.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	All habitats	Population size estimates are based on ratios with wood mice.
No occupancy data.	All habitats	

Future prospects

Table 5.4g An assessment of the future prospects of the water shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

5.5 Lesser white-toothed shrew *Crocidura suaveolens*

Habitat preferences

The lesser white-toothed shrew is the only shrew species present on the Isles of Scilly, where it is found on all of the larger islands. It makes use of all habitat types, as long as adequate cover can be found. Common in habitats with tall vegetation such as bracken, hedgerows and woodlands (Harris and Yalden, 2008), it is also reported to use foreshores of the Isles of Scilly (Temple and Morris, 1997). Males have larger home ranges than females, although both are less than 100m in length (Harris and Yalden, 2008).

Status

Non-native (naturalised), but possibly native.

Little information is available. The species is considered most likely to have been introduced in the Bronze Age, but it could have been present since before the last Ice Age as the Isles of Scilly are adjacent to the western extent of its range. Genotyping to assess phylogeny has not been conducted.

Conservation Status

- IUCN Red List (GB: NT; England: [NT]; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

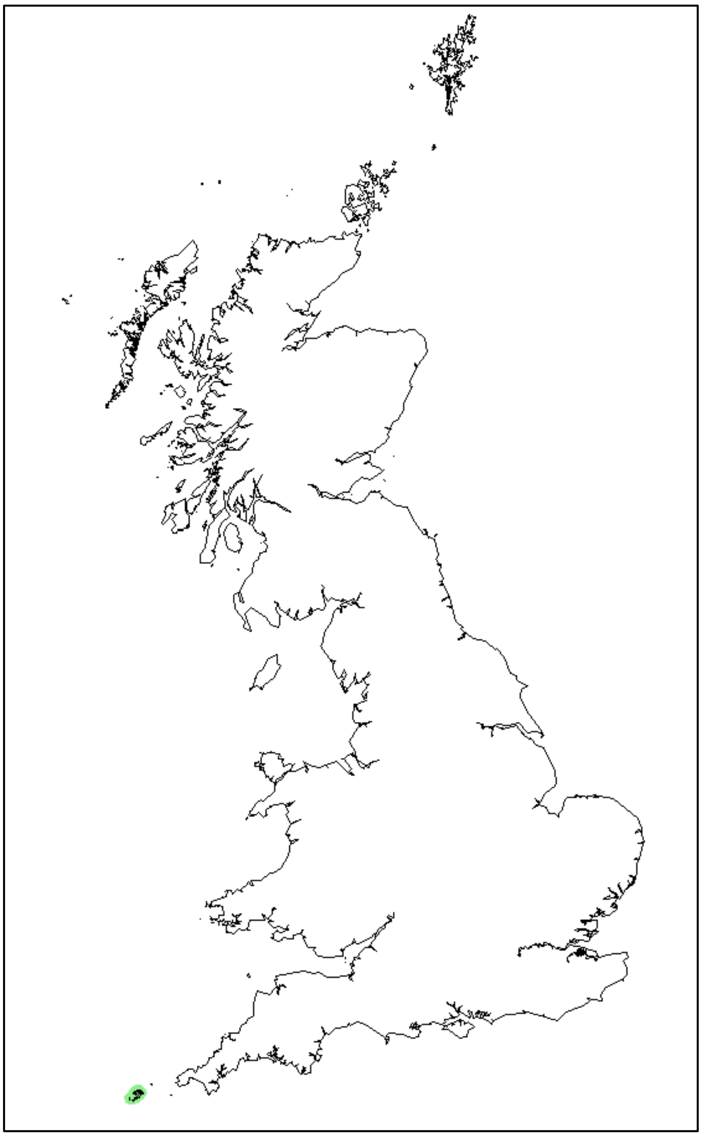


Figure 5.5a Current range of the lesser white-toothed shrew in Britain. Range is based on presence data collected between 1995 and 2016.

Species-specific methods

The species is a generalist, and therefore all available natural habitat was considered suitable on the islands where the species is recorded.

Results

No papers were identified by the literature for population density or occupancy. The population density estimates are therefore taken from Harris et al. (1995).

Table 5.5a Median density estimates with 95% confidence intervals for lesser white-toothed shrew, calculated using data obtained from Harris et al. (1995).

Habitat	Area within range	Density	-95%CI	+95%CI	Source*	n**	%Occ†
Shoreline	70km	100 (km^{-1})	-	-	Harris et al. (1995)	1	n/a
Hedgerows	100km	20 (km^{-1})	-	-	Harris et al. (1995)	1	n/a
All other natural habitats	500 ha	10 (ha)	-	-	Harris et al. (1995)	1	n/a

* Literature sources.

** Number of estimates from each literature source.

Table 5.5b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 5.5a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha) *	Population size	-95%CI	+95%CI
England	500	[14,000]	n/a	n/a
Britain	**500**	**[14,000]**	**n/a**	**n/a**

* Excluding linear features.

Critique

No percentage occupancy data were available; the population size may therefore be overestimated. However, footprint tunnel surveys in 2016 recorded animals in all surveyed habitats (foreshore, coastal grassland, scrub and heathland) (Steve Adams, *pers. comm.*). The density estimates are very out of date, and may have altered following a reduction in predation pressure (namely, recent rat eradication and a decline in the number of domestic cats).

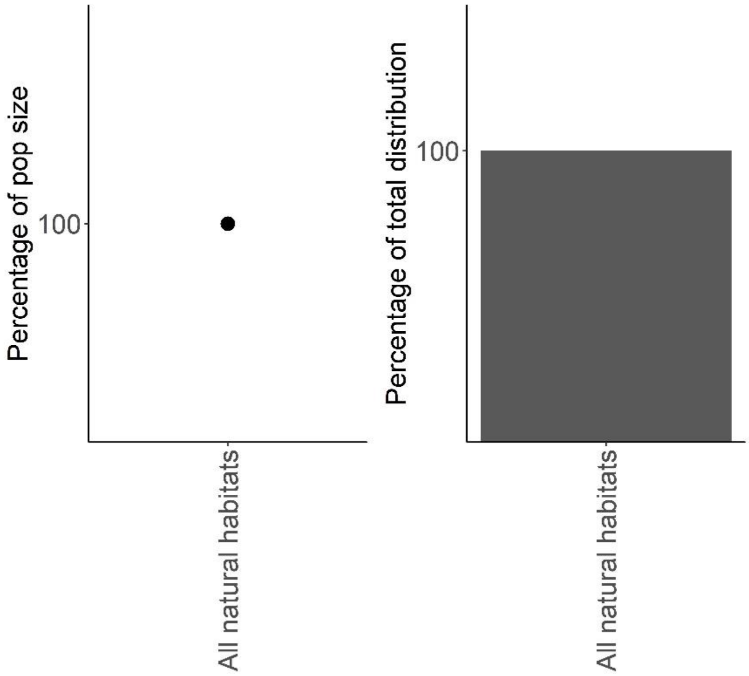


Figure 5.5a Left: The percentage of the total population of common shrews accounted for by each habitat type. Error bars could not be calculated for this species. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows and shore-line) have been omitted.

Table 5.5c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**		
			Shoreline	**Hedgerow**	**All other natural habitats**
Location of study sites	0	Estimates from one location	0	0	0
	1	Estimates restricted			
	2	Estimates widespread			
Sample size	0	<10 density estimates	0	0	0
	1	10-30 density estimates			
	2	>30 density estimates			
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**0**	**0**	**0**
		Overall reliability score	**0**		

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated population size as 14,000 using the same information on habitat availability and density. It is therefore not possible to infer any trends over time, since both reports are subject to the same errors.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature search. However, monitoring on St Agnes, Gugh and Bryher has been conducted since 2013 following rat eradication. This has shown an increase in the proportion of occupied footprint tunnels on St Agnes and Gugh. On Bryher, there was a rapid decline in 2014 and only a partial recovery since then.

Table 5.5d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995), together with the footprint tunnel monitoring project, and trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993)..

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable		England		
	Decrease				
	Data deficient				

Drivers of change

Table 5.5e Drivers of population change for the lesser white-toothed shrew between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Predation.	Rat eradication has been successful, and the numbers of domestic cats are declining.	RSPB/Steve Adams (*pers. comm.*)	Positive

Data deficiencies

Table 5.5f Areas where further research is required to improve the reliability of population size estimates for the lesser white-toothed shrew.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	All	Density estimates are from Harris et al. (1995).
No occupancy data.	All habitats	
Inadequate density estimates.	All except foreshore and hedgerows	A single density estimate was used for all habitats.

Future prospects

Table 5.5g An assessment of the future prospects of the lesser white-toothed shrew, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

6 LAGOMORPHA

6.1 European rabbit *Oryctolagus cuniculus*

Habitat preferences

The rabbit is found in a wide variety of habitats, but prefers those with short grass such as improved grasslands or arable areas. Peaks in population size are found in areas with sandy soils and chalk, as opposed to clay soils (Cowan, 1991; Harris et al., 1995).

Rabbit density is positively associated with livestock grazing, owing to the higher nitrogen content of grazed swards (Iason et al., 2002; Bakker et al., 2005; Lush et al., 2014). More specifically, the rabbit prefers shorter grass swards (Smith et al., 2005; Petrovan et al., 2011a) with low plant diversity (i.e. intensively grazed pasture; Lush et al., 2014), and with predator control measures in place. Livestock production in the UK is in decline (UK National Ecosystem UK National Ecosystem Assessment, 2011). However, several features of less intensively managed landscapes, such as the presence of field margins, hedgerows, and woodland, are also beneficial (Trout et al., 2000; Petrovan et al., 2011a). These features presumably offer the necessary cover to escape predators (Iason et al., 2002), while still providing high quality forage (Bakker et al., 2005).

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: NT.).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

The status of rabbits as ‘near threatened’ on the IUCN Red List is based on the species’ native range, which does not include Britain.

Species' distribution

A distribution map is presented in Figure 6.1a. Gaps in the species' distribution in Scotland are likely to represent a lack of survey effort, rather than true absences.

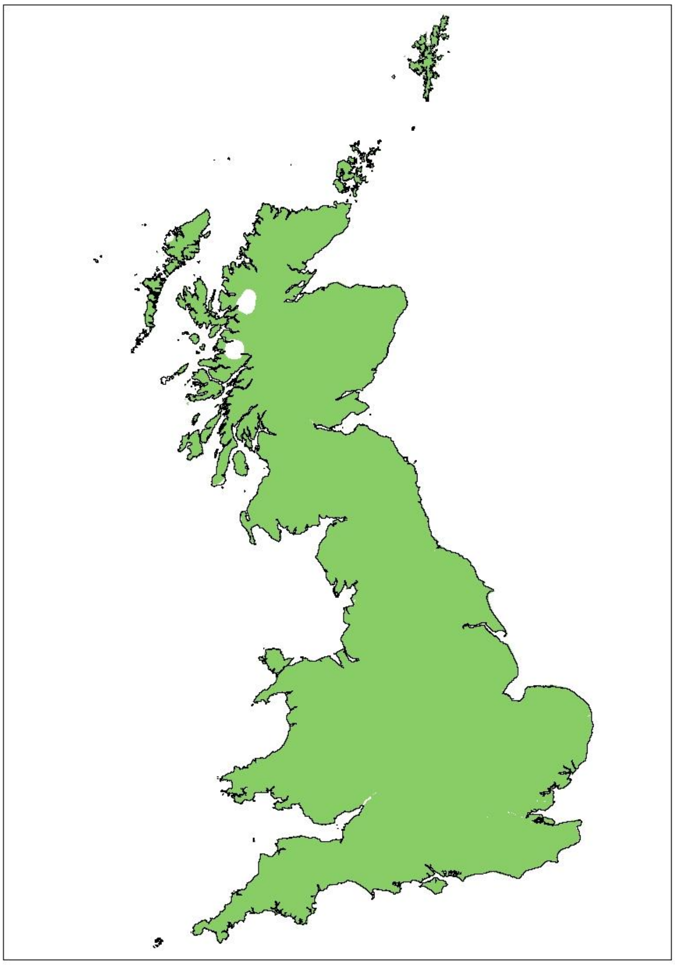


Figure 6.1a Current range of the European rabbit in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

Five papers were identified by the literature search: one contained estimates of pre-breeding population density; two provided post-breeding estimates; one provided an index of rabbit abundance; and one provided evidence of a temporal trend.

Population density estimates per habitat are provided in Table 6.1a, and total population size estimates in Table 6.1b.

Table 6.1a Median density estimates with 95% confidence intervals for European rabbits, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^2)	-95%CI	+95%CI	Source*	n**	%Occ†
Improved grassland	73,100	48.3	26.3	600	Petrovan et al. (2011a) ††	7	n/a
Arable and horticulture	62,600	250	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	13,100	200	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	14,400	200	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	19,700	250	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	12,400	500	-	-	Harris et al. (1995)	1	n/a
Supra-littoral rock	32	500	-	-	Harris et al. (1995)	1	n/a
Supra-littoral sediment	300	500	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
†† Raw data were supplied by the authors.

Table 6.1b Area of suitable habitat (not adjusted for occupancy) within the species' range. Values were obtained by multiplying population density estimates with the area of habitat within the species' distribution. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	115,000	[21,300,000]	-	-
Scotland	61,300	[11,800,000]	-	-
Wales	19,100	[2,910,000]	-	-
Britain	**196,000**	**[36,000,000]**	**-**	**-**

Critique

No percentage occupancy data were available, so the population size for this species is overestimated. 42% of the population is attributed to arable and horticultural land, for which the density estimate is taken from Harris et al. (1995). Most of the land within the species' range consists of arable and horticulture (32%) and improved grassland (37%).

The density estimates derived from Harris et al. (1995), and hence the overall population size estimates, are somewhat at odds with the known preference of rabbits for areas with short grass swards and low plant diversity, such as improved grassland (Lush et al., 2014). For example, the densities given for unimproved grassland and coniferous woodland are 500 rabbits km^{-2} and 200 rabbits km^{-2} respectively, whereas the recent estimate for improved grassland is only 48 rabbits km^{-2}. However, given the extreme variability in rabbit abundance, a large sampling effort is required to produce robust evidence on median densities with reasonable precision. Therefore, using a single central estimate for each study is likely to introduce considerable error. The density estimate for improved grassland is based on a relatively small amount of data (one paper with seven replicates; Petrovan et al., 2011a), resulting in 95% confidence limits of $30km^{-2}$ to $60km^{-2}$. These densities were estimated using spotlight counts, which may only represent ~60% of the total number of rabbits within the unit area (Poole et al., 2003; Petrovan et al., 2011a), although this percentage may vary greatly depending on the area under study. Further data would provide a more robust estimate, but it is likely that a density at the upper confidence limit is more representative for this habitat type. The estimate for improved grassland was $250km^{-2}$ in Harris et al. (1995), which though considerably higher, falls within the confidence limits.

Estimates from Harris et al. (1995) were based on the authors' adjustment of over-wintering population estimates from high density areas.

Factors such as outbreaks of myxomatosis and rabbit haemorrhagic disease have severe local impacts (Petrovan et al., 2011b), but these are rather poorly understood on national scales. The long-term impacts of these diseases on populations are also unclear (beyond anecdotal evidence of regional recoveries from previous population crashes), making it particularly difficult to extrapolate from historical evidence on habitat-specific densities. In addition, rabbit populations are inherently highly variable, even in the absence of disease, so there is considerable uncertainty in density estimates both within and between habitat types. The application of a single median density (current method) or single adjusted density (Harris et al., 1995), particularly where data are limited, may not, therefore, result in a reliable population size estimate. A stratified survey approach, using different geographical regions and habitat types, would improve the current estimates.

Stepwise deletion and replacement of each of the seven replicates for improved grassland resulted in two alternative population size estimates, both of which differed from the original by 29% (46,000,000), although the lack of confidence limits for the original population size estimates means that the significance of this cannot be formally assessed.

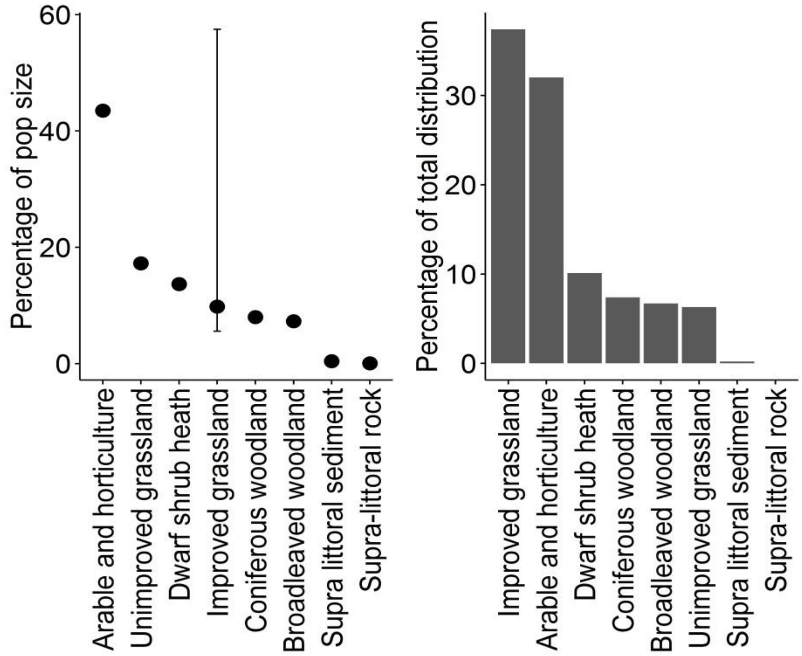


Figure 6.1b Left: The percentage of the total population of European rabbits accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 6.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	Improved grassland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	0
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

Changes through time

Comparison to Harris et al. (1995)

Population size estimates in Harris et al. (1995) were 37,500,000; 24,500,000 in England, 9,500,000 in Scotland and 3,500,000 in Wales. The density estimates for the majority of habitats were taken from Harris et al. (1995), so a comparison of population sizes is limited to differences in range size and habitat availability.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a 9% reduction in population size. The lack of confidence limits around the current estimate means that the significance of this reduction is unclear.

Other evidence of changes through time

The National Gamebag Census reported a decrease of 24% (95%CI 45% decrease to 4% increase) in the number of rabbits culled between 1995 and 2014 in the UK. The NGC survey, however, does not account for effort, so it may not represent a true decline in population size.

The Breeding Bird Survey inferred a population decrease of 48% (95% CI 56%-33%) between 1995 and 2012 in the UK (Wright et al., 2014). Two experts also suggested that the population may have declined in size, but highlighted that the situation is complex because of the wide range of rabbit densities present in different areas, and uncertainties about the rate of change.

Table 6.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		All countries*		
	Data deficient				

* Based on BBS trend data and expert opinion.

Drivers of change

Table 6.1e Drivers of population change for the European rabbit between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Disease/pesticides.	Recovery from myxomatosis since the 1950s epidemic.	Petrovan et al. (2011a)	Positive
	The impact of rabbit haemorrhagic disease.	Petrovan et al. (2011a)	Negative
Management (control).	Rabbits are culled where damage is caused to agriculture. This driver is, however, complex, and culling effort may have been reduced owing to lower demand for rabbit meat or fur.		Negative

Data deficiencies

Table 6.1f Areas where further research is required to improve the reliability of population size estimates for the European rabbit.

Data deficiencies	Habitat	Details
Density estimates are >10 years old.	All habitats except improved grassland	Density estimates were taken from Harris et al. (1995).
Limited density estimates.	Improved grassland	One paper contained density estimates with fewer than 10 replicates. Sensitivity analysis suggests that the population estimate is highly dependent on two data points.
Density estimates do not represent within-habitat variability.	All habitats except improved grassland	No range or confidence limits were available for the density estimates.
Density estimates do not represent within-habitat variability.	All other habitats	It was not possible to calculate confidence intervals owing to lack of data. Rabbit populations are known to be highly variable within and between habitats.
No occupancy data.	All habitats	

Future prospects

Table 6.1g An assessment of the future prospects of the European rabbit, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline*
Range	Stable
Habitat	Decline

* Note that much of the expected decline is because of disease impacts, and there is uncertainty about future epidemiological patterns and population impacts.

6.2 Brown hare *Lepus europaeus*

Habitat preferences

The brown hare is found mainly in lowland arable or pastoral land. It is frequently found in open areas, where it uses its swift running speed to evade predators, but it also requires shelter for resting and breeding. Its abundance is positively associated with habitat and plant species diversity, the presence of hedgerows, and unfarmed habitat (Tapper and Barnes, 1986; Smith et al., 2004; Lush et al., 2014), and negatively associated with grazing intensity. The brown hare selects habitat based on structure, rather than the availability of nutrients, which is thought to result from the need for cover from predators and for surface resting sites (Smith et al., 2004; Lush et al., 2014). Population density may be adversely affected by shooting, competition with farm livestock for food, and the intensification of agricultural practices (Hewson and Hinge, 1990).

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

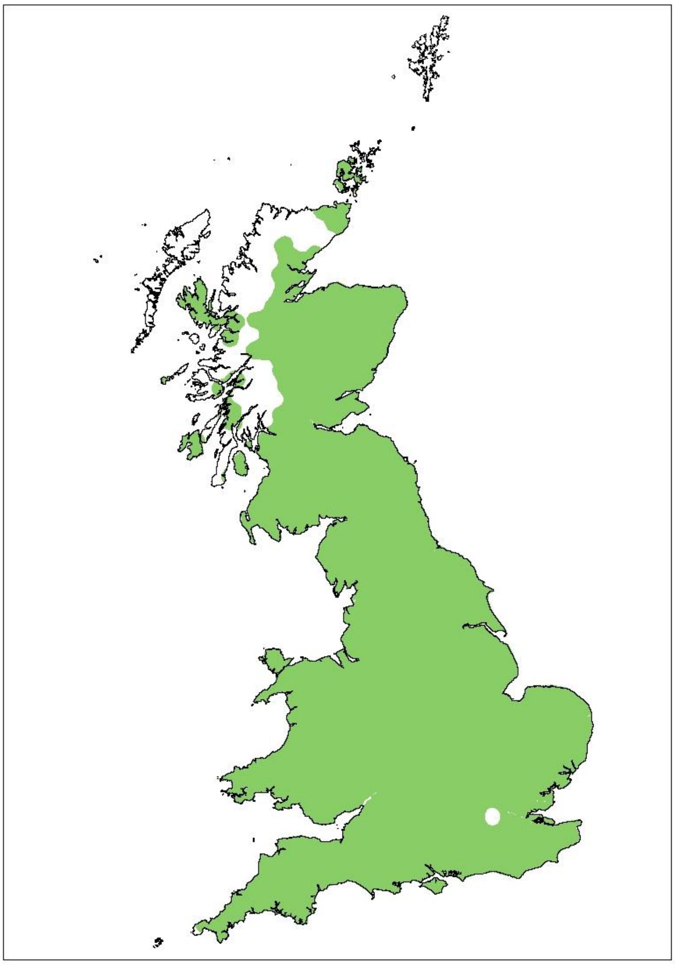


Figure 6.2a Current range of the brown hare in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Population size estimates were based on density data for arable habitat and improved grassland only. Although brown hares may be found elsewhere, arable land and improved grassland constitute their core habitat. The use of occurrence data from other habitats was judged likely to introduce double counting, because most animals observed here would already be accounted for in the arable and grassland data.

Results

Twelve papers and one NGO report were found for the brown hare, where seven contained pre-breeding estimates of population size, five contained post-breeding estimates or temporal trend data, and one contained occupancy data (Hutchings and Harris, 1996). Percentage occupancy was measured as the number of positive 1km survey squares (456 of 738); habitat-specific occupancy values were not available so this percentage was applied to all habitats. Population density estimates are provided in Table 6.2a, and population size estimates in Table 6.2b.

Table 6.2a Median density estimates with 95% confidence intervals for brown hares, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	62,500	11.0	8.25	29.6	Bradshaw (1993)	2	61.8
					Heydon et al. (2000)	6	
					Hutchings and Harris (1996)	2	
					Rothschild and Marsh (1956)	2	
					Temple et al. (2000)	2	
Improved grassland	68,800	3.63	2.55	26	Bradshaw (1993)	2	61.8
					Heydon et al. (2000)	1	
					Hutchings and Harris (1996)	2	
					Parrott et al. (2012)	5	
					Petrovan et al. (2011b)	2	
					Rothschild and Marsh (1956)	2	
					Temple et al. (2000)	2	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 6.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 6.2a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	93,900	454,000	336,000	1,480,000
Scotland	24,300	87,700	64,000	342,000
Wales	13,100	37,300	26,800	171,000
Britain	**131,000**	**579,000**	**427,000**	**1,990,000**

Critique

The population estimate for brown hares is based on densities in arable and horticultural habitats (73%), and in improved grassland (27%; Figure 6.2b). Twelve and 14 individual estimates, respectively, were obtained for these habitat types. Although population size was adjusted to reflect occupancy, these data are not habitat-specific; the brown hare population is also patchily distributed throughout the country, with 19% of occupied squares found in three counties (Hutchings and Harris, 1996). Percentage occupancy data are therefore likely to introduce inaccuracies when applied to the whole country. A reliability assessment is provided in Table 6.2c.

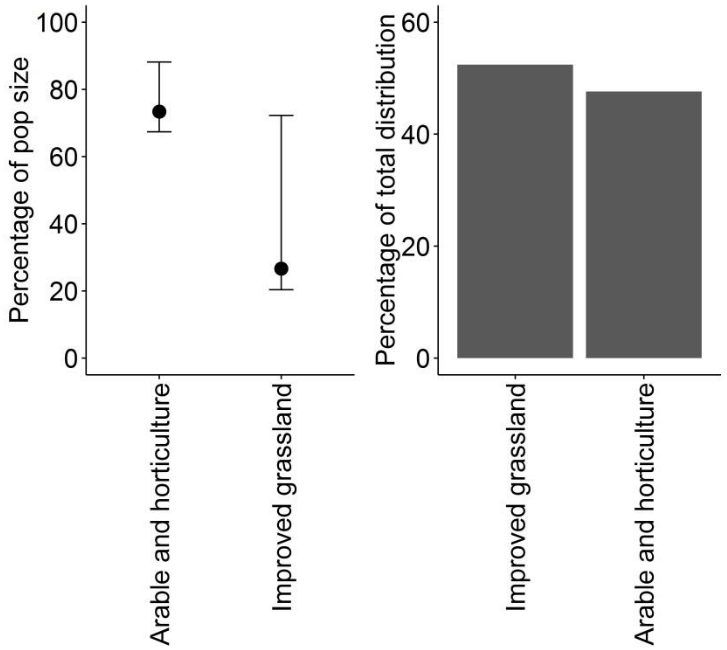


Figure 6.2b Left: The percentage of the total population of brown hares accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 6.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	**Improved grassland**
Location of study sites	0	Estimates from one location		
	1	Estimates restricted*	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates		
	1	10-30 density estimates	1	1
	2	>30 density estimates		
Occupancy data available?	0	No		
	1	Yes	1	1
		Habitat score	**3**	**3**
		Overall reliability score	**3**	

* Although studies covered a wide geographical range, they did not capture the extremely large variability in density and occupancy known to occur between eastern and western counties of England. Therefore, a score of 1 has been allocated to this factor.

Changes through time

Comparison to Harris et al. (1995)

In Harris et al. (1995), population size estimates were 817,000 in total, comprising of 572,000 in England, 187,250 in Scotland, and 58,000 in Wales. These estimates were provided by the first National Brown Hare Survey, which was later published by Hutchings and Harris (1996). These figures were derived from extensive surveys and subsequent population density estimates, grouped by land class rather than habitat type, and included occupancy data by incorporating sampled areas with no detection of hares. By contrast, the current estimate is based on the two main habitats for brown hares only (see 'Species-specific methods').

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size only 5% different from the original (and within the original confidence limits). These methodological differences are therefore unlikely to affect comparisons between the two reviews materially.

Although the population size estimated by Harris et al. (1995) is higher than the current estimate, it falls within the confidence limits. This fact, coupled with the differences in methodology and habitat associations, means it is difficult to make direct comparisons between the current review and that of Harris et al. (1995).

Other evidence of changes through time

The National Gamebag Census reports a 38% (95%CI = 3%-76%) increase in brown hares shot for game from 1995 to 2009. This trend may not, however, represent a change in the absolute population size, as survey effort may have differed between years.

Following the first National Brown Hare Survey, which estimated brown hare population size to be 817,000 in 1991-1993 (Harris et al., 1995; Hutchings and Harris, 1996), a second survey in 1997-1999 (Temple et al., 2000) estimated the population to be 752,608 (95%CI = 714,911-790,305). As this decline is relatively small, its significance is difficult to determine, particularly given the lack of confidence limits for the former estimate. Even if relatively tight confidence limits are assumed, the difference is unlikely to be significant. A summary of trends in population size and range is provided in Table 6.2d.

Table 6.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 6.2e Drivers of population change for the brown hare between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Changes to agricultural practices, particularly the switch towards winter wheat, may reduce food and shelter opportunities. Competition with livestock for food.	Hewson and Hinge (1990)	Negative
Management (control).	Suppression of the population during specified times of year.	Tapper and Parsons (1984)	Negative
Climate change.	Alteration of agricultural practice, particularly the switch away from arable crops in eastern counties, may be detrimental to hare populations, although potentially offset by increases in arable production in the north and west.	Fezzi et al. (2014)	Negative

Data deficiencies

Table 6.2f Areas where further research is required to improve the reliability of population size estimates for the brown hare.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Arable and horticulture	The most recent density estimate is from Heydon et al. (2000).
Managed populations.	Arable and horticulture	Population management is not taken into account in the population size estimate.
Occupancy and density estimates.	All	It is not currently possible to adjust for known variability in density and occupancy between eastern and western counties owing to a lack of data.

Future prospects

Table 6.2g An assessment of the future prospects of the brown hare, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Decline

6.3 Mountain hare *Lepus timidus*

Habitat preferences

The mountain hare in Scotland is found primarily on heather moorland at an altitude of 300m to 900m (Flux, 1970). In the Peak District, the species was introduced in the late 19th century for sport, having been extinct during historical times (Anderson and Yalden, 1981), and is found primarily in areas of common heather and cotton-grass (Harris and Yalden, 2008). During the day, it uses resting sites at higher altitudes, creating forms in areas with extensive cover. At night, it travels to lower ground to use hill pastures, areas of wild grassland or high altitude moorland to feed: the preferred food is the current year's growth of young heather (Hewson and Hinge, 1990). Where available, woodland is often used for shelter (Thirgood and Hewson, 1987), protection from predators and additional food sources (Patton et al., 2010). The home ranges of mountain hares are notably bigger than those of brown hares. Day resting sites and night feeding sites are often located a relatively large distance apart, owing to the patchy distribution of food and suitable resting places (Hewson and Hinge, 1990). Population densities are higher on moorland overlying base-rich than acidic rock, and in the east of Scotland compared to the west. The highest densities occur on moorland managed for grouse (Harris et al., 1995).

Status

Native.

Conservation Status

- IUCN Red List (GB: NT; England: n/a; Scotland: [NT]; Wales: n/a; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. UK: Favourable; England: Favourable; Scotland: Favourable; Wales: n/a).

Species' distribution

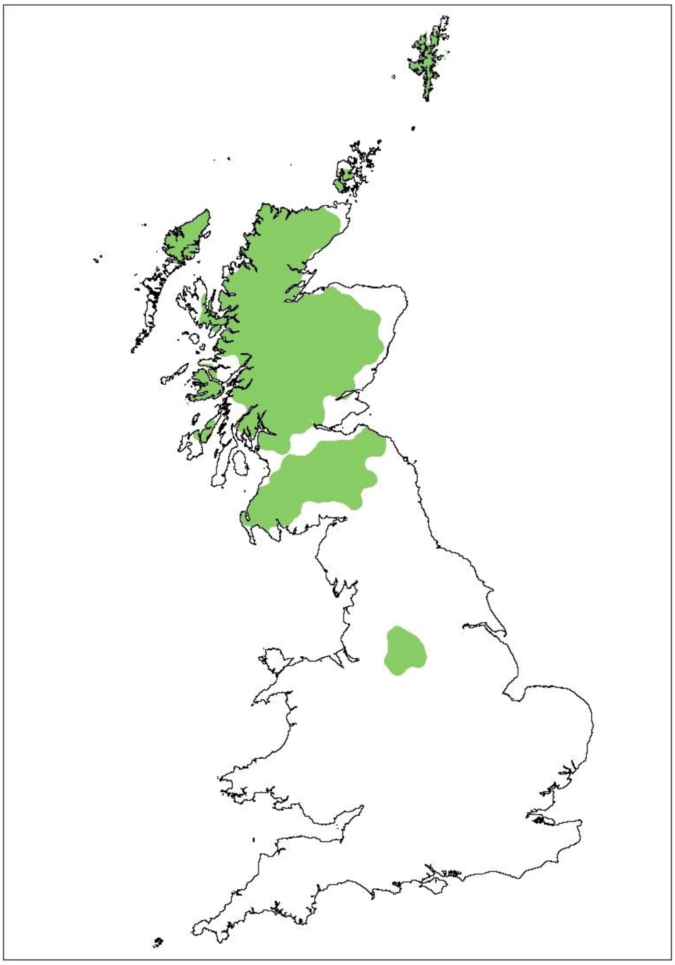


Figure 6.3a Current range of the mountain hare in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Population density estimates were available for dwarf shrub heath and montane habitats. Density estimates for montane habitats were provided during expert consultation.

In the LCM2007, montane habitats are defined as any area above 600m in all areas north of the Midlands in England, regardless of habitat type. Considering the relatively small home range of mountain hares and their requirement for heather moorland, montane habitats in this context were not considered well enough defined to include in the current analysis. Population size is therefore estimated for dwarf shrub heath only.

Results

Eight papers were identified by the literature search: two papers reported pre-breeding population estimates; two gave post-breeding estimates only; two contained details of the species' distribution; and two provided details of relative changes in population density or home range size. Newey et al. (2003) estimated density using both distance sampling and capture-mark-recapture (CMR) at two of the four sites studies (with distance sampling used at the two remaining sites); smaller differences in density estimated using CMR were observed between the two sites than density estimated from distance sampling, suggesting trap saturation at the site with higher density. Distance sampling estimates were therefore deemed more accurate and were used in the current analysis. Population density estimates are shown in Table 6.3a, and population size estimates in Table 6.3b.

Table 6.3a Median density estimates with 95% confidence intervals for mountain hares, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Dwarf shrub heath	13,500	10	6	39	Newey et al. (2003) Knipe et al. (2013)	4 9	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 6.3b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 6.2a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size*	-95%CI	+95%CI
England	250	2,500	1,500	9,500
Scotland	13,200	132,000	79,500	516,000
Britain	**13,500**	**135,000**	**81,000**	**526,000**

*Population size is for dwarf shrub heath only.

The Article 17 Report on mountain hare population size 2007-2012 is shown in Table 6.3c (Joint Nature Conservation Committee, 2013b). The estimate from the current review is very much smaller, though the geographical range is similar (Table 6.3d).

Table 6.3c Article 17 Report on mountain hare population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	10,000	10,000
Scotland	350,000	350,000
Wales	n/a	n/a
Britain	**360,000**	**360,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

Table 6.3d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013b).

Country	**Extent of occurrence (km^2)**	**Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)**
England	2,400	n/a
Scotland	57,400	n/a
Wales	0	n/a
Britain	**59,800**	**62,970**

Critique

No percentage occupancy data were available; the population size is therefore overestimated for this species. The estimate is also derived from just one habitat type (dwarf shrub heath). The exclusion of areas classified as montane habitat means that all areas above 600m are removed, regardless of habitat type. Areas of heather moorland do, however, exist above 600m in the central and eastern Highlands. As mountain hares range up to 900m, a cut-off of 600m will certainly have excluded some occupied suitable habitat. In the north west of Scotland, the montane zone can descend to 300-400m, while in the east it can ascend above 600m. In the north west of Scotland, therefore, some montane habitats will be included in the LCM ‘dwarf shrub heath’ category, while in the east, some non-montane heaths will be excluded. Sole use of dwarf shrub heath habitat for the estimates may also exclude extensive areas of moorland dominated by grasses (mainly in the western Highlands), although mountain hares are present at low density in these areas.

The population density of mountain hares is highly variable under differing environmental conditions within dwarf shrub heath, with particularly high densities in moorland managed for grouse shooting (around 30-69km^{-2}, but exceptionally 200km^{-2} or more (Harris and Yalden, 2008)), as well as in the eastern parts of Scotland compared to the west coast. The population density estimates used in the current assessment are all taken from one location on moorland managed for grouse in the central Highlands. These estimates do not, therefore, represent the range of densities likely to be found over the species’ distribution, but are based on areas with favourable habitat. Despite the considerable uncertainty surrounding the estimates of population size, surveys in the Peak District National Park suggest a population size of 1,500-5,000 in England (Thomas Rhodri, Peak District National Park Authority, *pers. comm.*) which accords with our estimate. Reliability scores are provided in Table 6.3e.

Table 6.3e Reliability assessment for each habitat representing >25% of the species’ distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Dwarf shrub heath
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 density estimates	
	1	10-30 density estimates	1
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**2**
		Overall reliability score	**2**

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated the total population size to be approximately 350,500, comprising 350,000 in Scotland and 500 in England. This estimate was based on mean densities of $2km^{-2}$ on the Scottish Islands and north west of the Great Glen, and $20km^{-2}$ within the rest of their range. These densities were not habitat-specific, but applied to the occupied area within the species' range, where 50% of the area was assumed to be occupied. As the current estimate is based on dwarf shrub heath only, a comparison of population sizes between the two time periods is not meaningful because of methodological differences.

Some mountain hare populations fluctuate in cycles of approximately 9 years, although this time period is subject to variation (Newey et al., 2007). Our assessment does not provide a population size at a defined point in this cycle, so an assessment of the population size against a single previous estimate would not provide any meaningful information on the overall trend of the mountain hare population.

Other evidence of changes through time

The GWCT National Gamebag Census found a decrease of 40% (95%CI 68% decrease to 20% increase) between 1995 and 2009 (Aebischer et al., 2011), although the trend is non-significant. A summary of trends in population size and range is provided in Table 6.3f.

Table 6.3f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	Scotland		England	

Drivers of change

Table 6.3g Drivers of population change for the mountain hare between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat loss.	Altered land use and fragmentation can result in the loss of foraging opportunities and shelter, which may be detrimental to survival.	Patton et al. (2010)	Negative
Management (control).	Hares are hunted for sport, to reduce damage to forestry, and in the belief that they contribute to the transmission of disease to grouse.	Newey et al. (2008) Patton et al. (2010)	Negative
Hybridisation and competitive exclusion.	Hybridisation and competitive exclusion may become a threat where ranges overlap.	Thulin et al. (2003)	Negative

Data deficiencies

Table 6.3h Areas where further research is required to improve the reliability of population size estimates for the mountain hare.

Data deficiencies	Habitat	Details
Density estimates do not represent within-habitat variability.	All habitats	Density is highly variable, and can, exceptionally, reach 200km^{-2} in moorland managed for red grouse.
No density estimates for the specified habitat.	All except dwarf shrub heath	A lack of habitat-specific density estimates, coupled with difficulty in aligning densities to LCM2007 habitat categories, reduces the certainty of population estimates.
Multiannual population cycles.	All habitats	Around 50% of populations exhibit 9-year population cycles.
No occupancy data.	All habitats	

Future prospects

Table 6.3i An assessment of the future prospects of the mountain hare, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline*
Range	Stable
Habitat	Decline

* Decline is assumed because of the likely decline in habitat quality.

7 RODENTIA

7.1 Red squirrel *Sciurus vulgaris*

Habitat preferences

The red squirrel occurs in both conifer and broadleaved woodland, as well as in mixed forests and parks and gardens (Harris and Yalden, 2008). It eats a wide range of foods, but tree seeds and fruits are particularly important, followed by tree shoots, buds, flowers, berries and lichens (Moller, 1983; Gurnell et al., 2015a). Woodlands with mixtures of tree seeds provide a more reliable year-to-year food supply. In mixed conifer forests, home range selection is based on the availability of seed from different species throughout the year (Lurz et al., 2000). Sitka spruce, which is widely planted in managed woodlands, has unreliable fruiting cycles, and there is a negative relationship between the proportion of Sitka spruce in woodlands and the density of red squirrels (Lurz et al., 1998).

Status

Native.

Conservation Status

- IUCN Red List (GB: EN; England: [EN]; Scotland: [NT]; Wales: [EN]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 7.1a. There has been considerable recording effort in Scotland since the Saving Scotland's Red Squirrels partnership project was launched in 2012, and particularly since the development in 2015 of an interactive website for recording. It is therefore possible to present a detailed distribution map. The gap in the species' distribution in the Central Belt is likely to extend to the west of Glasgow and east towards Edinburgh, with only very sparse records in this region. The intensive survey effort carried out by the Saving Scotland's Red Squirrels project suggests that this gap is real, and not an artefact of the smoothing process used to create the current distribution map. In England, the Red Squirrels Northern England Project has, for over 5 years, helped to stabilise red squirrel populations and enable them to spread outside their strongholds. The presence records in Surrey are from escaped captive animals and not from an established population.

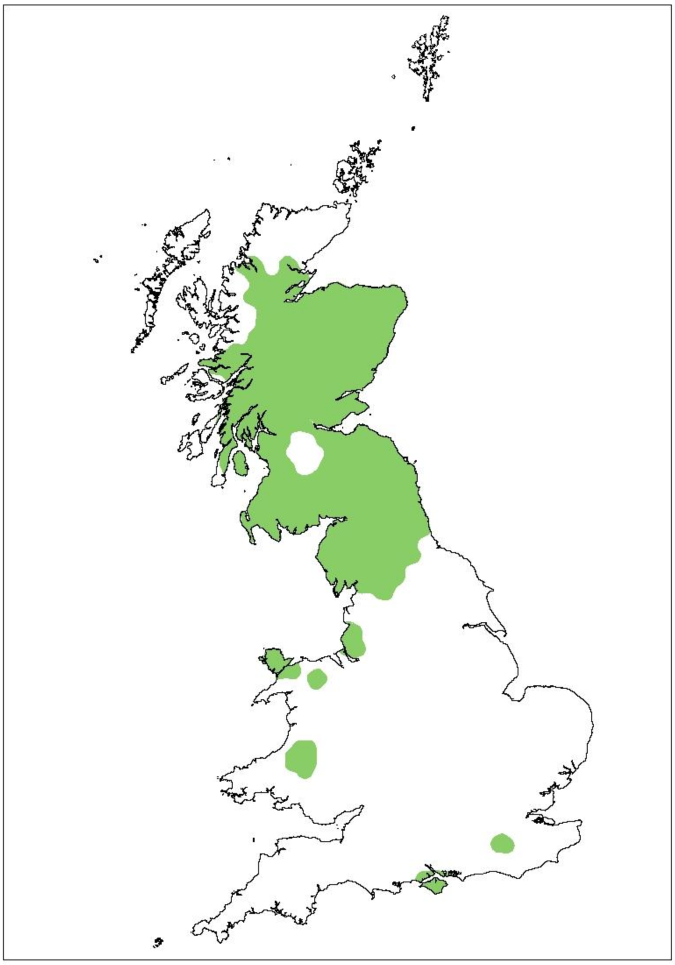


Figure 7.1a Current range of the red squirrel in Britain. To reflect the current distribution and permit assessment of changes in the species' range through time, the maps are based on presence data collected between 2010 and 2016, rather than 1995 and 2016 (the period used for most other species in this review). This is because the distribution has been undergoing rapid flux in response to the spread of grey squirrels. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. Conversely, occasional records from the same area, but which are not derived from an established population, can lead to areas of presence being indicated on the map (for example, the area in Surrey shown as occupied). The distribution of the red squirrel in central Scotland is likely to be less extensive than shown, and may not be contiguous from the south of Scotland to the central Lowlands. This is because there is a series of adjacent hectads to the west of the current gap, and a further two hectads to the east, that each contain only a single record. If these records are erroneous, there will be a break between the northern and southern parts of the red squirrel's range in Scotland.

Species-specific methods

As red squirrels are most likely to occur in mature woodlands, all recently planted (<10 years) and felled woodlands, as defined in the LCM2007, were removed from the analysis.

Results

The literature search returned 19 papers. Of these, 5 contained pre-breeding population density estimates; 5 gave post-breeding density estimates; one contained population sizes but no study areas; one contained estimates using visual counts only, which was deemed unreliable (John Gurnell, *pers. obs.*); and 7 examined the effects of habitat variables or gave data only indicating presence or absence. Population density estimates are provided in Table 7.1a, and population size estimates in Table 7.1b.

Table 7.1a Median density estimates for red squirrels with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (ha)	Density (per ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ †
Broadleaved woodland	331,000	0.23	0.17	0.64	Kenward et al. (1998)	1	n/a
					Cartmel (2000)	5	
Coniferous woodland	849,000	0.25	0.19	0.4	Lurz et al. (1998)	12	n/a
					Cartmel (2000)	4	
					Wauters et al. (2000)	5	
					Bryce et al. (2005)	12	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.1b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Figure 7.1a with the area of habitat in the species' distribution. Small discrepancies between this calculation and the population sizes shown are owing to rounding errors.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	166,000	38,900	29,500	91,000
Scotland	976,000	239,000	181,000	444,000
Wales	38,000	9,190	6,970	18,200
Britain	**1,180,000**	**287,000**	**218,000**	**553,000**

Critique

No percentage occupancy data were available; the population size for this species is therefore overestimated. Most of the population estimate is derived from coniferous woodland (73%; Figure 7.1a). The density estimate for this habitat is based on 33 individual estimates from four papers. Broadleaved woodland contributes the remaining 27% of the population, with the density for this habitat derived from six individual density estimates in two papers (see Table 7.1a). Stepwise deletion of each of these density estimates did not significantly alter the population size, the biggest difference being an increase in overall population size by 8%.

Although young (<10-year-old) coniferous woodland was excluded from the population estimation, there may still be considerable overestimation resulting from the inclusion of extensive Sitka spruce plantations that support only very low red squirrel densities. This is likely to be a particular issue in Scotland and Wales, where Sitka forms 58% and 60% of managed conifer woodlands compared with 25% in England (Forestry Commission, 2014). Given the limited number of habitats contributing to the overall population of red squirrels, additional up-to-date surveys in both broadleaved and different types of coniferous woodland (e.g., Scots pine, lodgepole pine, larch, Norway and Sitka spruce) would be beneficial. If a crude adjustment were made to the population estimates for Scotland by simply excluding Sitka spruce from the calculation of total area of suitable habitat, then the Scottish estimate would become 148,000 (95%CI = 112,000-275,000), and the best estimate for Great Britain would decline by 91,000 to 196,000 animals. Both the Scottish and British revised figures lie outside the confidence limits of the previous estimates. These adjustments are clearly oversimplistic since the original calculations accounted to some extent for the presence of Sitka spruce: the population density estimates used for coniferous woodland included mixed-species woodland that incorporated some Sitka spruce. More realistic future adjustments could characterise the abundance of monoculture Sitka spruce woodlands and score their occupancy as zero, and they could also adjust the density estimates according to the proportion of Sitka spruce present in a woodland. In the interim, it should be concluded that the true population size may well be toward the lower confidence interval presented here.

There is also likely to be overestimation of the population size estimates for England, Wales and the south of Scotland because the range overlap with grey squirrels in these areas is likely to depress population densities below the values used in this report. Reliability assessments per habitat are provided in Table 7.1c.

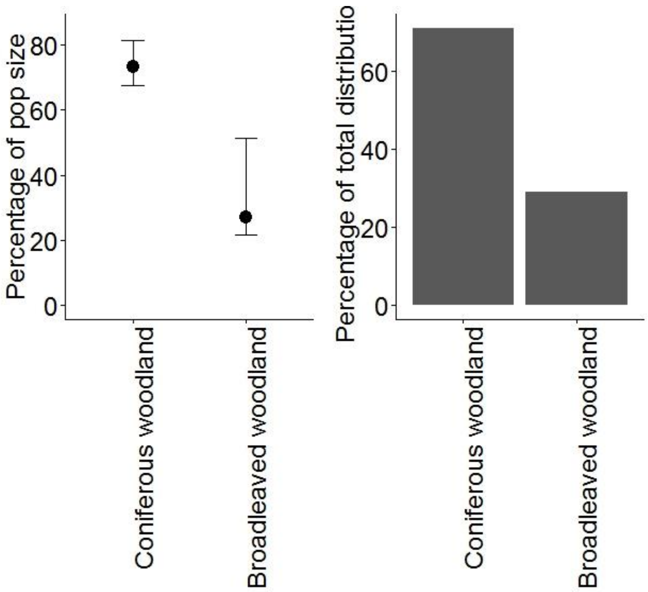


Figure 7.1a Left: The percentage of the total population of red squirrels accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 7.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**	
			Coniferous woodland	**Broadleaved woodland**
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates		0
	1	10-30 density estimates		
	2	>30 density estimates	2	
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**3**	**1**
		Overall reliability score	**2**	

* Populations may be unstable owing to inter-annual cycles, documented fluctuations in population size, or management.

Changes through time

Comparison to Harris et al. (1995) and Arnold (1993)

Population size was estimated by Harris et al. (1995) to be 161,000, with 30,000 in England, 121,000 in Scotland and 10,000 in Wales in 1995. These estimates are based on the median population density of 0.55ha^{-1} in both coniferous and broadleaved woodland, which is almost three times higher than the current median density estimate (Table 7.1a). The population size in Harris et al. (1995) was adjusted for the proportion of woodland greater than 15 years old, and also for occupancy rates. These adjustments were based on expert opinion (John Gurnell, *pers. obs.*) and different values were used per country. The resulting total occupied area was 300,000ha, compared with 1,180,000ha in the current review.

The percentage occupancy values employed by Harris et al. (1995) were not reported, and so could not be applied in our calculations. Considering that the range size of red squirrels has also changed substantially since 1995, the final value for the area occupied could not be used, either. The area in the current estimate was, therefore, not adjusted for occupancy, and totalled 1,180,000ha of woodland within the species' distribution. The lack of occupancy data and the consequently larger area used for the population size calculations precludes direct comparisons of population sizes between the two time periods. These differences also explain why the current estimates of population size appear larger than those given by Harris et al. (1995), despite the evident contraction of the geographical range of the species.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (broadleaved woodland and coniferous woodland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size only 4% larger than the original (and within the original confidence limits). These differences are unlikely to affect the conclusions materially.

The distributions in England and Wales are considerably more restricted than were reported by Arnold (1993; >90% loss in each country), with populations in East Anglia, the Humber Estuary, Derbyshire, Pembrokeshire, Carmarthenshire and Denbyshire having been lost, and those in Lancashire and Gwynedd significantly reduced. In Scotland, the distribution remains approximately as described by Arnold (1993).

Other evidence of changes through time

The distribution of red squirrels is reported to have declined since 1995 (Gurnell et al., 2014). Local grey squirrel control in the north of England and Scotland appears to have stabilised numbers in the last few years (John Gurnell, *pers. obs.*), although there is naturally high variability in red squirrel populations so short-term trends should be interpreted cautiously. A summary of trends in population size and range is provided in Table 7.1d

Table 7.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		Scotland*	England Wales*	
	Data deficient				

* Population trends are from Gurnell et al. (2014), rather than by comparison with Harris et al. (1995).

Drivers of change

Table 7.1e Drivers of population change for red squirrels between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Disease impact.	Widespread population suppression caused by squirrel pox.	Chantrey et al. (2014)	Negative
	Recent emergence of adenovirus.	Everest et al. (2014)	Negative
Competition.	Competition with grey squirrels for resources, leading to reduced recruitment and breeding success.	Gurnell et al. (2004b) Gurnell et al. (2015b)	Negative
Habitat quality.	Pathogenic tree disease affecting, e.g., larch and pine will influence habitat availability and key food resources. There has been a considerable increase in the proportion of Sitka spruce — which is unfavourable for red squirrels — particularly in Scotland over recent decades. Changes in subsidy patterns may reduce this, but there are disease and yield considerations that limit replanting with Scots pine and some other more favourable species.	Lurz et al. (1998) Shuttleworth et al. (2012) Forestry Commission (2014)	Negative
	Climate warming may lead to the planting of new commercial conifer crop species, although decisions are likely to be influenced by disease mitigation.	Gurnell et al. (2015a)	Uncertain
Conservation measures.	Control of grey squirrels has prevented further encroachment into red squirrel range in the Scottish Borders and Aberdeenshire, eradicated them from Anglesey, and has reduced competition where ranges overlap.	John Gurnell (*pers. obs.*)	Positive

Data deficiencies

Table 7.1f Areas where further research is required to improve the reliability of population size estimates of red squirrels.

Data deficiencies	Habitat	Details
Limited density estimates for key habitat.	Broadleaved woodland	There are 5 individual population density estimates.
Density estimates do not represent within-habitat variability.	Broadleaved and coniferous woodland	Population density is likely to be highly variable, depending on woodland condition and other factors not accounted for in the current assessment.
Density estimates are more than 10 years old.	Broadleaved woodland	The most recent estimate is from 2000.
No occupancy data.	All habitats	

Future prospects

Table 7.1g An assessment of the future prospects of the red squirrel, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline*
Range	Decline*
Habitat	Stable

* Rate may slow in response to the control of grey squirrels.

7.2 Grey squirrel *Sciurus carolinensis*

Habitat preferences

The grey squirrel lives in a wide variety of habitats, including broadleaved forests, mixed and coniferous forests, urban and suburban areas, and parks and gardens. It feeds primarily on the nuts and seeds of trees and shrubs, but maintains a varied diet and switches to different sources of food depending on availability at different times of year (Moller, 1983). Over-winter survival, and subsequent population density, are related to food availability and the severity of winter weather (Gurnell, 1996). In urban areas, population densities increase with the level of urbanisation (Baker and Harris, 2007; see also Bonnington et al., 2014), with grey squirrels making use of anthropogenic sources of food, such as bird seed in gardens. The grey squirrel can survive in highly fragmented, functionally isolated landscapes (Stevenson-Holt et al., 2014). Its generalist foraging behaviour and ability to adapt to different habitats and food sources have aided its spread throughout Britain.

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 7.2a. Extensive recording efforts in Scotland by the Saving Scotland's Red Squirrels partnership project, led by the Scottish Wildlife Trust, allows the production of a detailed distribution map for Scotland. The project has confirmed a gap in the distribution of grey squirrels between Dundee and an isolated population in Aberdeen. This gap has been filled by the smoothing process used to create the current distribution maps (see Methods section 2.5), rather than by records in this area. The established population of grey squirrels in Aberdeen does not extend to the north coast in Aberdeenshire/Banffshire; this area contains very sparse records that are likely to be derived from occasional individuals rather than established populations.

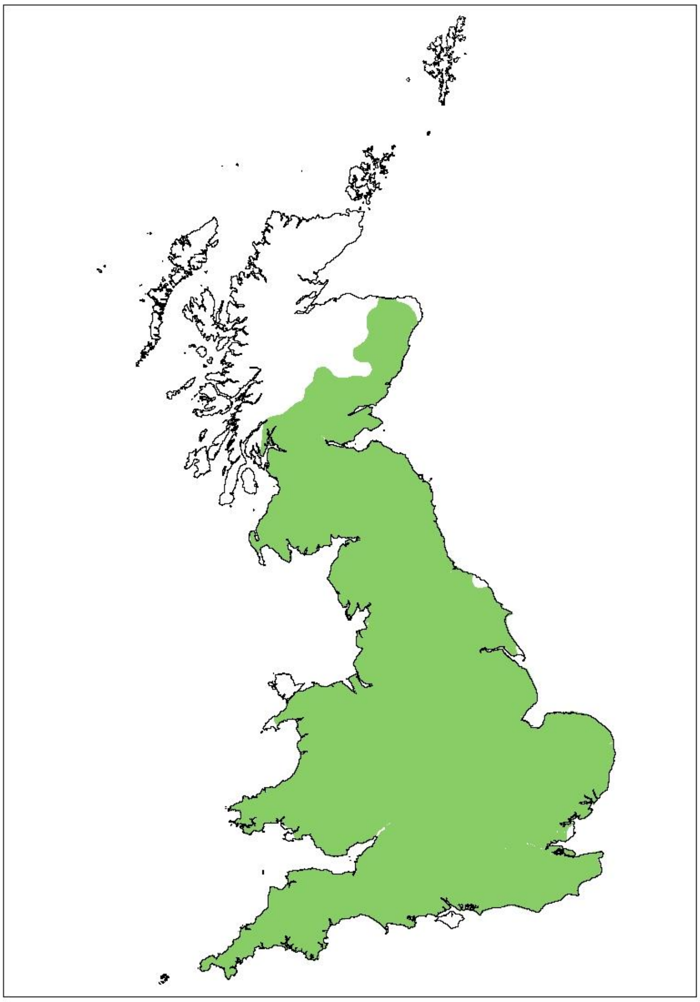


Figure 7.2a Current range of the grey squirrel in Britain. To reflect the current distribution and permit assessment of changes in the species' range through time, the maps are based on presence data collected between 2010 and 2016 (rather than 1995 and 2016 as for most other species in this review). This is because the distribution has been undergoing rapid expansion over the last 20 years, and the time-frame has been matched to that used for red squirrels. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

As mature woodlands provide a more suitable habitat for grey squirrels, all recently planted (<10 years) and felled woodlands, as defined in the LCM2007, were removed from the analysis (John Gurnell, *pers. obs.*).

Results

The literature search identified 18 relevant papers. Of these, eight contained pre-breeding population density estimates, four contained post-breeding estimates, and two were literature reviews that were used to obtain additional references included in the review. The remaining four papers contained details of the effect of environmental variables on relative density or demography. Population density estimates are provided in Table 7.2a, and population sizes in Table 7.2b.

Table 7.2a Median density estimates for grey squirrels with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	N**	%Occ†
Broadleaved woodland	1,160,000	1.90	0.80	2.45	Gurnell (1983) Gurnell (1996) Kenward et al. (1998) Cartmel (2000)	2 12 1 6	n/a
Urban and gardens	1,350,000	0.19	0.18	0.20	Bonnington et al. (2014)	3	n/a
Coniferous woodland	791,000	0.31	0.21	0.87	Bryce et al. (2005) Cartmel (2000) Gurnell et al. (2004a) Smith (1999)	1 2 18 17	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.2a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	2,260,000	1,940,000	957,000	2,560,000
Scotland	709,000	478,000	249,000	808,000
Wales	333,000	283,000	139,000	423,000
Britain	**3,300,000**	**2,700,000**	**1,340,000**	**3,790,000**

Critique

No percentage occupancy data were available; the population size is therefore overestimated for this species. Broadleaved woodland contributes 81% of the estimated grey squirrel population, and forms 35% of the suitable habitats within their distribution (Figure 7.2b). The population density estimate for broadleaved woodland is based on 21 individual density estimates from four papers, although the most recent of these is from 2000 (Cartmel, 2000). Coniferous woodland forms 27% of suitable habitats within the geographical range (Figure 7.2b), and population densities — based on 38 estimates from four papers — are much lower in this habitat.

There is considerable inter-annual variation in grey squirrel density, depending largely on tree seed availability. Further surveys of population density in years with different tree-seed abundance are therefore advised for all suitable habitats. Despite the removal of young (<10-year-old) coniferous woodland, much of the remaining commercial conifer forest included in the estimate is also too young to support grey squirrel populations. Also, extensive Sitka spruce plantations, which form 58% of productive coniferous woodland in Scotland (Forestry Commission, 2014), are included in the calculations despite having very low grey squirrel densities. Whilst these factors may have resulted in some overestimation of the population size, their impact will be smaller than for red squirrels because conifer woodlands in general support only low densities of grey squirrels. Nevertheless, the application of density estimates to finer scale habitat classifications may reduce this error in future assessments. A reliability assessment is provided in Table 7.2c.

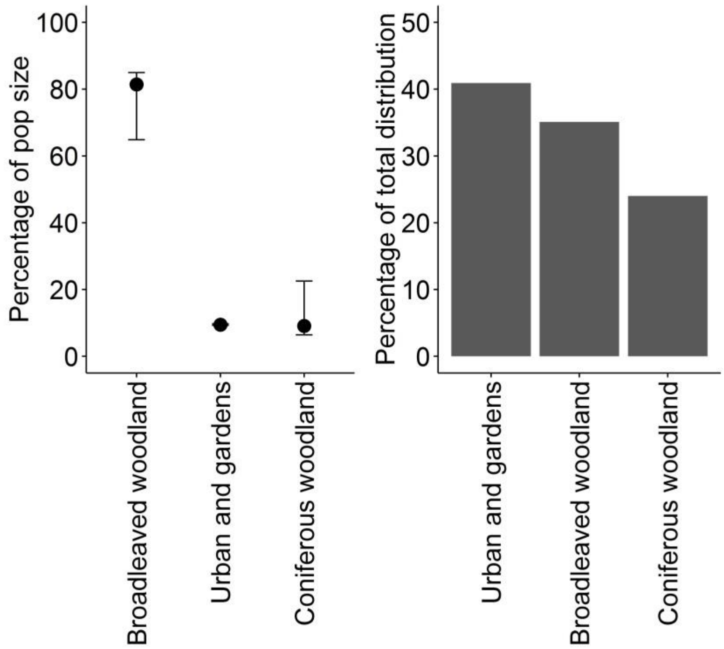


Figure 7.2b Left: The percentage of the total population of grey squirrels accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 7.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Coniferous woodland	**Broadleaved woodland**	**Urban and gardens**
Location of study sites	0	Estimates from one location			0
	1	Estimates restricted	1	1	
	2	Estimates widespread			
Sample size	0	<10 density estimates			0
	1	10-30 density estimates		1	
	2	>30 density estimates	2		
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**3**	**2**	**0**
		Overall reliability score	**1.7**		

Changes through time

Comparison to Harris et al. (1995)

Total population size was reported as 2,520,000 in Harris et al. (1995), with 2,000,000 in England, 200,000 in Scotland and 320,000 in Wales. These calculations were based on density estimates for woodlands as well as urban areas, although the population density used for urban areas was particularly low ($0.1ha^{-1}$). The methods used to estimate population size were similar to those in this review, but the relatively low reliability of our estimate, and lack of data on the percentage of occupied habitat, mean that a comparison is not advised. Population density as currently reported for broadleaved woodlands is much higher than the estimate used by Harris et al. (1995). This is most likely to be the result of within-habitat variation in population density (Peter Lurz, *pers. comm.*), rather than because of an actual increase in density.

Other evidence of changes through time

No other evidence on temporal trends was found during the literature search. A summary of trends in population size and range is provided in Table 7.2d.

Table 7.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	Scotland	England Wales		
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 7.2e Drivers of population change for grey squirrels between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Continuing range expansion following species introduction.	Colonisation of suitable habitat.	Mayle and Broome (2013)	Positive
Management (control).	Localised suppression.	Mayle and Broome (2013)	Negative

Data deficiencies

Table 7.2f Areas where further research is required to improve the reliability of population size estimates of grey squirrels.

Data deficiencies	Habitat	Details
Limited density estimates for specified habitat.	Urban and gardens	There are three recent estimates (post-1995).
Density estimates do not represent within-habitat variability.	Broadleaved and coniferous woodland	Population density is likely to be highly variable between years (depending on masting) and woodlands.
Density estimates are more than 10 years old.	Broadleaved woodland	The most recent estimate is from 2000.
No occupancy data.	All habitats	
Managed populations.	All habitats	Management is not taken into account in the current assessment.

Future prospects

Table 7.2g An assessment of the future prospects of the grey squirrel, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase*
Range	Increase
Habitat	Stable

*Rate of increase may slow in response to control.

7.3 Eurasian beaver *Castor fiber*

Habitat preferences

The beaver primarily occupies riverine and wetland habitats. The species requires year-round access to fresh water with suitable herbaceous vegetation which provides forage and materials for dam-building (Macdonald et al., 1995). A keystone species, the beaver often modifies sub-optimal habitats extensively, by building dams, burrows and lodges. Foraging predominantly within 20m of the water's edge, it eats a wide variety of aquatic and terrestrial vegetation, including willow, poplar and alder trees, grasses and forbs (Gurnell et al., 2008; Gaywood et al., 2015).

The beaver was nearly extinct in Europe, with only approximately 1,200 animals remaining in eight populations, at the start of the 20th century (Halley and Rosell, 2003). Following extensive conservation efforts, reintroductions in many areas (e.g., Scotland (Gaywood, 2018)), and new legal protection, it has made a considerable recovery (Halley et al., 2012).

Status

Native (reintroduced).

Conservation Status

- IUCN Red List (GB: EN; England: n/a; Scotland: n/a; Wales: n/a; Global: EN.)
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Under Article 22 of the Habitats Directive, Britain has a duty to consider the reintroduction of extinct native species (Macdonald et al., 1995). Wild beavers were reintroduced into Scotland in 2009 and into England in 2015 by means of trial releases. In November 2016, the Cabinet Secretary for Environment and Climate Change for the Scottish Government announced that beavers could remain in Scotland (Gaywood, 2018), and work began to put in place the full legal protection which would be afforded to the species under UK and EU legislation. The legal status of protection in England is currently under ministerial consideration, as the species has only relatively recently been reintroduced as part of a trial release.

Species' distribution

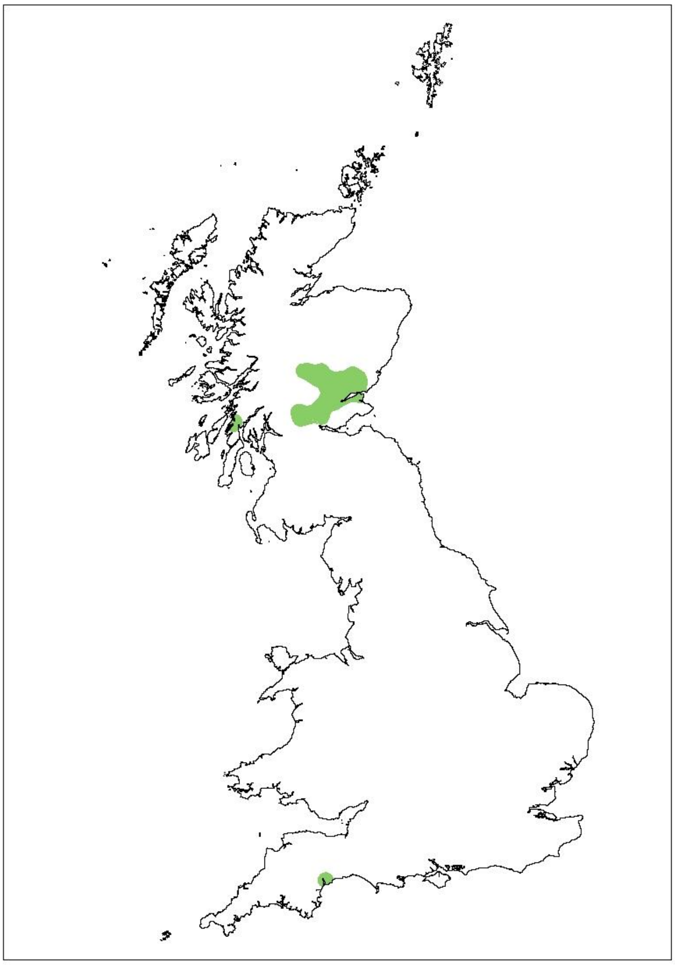


Figure 7.3a Current range of the Eurasian beaver in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Known populations of beavers are currently limited to three areas of Britain, for which total population size estimates are available. These estimates were combined to give an estimated population size of free-living beavers in Britain.

Results

Table 7.3a The size of free-living Eurasian beaver populations in Britain.

Location	Population Size	-95%CI	+95%CI	References	n
Tayside*	146	106	187	Campbell et al. (2012)	1
Knapdale & Argyll*	10	-	-	Gaywood et al. (2015)	1
River Otter, Devon	12	-	-	Devon Wildlife Trust (Mark Elliott, *pers. comm.*)	1
Total	**168**	-	-		

* A revised estimate of population size in Scotland is currently in preparation.

Changes through time

Comparison to Harris et al. (1995)

Beavers were not assessed in Harris et al. (1995). A summary of trends in population size and range is provided in Table 7.3b.

Other evidence of changes through time

Owing to their recent reintroduction, a detailed assessment of temporal trends has not yet been made.

Table 7.3b Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	Scotland England*			
	Stable				
	Decrease				
	Data deficient				

* Increase in range and population size owing to the species' reintroduction.

Drivers of change

Table 7.3c Drivers of population change for the Eurasian beaver between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Species' introduction.	There was a trial release in Knapdale in 2009, followed by reports of beavers in Tayside in 2012, thought to be the result of unlicensed releases. There was a trial release on the River Otter, England, in 2015.	Campbell et al. (2012) Gaywood et al. (2015)	Positive
Management (control).	Beavers provide ecosystem services such as increased ground water storage, flow stabilisation and flood prevention, but there are also concerns about negative socioeconomic impacts resulting from canal construction and felling of trees of commercial value. These conflicts of interest, along with uncertain legal protection for beavers in Scotland, have resulted in persecution.	Gaywood et al. (2015) Tayside Beaver Study Group (2015) Gaywood (2018)	Negative

Data deficiencies

Table 7.3d Areas where further research is required to improve the reliability of population size estimates for the Eurasian beaver.

Data deficiencies	Habitat	Details
Population sizes are based on total counts rather than density estimates.	Riparian	In small populations, numbers can be counted with reasonable accuracy.

Future prospects

Table 7.3e An assessment of the future prospects of the Eurasian beaver, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

7.4 Hazel dormouse *Muscardinus avellanarius*

Habitat preferences

The hazel dormouse is found primarily in broadleaved woodland. It is traditionally associated with early successional stages of woodland, as well as coppice, which is structurally similar (Bright et al., 2006; Juskaitis and Büchner, 2013), but recent studies have shown that it occurs in a range of wooded habitats including scrub, coniferous plantations and hedges (Chanin and Woods, 2003). Rather than being a strict habitat specialist, the hazel dormouse is therefore now seen as more adaptable (Juskaitis and Büchner, 2013). Similarly, although early studies in species-rich habitats showed that the species exploits a wide range of high quality plant foods (flowers, buds, seeds and fruits), it is now known also to occupy habitats with low food species diversity. An omnivorous diet, including significant quantities of insects, may permit this flexibility (Juskaitis and Büchner, 2013).

Status

Native.

Conservation Status

- IUCN Red List (GB: VU; England: [VU]; Scotland: n/a; Wales: [VU]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. UK: Bad; England: Bad; Scotland: n/a; Wales: Bad).

Species' distribution

Wales and southern England are the strongholds for this species. There is currently only one known population in Cumbria: the area presented in the distribution map (Figure 7.4a) may, therefore, be an overestimate. Owing to the levels of interest in dormice, and recording schemes such as the Great Nut Hunt (1993, 2001 and 2009-11) and the National Dormouse Monitoring Programme (Wembridge et al., 2016a), the gaps shown in the species' distribution in Wales may represent true gaps.

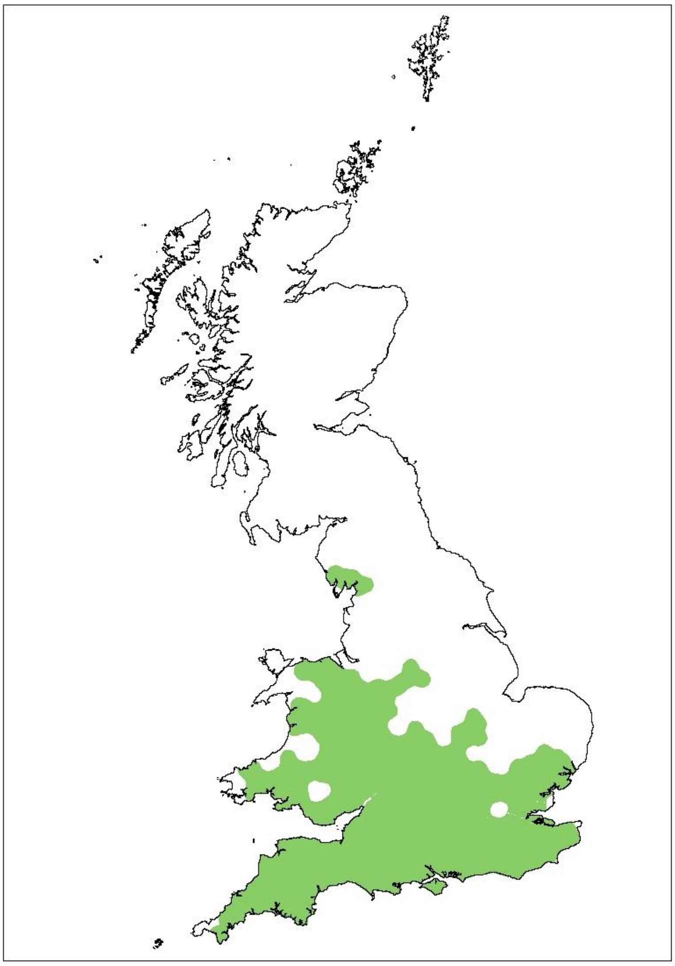


Figure 7.4a Current range of the hazel dormouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Percentage occupancy was available from two papers (Bright et al., 1994; Bright et al., 1996), and was based on the percentage of sites surveyed that contained signs of dormice (i.e., gnawed nuts). In Bright and Morris (1996), participants were asked to search hazel scrub for signs of dormice, but were not given prior knowledge of sites which were likely to contain dormice. Survey effort, and the availability of hazel nuts, were not standardised, potentially leading to some false-negative results. In Bright et al. (1994), woodlands stratified by age, area and isolation were selected at random, but surveys were conducted only where hazel scrub was heavily fruiting to maximise the probability of detecting dormice, and reduce the risk of false negatives. Survey effort was standardised between sites. The percentage occupancy used in this review was therefore derived from Bright et al. (1994), because more of the potential biases were addressed. Percentage occupancy for hedgerows was taken from Bright and MacPherson (2002), where occupancy was measured from hedgerows in 50 sites. The population estimates for hedgerows, also derived from Bright and MacPherson (2002), were converted into densities (per hectare) using the length and width of hedgerows. As the current analysis considers hedgerows as a linear habitat, these areas were converted to the number of hazel dormice per km, assuming each kilometre of hedgerow had an average width of 3m.

Results

Seventeen relevant papers were returned from the literature search, with four containing pre-breeding population density estimates. The remaining papers contained details of the species' presence, assessments of survey methods, or relative measures of population density. One paper (Bright and Morris, 2005) contained pre-breeding estimates from expert opinion that were included in another source (Bright et al., 2006). Population density estimates are provided in Table 7.4a, and population size estimates in Table 7.4b.

Table 7.4a Median density estimates for hazel dormice with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerow length and density within hedgerows are presented as km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	734,000	3.0	1.0	8.2	Bright et al. (2006) †† Chanin and Gubert (2011) Trout et al. (2012)	5 2 2	34%
Coniferous woodland	229,000	2.0	1.6	7.3	Expert opinion	3	34%
Hedgerows	275,000 (km)	0.26	0.15	0.36	Bright and MacPherson (2002) Bright et al. (2006)	2 1	35.5%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
†† Estimates from this reference are based on expert opinion.

Table 7.4b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.4a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	764,000	757,000	298,000	2,110,000
Scotland	0	0	0	0
Wales	200,000	172,000	90,700	529,000
Britain	**964,000**	**930,000**	**389,000**	**2,640,000**

* The lengths of hedgerows are 238,000km in England and 372,000km in Wales.

The Article 17 Report on hazel dormouse population size 2007-2012 is shown in Table 7.4c (Joint Nature Conservation Committee, 2013b). The estimated population size in the current review is more than an order of magnitude larger, though there are methodological differences between the two reports. The geographical ranges reported in the Article 17 report and the current review are similar (Table 7.4d).

Table 7.4c Article 17 Report on hazel dormouse population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	37,500	37,500
Scotland	0	0
Wales	7,500	7,500
Britain	**45,000**	**45,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

Table 7.4d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013b).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	67,600	n/a
Scotland	0	n/a
Wales	14,700	n/a
Britain	**82,300**	**86,890**

Critique

Most of the population is found in broadleaved woodlands (83%), and this accounts for 76% of the species' distribution (Figure 7.4b). The population density estimate for broadleaved woodland ranged from $1ha^{-1}$ to $8.2ha^{-1}$, based on nine density estimates reported in three papers. Experts in the field provided population density estimates which broadly agree with those found in the literature, with best-guess density estimates ranging from $1ha^{-1}$ to $10ha^{-1}$ for broadleaved woodland, and $0km^{-1}$ to $28km^{-1}$ for hedgerows. The reliability score for population estimates in broadleaved woodland is shown in Table 7.4e.

Percentage occupancy values were estimated from surveys of woodlands containing hazel only. The possibility of dormice living in a wider range of habitats (including those where hazel was absent) was not considered. As recent research suggests that hazel dormice are not specialised to hazel coppice, and are much more adaptable to other habitat types (Juskaitis and Büchner, 2013), the percentage occupancy value of 34% may not be representative of all habitats, and could be a significant underestimate (Paul Chanin, *pers. comm.*). Conversely, permanent populations are unlikely to be found in woodlands < 20ha, even though these form a significant proportion of woodlands in the species' range (Tony Mitchell-Jones, *pers. comm.*); therefore, occupancy may be lower than 34% in some regions.

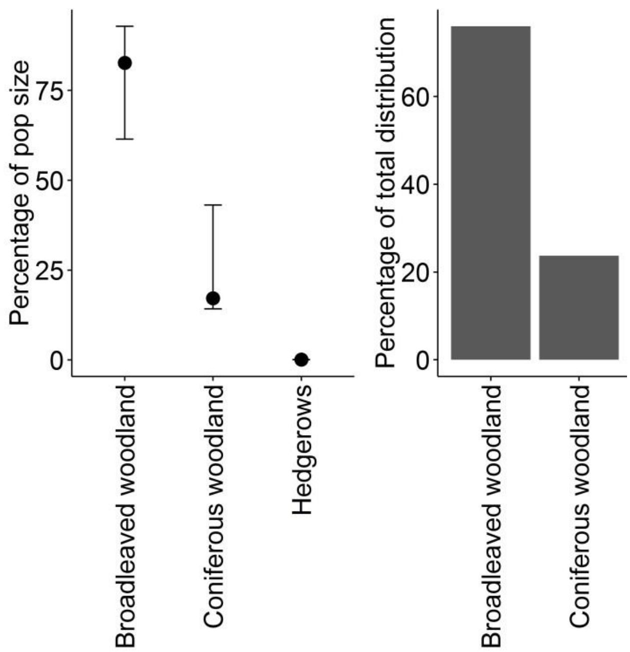


Figure 7.4b Left: The percentage of the total population of hazel dormice accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.4e Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Broadleaved woodland
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**2**
		Overall reliability score	**2**

Changes through time

Comparison to Harris et al. (1995)

Population estimates from Harris et al. (1995) were limited to ancient woodlands, and were reported as 500,000 in Britain, 465,000 in England and 35,000 in Wales. The principal difference between the current review and Harris et al. (1995) is that the latter used a higher estimate of density but a narrower range of habitats. A population density of $5ha^{-1}$ was applied to ancient woodlands based on expert adjustment of the density of 8-$10ha^{-1}$ found in prime habitat. The current analysis includes all broadleaved woodland, rather than ancient woodland only, as well as hedgerows and coniferous woodland. 50% of available habitat was assumed by Harris et al. (1995) to be occupied, as opposed to 34% in the current analysis.

Population sizes are, therefore, unlikely to be directly comparable between the two time periods because of the methodological differences. A summary of trends in population size and range is shown in Table 7.4f.

Other evidence of changes through time

The National Dormouse Monitoring Programme (NDMP) has assessed trends in relative population size through counts of nest box occupancy in selected sites since 1993. During this period, there has been a steady decline in relative occurrence (numbers of adult dormice found in boxes), particularly in eastern areas, with a 48% (95%CI = 39%-55%) overall decline reported for the 10 years from 2005 to 2015 (Goodwin et al., 2017). Inferences about changes to population size depend on the relationship between nest box occupancy and true dormouse density; this relationship is currently unknown, and may vary over time if alternative nesting opportunities change. A genetic assessment of two British woodlands also revealed that a high proportion of the population was not encountered during nest box monitoring (Naim et al., 2011). Nevertheless, the trends in the NDMP appear consistent between shorter- and longer-term survey periods, and are robust to different levels of survey effort, suggesting that the NDMP currently provides the best available evidence on dormouse population trends (Goodwin et al., 2017).

Table 7.4f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		England Wales*		
	Data deficient				

* Range and population size trends are taken from Goodwin et al. (2017) rather than from comparison with Harris et al. (1995). Trends reflect monitored sites only; wider trends are unknown. A remnant population in Northumberland is believed to have become extinct since 2010 (Ian White, *pers. comm.*).

Drivers of change

Table 7.4g Drivers of population change for hazel dormice between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat loss.	Fragmentation; poor management can lead to a reduction in woodland species diversity and over-shading of understory. In the past, a decline in coppice management may have led to a reduction in range and density; revival of the practice in recent years has provided more optimal habitat.	Bright et al. (2006)	Negative
Habitat quality.	Climate change may cause changes in food availability. Increasing deer numbers may affect woodland understory.	Bright et al. (2006) Gill and Fuller (2007)	Negative

Data deficiencies

Table 7.4h Areas where further research is required to improve the reliability of population size estimates for hazel dormice.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Coniferous woodland	Density estimates are based on expert opinion.
No occupancy data for the specified habitat.	Coniferous woodland	Percentage occupancy is based on surveys of hazel coppice.
Limited density estimates for key habitat.	Broadleaved woodland, hedgerows	Median density is based on 9 and 3 density estimates, respectively.

Future prospects

Table 7.4i An assessment of the future prospects of the hazel dormouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

7.5 Edible dormouse *Glis glis*

Habitat preferences

The edible dormouse is found in coniferous, broadleaved and mixed woodlands, as well as orchards and gardens (Harris et al., 1995). It shows a dietary preference for oak acorns and beech nuts (Pilastro et al., 2003; Ruf et al., 2006). Habitat barriers are thought to have limited its spread, although specific habitat requirements in Britain have not been studied in detail. Most research has focused on the negative effects of edible dormice on forestry (Platt and Rowe, 1964; Jackson, 1994), with some research on demography and population dynamics in known habitats (Burgess et al., 2003; Morris and Morris, 2010). The edible dormouse exhibits unusual life history traits, being relatively long-lived with a short, often unsuccessful, breeding season (Morris and Morris, 2010). The reason for the restricted distribution in the south of England is unclear, but it is likely to reflect its slow reproduction and lack of dispersal behaviour, as well as habitat barriers (Morris and Hoodless, 1992), rather than a lack of suitable habitat in other areas (Morris and Morris, 2010).

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

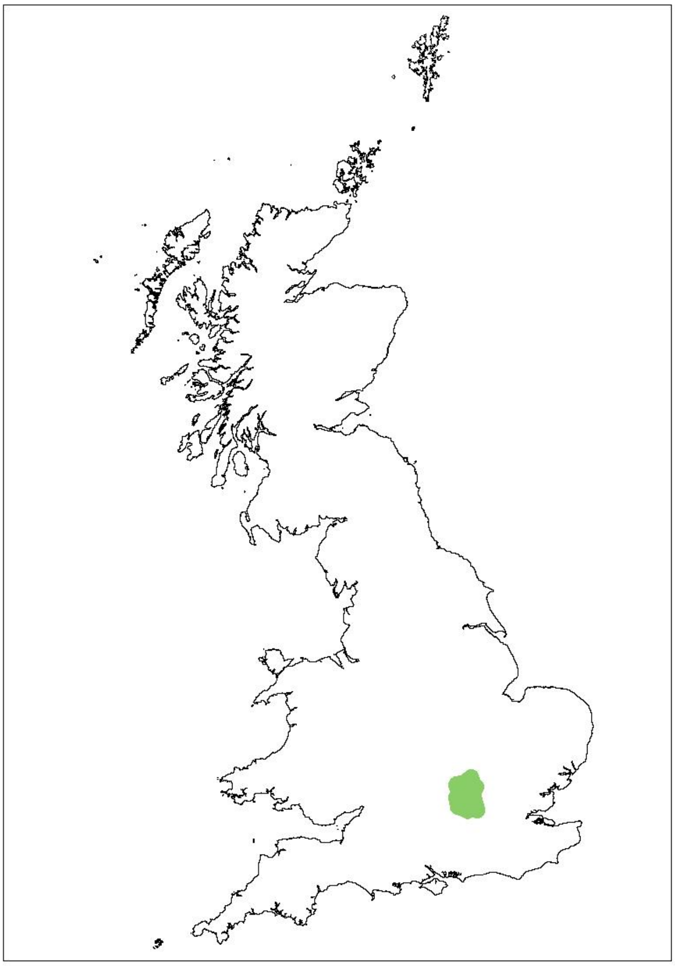


Figure 7.5a Current range of the edible dormouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

Four papers were identified by the literature search, one of which contained pre-breeding population density estimates. Two of the remaining papers contained density estimates (Morris and Temple, 1998; Burgess et al., 2003), but the data were subsequently included in the publication by Morris and Morris (2010). The final paper contained information on population dynamics. Population density estimates are provided in Table 7.5a, and population size estimates in Table 7.5b.

Table 7.5a Median density estimates for edible dormice with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (ha)	Density (ha^{-1})	-95CI	+95CI	Author*	n**	% Occ†
Broadleaved woodland	27,000	0.84	0.36	3.0	Morris and Morris (2010)	13	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.5b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 4.5a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	27,000	[23,000]	[9,800]	[82,000]
Britain	**27,000**	**[23,000]**	**[9,800]**	**[82,000]**

Critique

The 1995 population size estimate was based on a single pilot study, with an assumed occupancy of 66% of the woodland available within the species' range. The current review uses yearly estimates taken over a period of 13 years at the same location (Morris and Morris, 2010); variation in abundance between these estimates therefore reflects temporal, but not spatial, differences. No occupancy data were available, and so the calculated population size is an overestimate. The methodological differences make direct comparisons between the two reviews difficult.

Expert opinion resulted in a suggested population size of 200,000-300,000, based on the availability of domestic properties and woodland in the Chiltern area. Dormouse densities of 7 per domestic property (with a 60% occupancy rate) and 15 per hectare of woodland, were assumed, and the population size was halved to provide an estimate for a non-peak year (Roger Trout, *pers. comm.*). The surveys on which these calculations are based are likely to have been conducted in high density areas. It is unclear whether they are representative of the whole species' range. It is also possible that there could be some double counting, as dormice using buildings are likely to forage in nearby woodland.

Reported population densities are lower in the northern parts of the species' range in mainland Europe (e.g., 0.8-2ha^{-1} and 2ha^{-1} in Lithuania and Latvia, respectively (Pilāts et al., 2009; Juskaitis et al., 2015)), than in central and southern Europe (10-50ha^{-1} (Rossolimo et al., 2001; Kryštufek and Flajšman, 2007; see Juskaitis et al., 2015)). The density estimate used for woodland in this review is therefore comparable to densities found in the northern periphery of the species' range, whereas the expert opinion estimate is closer to that reported for central and southern Europe. A reliability score is provided in Table 7.5c.

Table 7.5c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Broadleaved woodland
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	
	1	10-30 density estimates	1
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**1**
		Overall reliability score	**1**

Changes through time

Comparison to Harris et al (1995)

Harris et al. (1995) reported a population size of 10,000 for England, although this estimate was based on sparse data. The distribution of edible dormice does not appear to have expanded significantly in the last 20 years, and their population density is unlikely to have changed radically. However, the current population estimate is twice that proposed by Harris et al. (1995) (though an order of magnitude smaller than the estimate from expert opinion). Nationally, there are changes between the two reviews in the estimated availability of several key habitats, generated by a combination of true change and methodological differences, regardless of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — still yields a population estimate approximately twice that of Harris et al. (1995). The difference between the two reviews is therefore considered to result from the different methodologies, and the lack of robust occupancy data for either study, as well as from true population increase.

Other evidence of changes through time

The species' distribution has increased very slowly, most likely because of slow reproduction, lack of dispersal behaviour, and habitat barriers (Morris and Hoodless, 1992; Morris and Morris, 2010). There has been no systematic attempt to eradicate the species from Britain. A summary of trends in population size and range is shown inTable 7.5d.

Table 7.5d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	England*			

* Very slow increase owing to slow reproduction, lack of dispersal behaviour, and habitat barriers.

Drivers of change

Table 7.5e Drivers of population change of edible dormice between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Continued expansion into suitable habitat.		Positive
Habitat quality.	Fruiting cycles (climate change). Hibernation (climate change).		Positive Positive/Negative
Management (control).	Localised suppression.		Negative

Data deficiencies

Table 7.5f Areas where further research is required to improve the reliability of population size estimates for edible dormice.

Data deficiencies	Habitat	Details
No occupancy data.	Broadleaved woodland	Across the species' distribution.
No density estimates for specified habitat.	Urban and gardens	

Future prospects

Table 7.5g An assessment of the future prospects of the edible dormouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase**
Range	Increase*
Habitat	Increase

* Very slow increase owing to slow reproduction, lack of dispersal behaviour, and habitat barriers.
** Increase in population size is inferred from an increase in range.

7.6 Bank vole *Myodes glareolus*

Habitat preferences

The bank vole is found in a variety of habitats, including hedgerows, conifer plantations and road verges, but shows a strong preference for mature broadleaved and mixed woodland (Flowerdew et al., 2004). The diet comprises fruits, seeds and leaves from broadleaved trees, although other food sources such as flowers, grasses and moss are taken opportunistically. Unlike populations in mainland Europe, the bank vole forms a high proportion of its winter diet in Britain from dead leaves (Hansson, 1985).

Limited recent research is available on the factors affecting the population density of bank voles, although abundance is positively associated with the quality and size of woodlands. The species requires dense ground vegetation (Fernando et al., 1994). It is found frequently in field margins and hedgerows, which can support large resident populations (Gelling et al., 2007), but only rarely in arable fields (Harris and Yalden, 2008). Arable habitats were therefore excluded from the population estimates. There is no evidence of multi-annual cycles for bank voles in Britain (Flowerdew et al., 2004).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.6a. Gaps in the distribution in England and Wales are likely to represent areas lacking survey effort, rather than true absences. It is unclear whether the larger gaps in Scotland reflect a lack of recorder effort or true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

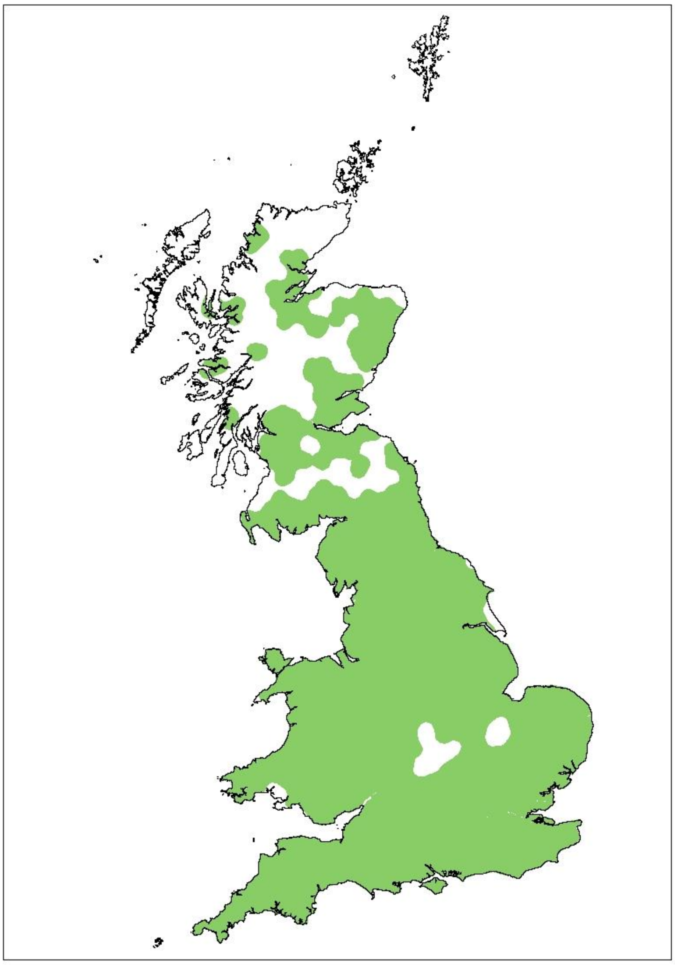


Figure 7.6a Current range of the bank vole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

The literature search (using both *Myodes glareolus* and *Clethrionomys glareolus* as species names) identified 63 papers. Fourteen provided information on population size or distribution; of these, six gave pre-breeding estimates of population density, and one contained percentage occupancy for hedgerows (Gelling et al., 2007). The remaining papers reported post-breeding estimates, assessed the relative effects of environmental variables on population size, or gave distribution data only. Population density estimates by habitat are shown in Table 7.6a, and total population size estimates in Table 7.6b.

Table 7.6a Median density estimates for bank voles with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided according to whether they were managed through an agri-environment scheme (AES and non-AES). Hedgerow length and density within hedgerows are shown in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,200,000	9	6	10	Flowerdew et al. (2004), Hare (2009)	147 2	n/a
Urban areas and gardens	1,320,000	4.5	3.0	7	expert opinion	2	n/a
Coniferous woodland	944,000	5	3.5	12	expert opinion	2	n/a
Dwarf shrub heath	966,000	1	0.05	3	expert opinion	1	n/a
Fen, marsh and swamp	6,900	30	0	50	expert opinion	1	n/a
Improved grassland	6,070,000	0.1	0	1	expert opinion	2	n/a
Unimproved grassland	900,000	1	0	3	expert opinion	3	n/a
Hedgerows (AES)	139,000 (km)	10.5	0	26	Shore et al. (2005) Broughton et al. (2014) Kotzageorgis and Mason (1997)	24 12 9	96%
Hedgerows (Non-AES)	311,000 (km)	6.3	1.2	8			

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.6b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.6a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	7,120,000	19,100,000	10,400,000	35,600,000
Scotland	2,520,000	5,390,000	3,130,000	11,900,000
Wales	1,770,000	2,930,000	1,560,000	6,560,000
Britain	**11,400,000**	**27,400,000**	**15,100,000**	**54,100,000**

* The lengths of hedgerows are 386,000km in England, 14,000km in Scotland, and 50,400km in Wales.

Critique

Percentage occupancy data were not available for most habitats; the population size is therefore overestimated. Most of the population estimate is derived from broadleaved woodland (39%). Yet broadleaved woodland forms a low proportion of the land cover within the species' range (Table 7.6a), and its importance is therefore largely a consequence of high density estimates relative to other habitats. Many of the density estimates for these other habitats were derived from expert opinion, highlighting the need for detailed surveys in habitats lacking empirical data.

Just over half of the habitat within range is improved grassland (Table 7.6a). It is estimated that this large habitat area supports only 1% of total bank vole population because of low population density ($0.1ha^{-1}$). This estimate is again the average of values provided by two expert opinions, and validation of the estimated values would improve future calculations of population size. Despite the low suitability of improved grassland and arable land for bank voles, a significant proportion of the population is found in hedgerows in these areas. It is possible that many individuals captured within improved grassland reside in hedgerows, and therefore the inclusion of both these habitats could have slightly overestimated the population. (Arable land was excluded.) A reliability assessment is provided in Table 7.6c. For density estimates based on expert opinion, a conservative score of 1 has been applied to the 'location of study sites' section.

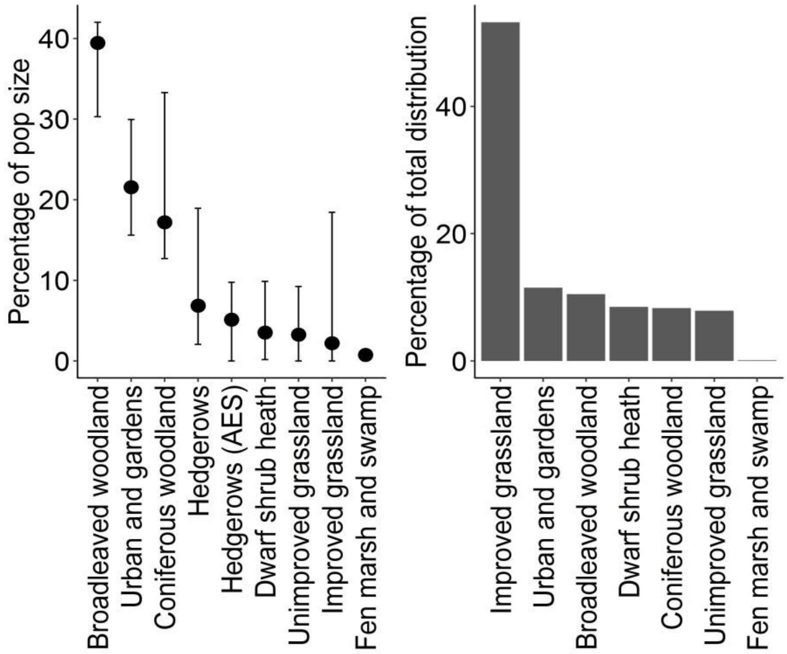


Figure 7.6b Left: The percentage of the total bank vole population accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.6c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Broadleaved woodland	**Unimproved grassland**	**Improved grassland**
Location of study sites	0	Estimates from one location			
	1	Estimates restricted		1	1
	2	Estimates widespread	1		
Sample size	0	<10 population density estimates		0	0
	1	10-30 population density estimates			
	2	>30 population density estimates	2		
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**3**	**1**	**1**
		Overall reliability score	**1.7**		

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total population size of 23,000,000, comprised of 17,750,000 in England, 3,500,000 in Scotland and 1,750,000 in Wales. Those values fall within the confidence limits of our estimates here, except in Scotland for which current estimates are somewhat higher. Population sizes were calculated by Harris et al. (1995) from density estimates for hedgerows, woodland, scrub and bracken only, where the density per habitat type was assigned using a combination of empirical data and expert opinion. The methods to estimate current population size are, therefore, similar, although the difference in selected habitats and use of expert opinion mean that comparisons should be made with caution.

Nationally, there are changes between the two reviews in the estimated availability of key habitats, generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size only 5% different from the original (and within the original confidence limits). These differences in assumed habitat areas therefore do not materially affect the comparison of population sizes between the reviews.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is shown inTable 7.6d.

Table 7.6d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland*	

Drivers of change

Table 7.6e Drivers of population change for bank voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Unknown.			

Data deficiencies

Table 7.6f Areas where further research is required to improve the reliability of population size estimates of bank voles.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Fen, marsh, swamp, heathland, grassland, different types of conifer forest, urban, suburban.	Density estimates are currently based on expert opinion.
No occupancy data.	All habitats except hedgerows.	

Future prospects

Table 7.6g An assessment of the future prospects of the bank vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

7.7 Field vole *Microtus agrestis*

Habitat preferences

The field vole is most abundant in rough low-productivity grassland under low intensity management and untreated by artificial fertilisers. Long tussocky grass allows for the formation of runs and nests (Gelling et al., 2007), and provides protection from aerial predators. The species' density is negatively associated with grazing pressure, but low intensity grazing, particularly by sheep, may be beneficial as it leads to more diverse vegetation structure (Schmidt et al., 2005). The field vole may also live in marginal habitats: it can occupy open grassy patches within fragmented woodlands, as well as moorlands, at low densities (Bellamy et al., 2000; Tattersall et al., 2000), and marginal rough grasslands may support high densities.

Linear habitats, such as hedgerows, are becoming an increasingly important because of habitat fragmentation and the loss of tussocky grasslands (Tattersall et al., 2002). Not only can hedgerows provide useful corridors, but the vegetation in hedgerow bottoms can also provide the sole habitat for the species, especially within pastoral landscapes with typical grazing intensities (Gelling et al., 2007).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 7.7a. Gaps in the species' distribution throughout the mainland are likely to represent areas lacking survey effort, rather than true absences. Further survey effort is recommended in these areas to increase confidence in the current distribution.

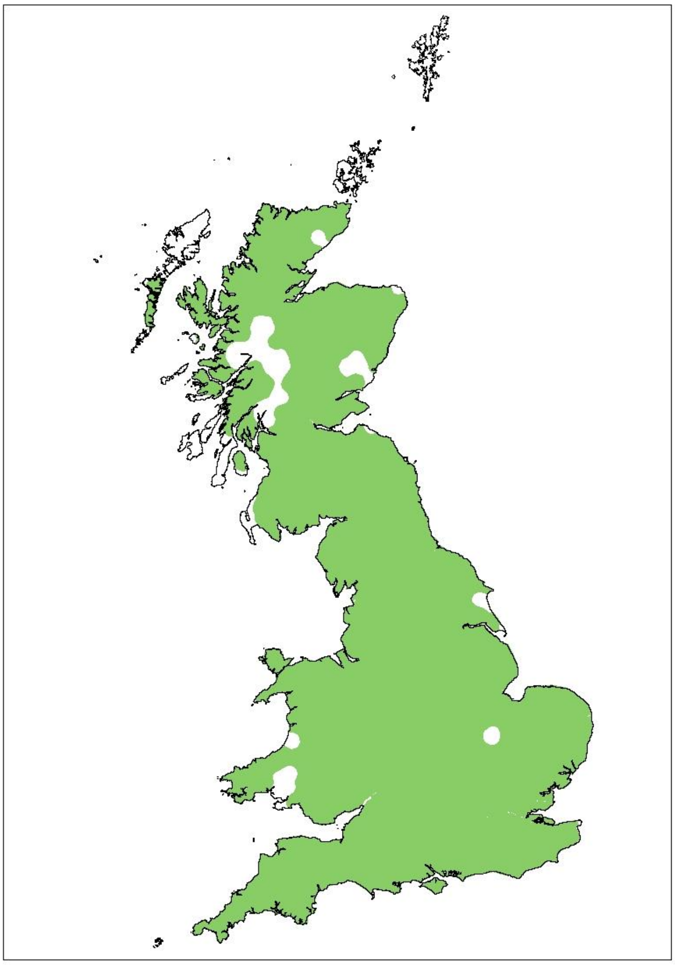


Figure 7.7a Current range of the field vole in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Field voles use long tussocky grass as their primary habitat, so it was assumed that animals within both arable and improved grassland would be transient individuals that would be largely accounted for by the estimates for hedgerows. Arable and improved grassland habitats were therefore not included in the calculation of population size. Annual and multi-annual cycles (three to four year periodicity) in population size are known in Britain, but non-cyclic populations are also present in different areas (Lambin, 2008). These differences make it difficult to assess the status of the population as a whole, or even the stage of the cycle reached at any given time. Therefore, whenever temporal data were available, troughs in population size were used as the best estimate of pre-breeding density, even if these were not obtained in the spring (Lambin et al., 2000; Loughran, 2006).

Results

Nine papers were identified by the literature search. Of these, four contained pre-breeding estimates of population size, four reported post-breeding estimates of population size, and one examined the effect of habitat variables on relative population size. Habitat-specific population densities and total population size estimates are presented in Table 7.7a and Table 7.7b, respectively.

Table 7.7a Median density estimates for field voles with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided according to whether they were managed through an agri-environment scheme (AES and non-AES). Hedgerow lengths are in km, and animal density within hedgerows is in km^{-1}.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Unimproved grassland	1,100,000	48	33.33	52.5	(Lambin et al., 2000) Flowerdew et al. (2004) Loughran (2006)	11 2 5	n/a
Bog	757,000	1	0	1	expert opinion	2	n/a
Broadleaved woodland	1,270,000	1	0	2	expert opinion	2	n/a
Urban and gardens	1,360,000	1.5	0.05	2.5	expert opinion	2	n/a
Coniferous woodland	1,310,000	1	0	1.5	expert opinion	2	n/a
Fen, marsh and swamp	8,570	5	0	12	expert opinion	1	n/a
Hedgerows (AES)	143,000 (km)	7.5	2	52	Broughton et al. (2014)	12	n/a
Hedgerows (Non-AES)	319,000 (km)	2	0	20			

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.7b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.7a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	3,090,000	28,600,000	16,900,000	44,000,000
Scotland	2,150,000	21,500,000	13,600,000	24,500,000
Wales	561,000	9,760,000	6,430,000	11,800,000
Britain	**5,810,000**	**59,900,000**	**37,000,000**	**80,300,000**

* The lengths of hedgerows are 396,000km in England, 18,700km in Scotland, and 47,300km in Wales.

Critique

No percentage occupancy data were available; the population size is therefore overestimated. Most of the estimated population (88%) is derived from unimproved grassland (Figure 7.7b). The population density used for this habitat is based on 18 sites reported in three papers. Unimproved grassland accounts for 19% of the habitat found within the species' distribution. Coniferous woodland, broadleaved woodland and urban areas form most of the remaining area (67%), although the densities within these habitats are thought to be low (Table 7.7a). Improved grassland was excluded from the analysis on the grounds that it offers poor habitat for field voles, and individuals using this environment would be accounted for by the hedgerow density estimates.

Grassland habitats have spectrally similar profiles in remotely sensed datasets, so areas of rough grassland may have been misclassified as improved grassland, and vice versa, in the LCM2007 dataset. The area given for rough grassland may therefore be inaccurate. We also grouped improved, neutral, acid and calcareous grasslands together for the analysis, combining those habitats most likely to be mistaken for each other but which are also functionally similar. Some of this 'improved grassland' grouping may support a low density of field voles, particularly if grazing intensity is low (Schmidt et al., 2005), but the habitat has been excluded from this analysis. A reliability assessment is provided in Table 7.7c.

.

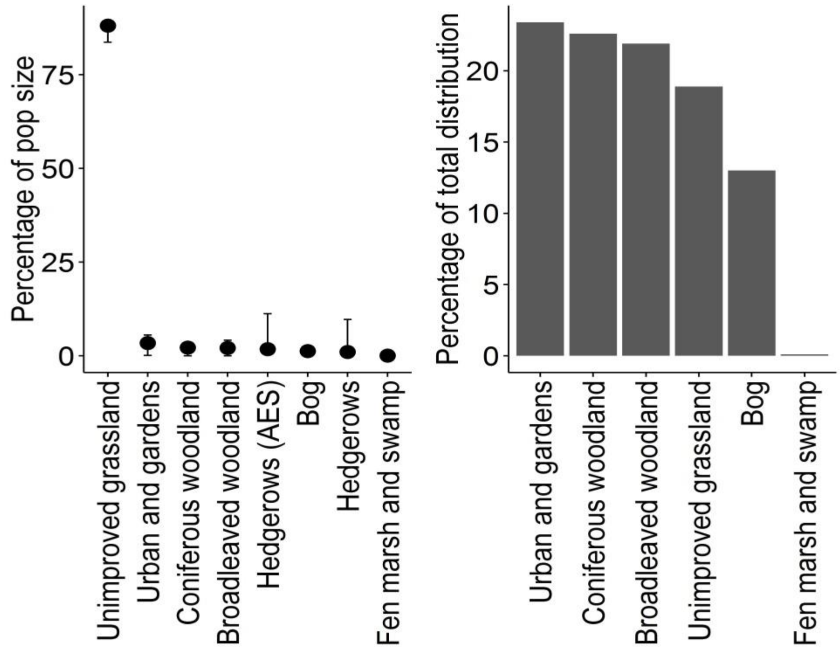


Figure 7.7b Left: The percentage of the total population of field voles accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.7c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**
			Unimproved grassland
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	1
	2	>30 population density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**2**
		Overall reliability score	**2**

Changes through time

Comparison to Harris et al. (1995)

The population size estimate in Harris et al. (1995) was 75,000,000. This estimate was based on the ratio of field voles to other small mammals in a range of samples, including live traps and owl pellets. Values of 1.9 field voles per wood mouse and 1.8 field voles per common shrew were derived across all samples. As this method is not based on the area of suitable habitat within the species' distribution, a comparison between population size estimates from Harris et al. (1995) and the current estimate is not advised.

Other evidence of changes through time

No other sources of data on temporal trends were found in the literature. However, a report from 1955 indicates a pre-breeding density of 118-530 field voles ha^{-1} (Lockie, 1955), and field vole 'plagues' have been regularly reported historically. Although population cycles continue, the pre-breeding densities for rough grassland in recent literature are at least half of those given in the 1955 report. This difference may reflect either a long-term change in population sizes, or simply that the available studies are not representative of the whole population. A summary of trends in population size and range is shown in Table 7.7d.

Table 7.7d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries		

Drivers of change

Table 7.7e Drivers of population change for field voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Habitat fragmentation, nitrogen deposition and lack of management.		Negative

Data deficiencies

Table 7.7f Areas where further research is required to improve the reliability of population size estimates for field voles.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Bog Broadleaved woodland Urban and gardens Coniferous woodland Fen, marsh and swamp	Density estimates are based on expert opinion.
Multiannual population cycles.	All habitats	Adds uncertainty to the population size estimate as the stage of the cycle is unknown.
No occupancy data.	All habitats	
Availability of rough pasture of appropriate structure.	Grasslands	Field voles are highly dependent on tussocky grassland. Availability is currently difficult to determine from LCM2007.

Future prospects

Table 7.7g An assessment of the future prospects of the field vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Decline

7.8 Orkney Vole *Microtus arvalis orcadensis*

Habitat preferences

The Orkney vole is a subspecies of the common vole *Microtus arvalis*. It has a larger body size than common voles, as well as other morphological differences (Berry, 1996). The species is present in most natural habitat types, conifer plantations and linear features throughout Orkney. However, it has largely disappeared from agricultural fields following the switch to high intensity production methods (Gorman and Reynolds, 1993; Gorman and Reynolds, 2003), and in these areas, ditches (including old peat-cuttings), fence-lines and road verges are important habitats.

Status

Non-native (naturalised — island endemic).

The species was introduced to the Orkney archipelago approximately 5,000 years ago (Martínková et al., 2013).

Conservation Status

- IUCN Red List (GB: VU; England: n/a; Scotland: [VU]; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

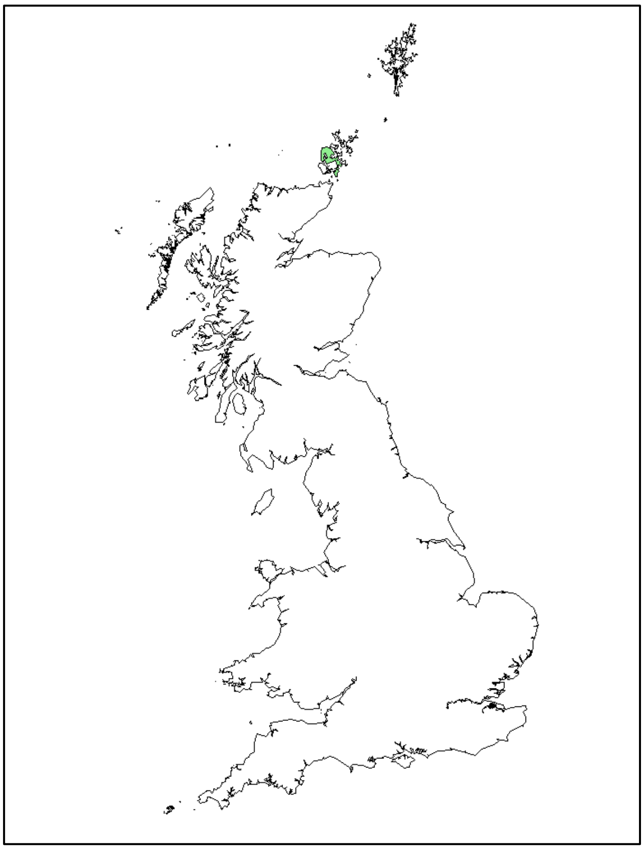


Figure 7.8a Current range of the Orkney vole in Great Britain. Range is based on presence data collected between 1995 and 2016.

Results

No estimates for pre-breeding population densities were available from the literature review, and no data were available from the previous population review (Harris et al., 1995). However, overall population estimates were made for the years 1998-1990: post-breeding estimates for Orkney were 3,000,000 on Mainland, 500,000 on Rousay, 300,000 on Westray, 200,000 on South Ronaldsay and 100,000 on Sanday (Reynolds, 1992). Pre-breeding populations can therefore be inferred to be 1,000,000-2,000,000. The population size is not known to experience cycles (see(Gorman and Reynolds, 2003; Harris and Yalden, 2008; Fraser et al., 2015a).

Critique

There was no evidence on which to base a population estimate for this review.

Table 7.8a Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All
Location of study sites	0	Estimates from <=1 location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**0**
		Overall reliability score	**0**

Changes through time

Comparison to Harris et al. (1995)

It is not possible to make a comparison because no estimate could be made, and the value in the previous report also had low reliability, having been based on expert opinion (Harris et al., 1995).

Other evidence of changes through time

There are significant concerns about declines in the population inferred from the loss of natural habitats to agriculture (falling from 81% of land cover in 1936 to 63% in the early 1990s) and the switch to high intensity methods of agricultural production, creating habitats with low suitability for voles (Reynolds, 1992; Gorman and Reynolds, 1993; Gorman and Reynolds, 2003). In addition, stoats were introduced to the archipelago in 2010, and are now present throughout Mainland, Burray and South Ronaldsay, posing a significant threat to vole populations (Fraser et al., 2015a) . Orkney voles are important prey for hen harriers, and reported declines in that species may also be indicative of a continuing decline in Orkney vole populations (Amar et al., 2003).

Table 7.8b Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease		Scotland*		
	Data deficient				

* Inferred from loss of habitat and the introduction of a non-native predator.

Drivers of change

Table 7.8c Drivers of population change for Orkney voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Habitat quality.	Agricultural intensification results in a reduction in suitable habitat and cover.	Reynolds (1992) Gorman and Reynolds (1993) Gorman and Reynolds (2003)	Negative
Habitat availability.	Conversion of natural habitats to agriculture.	Gorman and Reynolds (1993)	Negative
Predation.	Introduction of a non-native predator (stoats).	Fraser et al. (2015a)	Negative

Data deficiencies

Table 7.8d Areas where further research is required to improve the reliability of population size estimates for Orkney voles.

Data deficiencies	Habitat	Details
No density estimates.	All	
No occupancy data.	All	
No estimates of within-habitat variability.	All	

Future prospects

Table 7.8e An assessment of the future prospects of the Orkney vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

7.9 Water vole *Arvicola amphibius*

Habitat preferences

The water vole in Great Britain is primarily riparian, usually occurring within 2m of water. Although fossorial ecotypes not associated with water are common in continental Europe (Berthier et al., 2014), and populations have been identified in localised areas of Glasgow and on some Scottish islands (Telfer et al., 2003; Stewart et al., 2017), these are currently considered to be a small proportion of the total population. The riparian water vole prefers slow-flowing rivers, streams and marshes with tall dense vegetation (Strachan and Jefferies, 1993) that provides cover from avian predators (Lawton and Woodroffe, 1991). Reeds and grasses are used for food, cover and nesting material, while steep sandy banks allow it to construct extensive burrows above and below the waterline (Barreto et al., 1998b).

Unlike the larger colonies found in the lowlands, the upland water vole forms small scattered colonies, occupying dispersed patches of suitable habitat (Aars et al., 2001). These fragmented populations are vulnerable to stochastic variation and other threats (Capreolus Wildlife Consultancy, 2005). Nevertheless, upland areas and headwater streams are now the most important remaining sites for the water vole in some areas, despite low population densities (Walsh and Hall, 2005).

Over the last century, intensification of agriculture has had a number of adverse consequences for water vole habitat. Factors detrimental to water voles have included wetland drainage, the encroachment of cultivated land into riparian and wetland habitats, overgrazing, and the degradation of the structural and vegetative suitability of banks for water vole burrows because of cattle poaching. River bank reinforcement programmes, and increased frequency of spate events because of altered drainage patterns and weather changes, have also negatively affected the suitability of riparian habitat. Together with predation by the non-native American mink (*Neovison vison*), these changes have resulted in a drastic decline in water vole populations (Jefferies et al., 2003; Gow, 2008; MacPherson and Bright, 2011). This decline has led to the establishment of the UK Water Vole Steering Group and the development of mink control strategies, such as the Scottish Mink Initiative, part-funded by the SNH Species Action Framework.

Status

Native.

Conservation Status

- IUCN Red List (GB: EN; England: [EN]; Scotland: [NT]; Wales: [CR]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

Accurate maps are difficult to produce for this species; populations can disappear quickly where mink are present and may not recolonise, and variation in recording effort makes it difficult to determine areas of true absence. The National Water Vole Database and Mapping Project has collected presence records for water voles since 2008, and so map coverage is likely to be considerably more thorough than for other species of small rodents. The project does not, however, include systematic survey coverage for water voles, and so it is unclear whether gaps in the species' distribution are caused by low recorder effort. In Wales, neither a National Key Site for Water Voles (Llanelli), nor several other populations, are shown on the smoothed distribution map. The latest evidence collated by the Wales Wildlife Trust is therefore presented for comparison (Figure 7.9a).

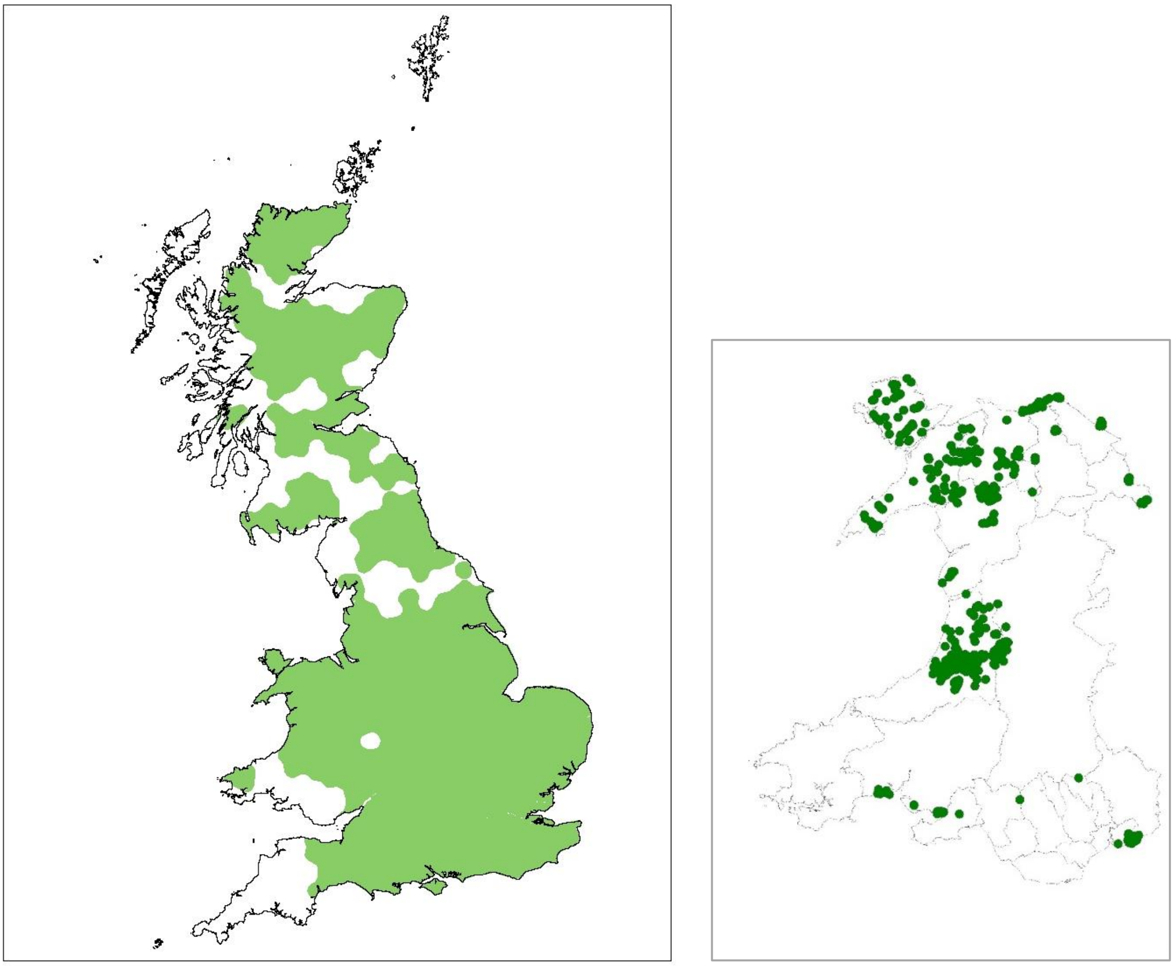


Figure 7.9a Left: Current range of the water vole in Britain. To reflect the current distribution against changes in the species' range through time, the range is based on presence data collected between 2005 and 2016 (rather than 1995 and 2016 as for most other species in this review). Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. Right: Current distribution records for Wales (Wales Wildlife Trust).

Species-specific methods

The lengths of riparian habitat for Scotland, Wales and each English region, were derived from Table 4 of Harris et al. (1995). These lengths were multiplied by the percentage of each region/country included in the species' distribution (Figure 7.9a), and then by the occupancy values per country or region (Table 7.9a). For Scotland, mean occupancy was taken from values in Strachan et al. (2000), Capreolus Wildlife Consultancy (2005), and Reynolds and Telfer (2000). The regional lengths for England were totalled to give an estimate for the whole country. The occupied lengths of riparian habitat were then multiplied by the abundance value (voles per 100m) provided in Table 7.9b, and summed to give the population size per country.

Results

24 papers and 10 government reports were identified by the literature search. Of these, 3 contained pre-breeding population density estimates, with the remainder containing post-breeding density estimates, habitat requirements, occupancy values or presence surveys. Percentage occupancy data are shown in Table 7.9a, population density estimates in Table 7.9b, and population size estimates in Table 7.9c.

Table 7.9a Percentage occupancy of water voles per region of Britain.

Region/Country	Occupancy (%)	River length occupied (km)	Source
North West	8.2	865	Strachan et al. (2000)
Yorkshire	7.1	971	Strachan et al. (2000)
Northumbria	8.7	874	Strachan et al. (2000)
South West	1.9	197	Strachan et al. (2000)
Wessex	23.1	1,830	Strachan et al. (2000)
Anglian	29.8	6,230	Strachan et al. (2000)
Southern	28.1	2,750	Strachan et al. (2000)
Thames	24	2,550	Strachan et al. (2000)
Severn Trent	14.1	3,050	Strachan et al. (2000)
Wales (overall)	5.7	1,130	Strachan et al. (2000)
Cairngorms (Bynack)	9	-	Capreolus Wildlife Consultancy (2005)
Cairngorms (Geldie)	39	-	Capreolus Wildlife Consultancy (2005)
Scotland (overall)	9.6	-	Strachan et al. (2000)
Scotland (Lothians)	2	-	Reynolds and Telfer (2000)
England	**See above**	**19,300**	
Scotland (mean)	**14.9**	**12,500**	
Wales	**5.7**	**1,130**	

Table 7.9b Median density estimates for water voles with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Length within range (km)	Density (per km)	-95%CI (per km)	+95%CI (per km)	Source*	n**	%Occ†
Riparian	219,000	4	3	10	Barreto and MacDonald (2000)	5	See Table 7.9a
					Capreolus Wildlife Consultancy (2005)	2	
					Mutch and Scottish Natural Heritage (2000)	6	
					Oxford (2004)	6	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.9c Length of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.9b with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Length of riparian habitat (km)	Population size	-95%CI	+95%CI
England	115,000	77,200	57,900	193,000
Scotland	83,900	50,000	37,500	125,000
Wales	19,800	4,500	3,400	11,300
Britain	**219,000**	**132,000**	**99,000**	**329,000**

Critique

Water vole population density depends on a number of factors that were not accounted for in our estimate. For example, density will be higher in areas with high vegetative cover and fewer mink. Non-linear wetland areas such as reed beds and grazing marsh can also support high population densities and potentially offer refuges from mink predation (Strachan and Moorhouse, 2006; MacPherson and Bright, 2010): these habitats were not included in our assessment because of a lack of i) data on occupancy, and ii) sufficiently fine resolution habitat data to permit identification of potentially suitable areas. Wider water channels may also contain higher densities of water voles than assumed here because separate populations can form on each bank (Harris et al., 1995), although recent evidence suggests that most surviving populations inhabit upper tributaries rather than main river channels owing to the presence there of mink (Telfer et al., 2001). The distribution of water voles can also change rapidly over time as local populations are lost to mink predation, or to a lesser extent because of habitat change: the occupancy data in this review may therefore be outdated despite being relatively recent (Strachan et al., 2000; Capreolus Wildlife Consultancy, 2005). Continuous monitoring of this species is therefore vital.

Population densities vary between upland and lowland areas, with headwaters offering potential refuges for water voles (Walsh and Hall, 2005). Stratification into lowland and upland areas may provide a more robust population estimate, although more measures of population density would be required to ensure that variation between these areas is represented in the dataset.

A reliability assessment is provided in Table 7.9d.

Table 7.9d Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**
			Riparian
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	1
	2	>30 population density estimates	
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**3**
		Overall reliability score	**3**

Changes through time

Comparison to Harris et al. (1995) and Strachan et al. (2000)

Harris et al. (1995) reported a total population of 1,169,000 water voles in Britain, comprising 752,000 in England, 376,000 in Scotland and 41,000 in Wales. The approaches taken by Harris et al. (1995) and the current review are comparable, except that the former obtained pre-breeding population estimates by adjusting summer population sizes, whereas the current review computed the pre-breeding population size directly from spring density estimates. Both reviews adjusted for the percentage of habitat occupied based on the findings of the Vincent Wildlife Trust's national water vole surveys. Harris et al. (1995) used the 1989-1990 surveys (Strachan and Jefferies, 1993), and the current review used the 1996-1998 surveys (Strachan et al., 2000), supplemented with additional data (see Table 7.9a).

Applying the same method as Harris et al. (1995), Strachan et al. (2000) estimated the overwintering population in 1996-1998 to be 262,500. These figures suggest a 78% decline in water vole population size between 1989-1990 and 1996-1998. The current review suggests a further decrease by 50% for the period 1998-2016. The occupancy values used in our estimate were measured in 1996-1998 (Britain; Strachan et al., 2000), supplemented by data collected in 2005 for upland Scotland (Capreolus Wildlife Consultancy, 2005), and the density estimates were derived in 2000-2005 (see Table 7.9b). Although trends in density are unclear, occupancy had decreased by 80% in most areas between 1989-1990 and 1996-1998 and, despite conservation efforts, the pressures of mink predation and habitat loss mean that this trend is highly likely to have continued. A notable exception may be parts of Scotland where systematic landscape-scale mink control has been conducted (Bryce et al., 2011; Gaywood et al., 2016; Robertson et al., 2017). For example, both upland and lowland regions of Aberdeenshire and the Cairngorms National Park have seen marked recoveries of water voles. Although recolonization is a slow process, particularly where starting population densities are low, water voles are now ubiquitous over large areas (Xavier Lambin, *pers. comm.*). A summary of trends in population size and range is provided in Table 7.9e.

Table 7.9e Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease	Scotland*	England Wales**		
	Data deficient				

* A small increase in range may be the result of successful mink control as well as increased recorder effort.
** Range shifts do appear to have occurred. Decreases in population size are most likely owing to decreased population density and occupancy.

Drivers of change

Table 7.9f Drivers of population change for water voles between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Predation.	Predation by American mink.	Ward (2005) Barreto and MacDonald (2000)	Negative
Habitat quality.	Change in land management: wetland drainage, arable cultivation and watercourse canalisation. Negative effects may be offset in some areas by improvements to water quality driven by the Water Framework Directive.	Gow (2008) Barreto et al. (1998a) Rushton et al. (2000)	Negative
Conservation effort.	Multiple captive breeding and reinforcement projects.	Gow (2008) McGuire et al. (2014)	Positive

Data deficiencies

Table 7.9g Areas where further research is required to improve the reliability of population size estimates for water voles.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Riparian	The most recent density estimates are from 2005.
Density estimates do not represent within-habitat variability.	Riparian	Density estimates are from limited locations only.
Occupancy data are outdated.	Riparian	Occupancy data were from 1996-1998 (and 2005 in upland Scotland), and occupancy is likely to have changed substantially since then.
Density and occupancy data for key habitat types are missing.	Fen, marsh, swamp, and grazing marsh	Systematic data are not available for Great Britain.

Future prospects

Table 7.9h An assessment of the future prospects of the water vole, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline*
Range	Stable
Habitat	Stable

* There is no evidence that recent national trends are likely to change, unless mink control has a major impact.

7.10 Harvest mouse *Micromys minutus*

Habitat preferences

Although predominantly associated with agricultural habitats (Love et al., 2000), the harvest mouse is also frequently found in reed beds, and in undisturbed areas of rough grassland such as road verges (Harris, 1979b; Dickman, 1986). The species is now more commonly found in boundary features such as hedgerows, field margins and ditches, than within the cropped areas of fields (Harris et al., 1995; Moore et al., 2003). Summer foraging and nesting largely take place in the stem-zone of tall vegetation, whereas boundary features and shorter vegetation are used after harvest.

It is probable that the harvest mouse has been adversely affected by changes to agricultural practices, such as the switch to shorter-stemmed cereal varieties (Harris, 1979b), and the transition to winter cereals that are cut before the breeding season (Harris, 1979a). However, it is difficult to quantify the scale of any impacts, not only because of a lack of baseline data, but also because there are large seasonal and annual fluctuations in population size.

Status

Native.

Conservation Status

- IUCN Red List (GB: NT; England: [LC]; Scotland: n/a; Wales: [VU]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.10a. Gaps in the species' distribution in England are likely to represent areas lacking survey effort, rather than true absences. Surveys are therefore recommended in these areas.

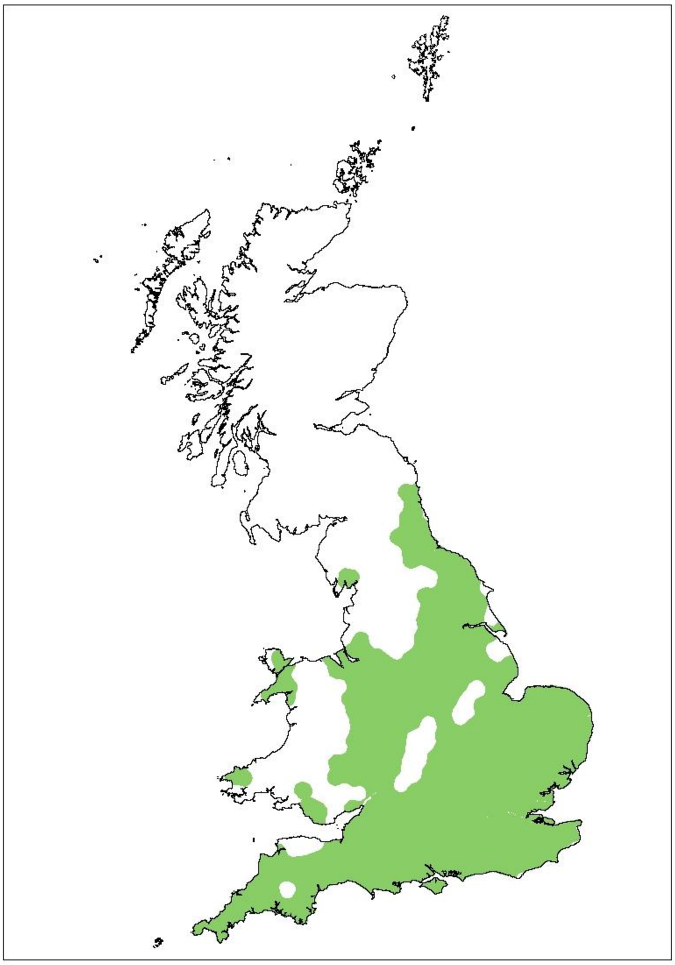


Figure 7.10a Current range of the harvest mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Very few population density estimates exist for the harvest mouse, particularly for the pre-breeding period. Harris et al. (1995) calculated population size on the basis of a ratio of 26.6 wood mice to one harvest mouse. Owing to a lack of pre-breeding population density estimates for harvest mice, the same approach was used in the current review. Additional data are now available that allow this ratio used by Harris et al. (1995) to be updated (Bellamy et al., 2000; Love et al., 2000; Moore et al., 2003; Woods et al., 2003; Askew et al., 2007; see Clapham, 2011).

Wood mouse population sizes per country (and their upper and lower 95% confidence limits) were divided by the geographical range size (extent of occupancy) to produce density estimates (ha^{-1}) for the entire area over which the harvest mouse could potentially be found. These values were converted into harvest mouse densities by dividing by the mean ratio of wood mice:harvest mice. These density estimates were then multiplied by the geographical range size (extent of occupancy) to produce the population estimates.

Results

Eleven papers and one government report were identified by the literature search. One paper contained pre-breeding estimates of population density (Clapham, 2011), and five reported post-breeding estimates and distribution data, or examined the relationship between habitat characteristics and harvest mouse presence. Five publications contained the ratio of wood mice to harvest mice; they are summarised by Clapham (2011) along with the ratios included by Harris et al. (1995) (see Table 7.10a for details). Population densities of harvest mice in relation to wood mice are shown in Table 7.10b, and population size estimates in Table 7.10c.

Table 7.10a Ratio of wood mice to harvest mice (Ratio WM:HM) (after Clapham (2011)). The mean ratio is shown in the final row.

Type of study	**Habitat**	**Ratio (WM:HM)**	**Source**
Trapping — Minimum Number Alive (MNA) (data from 2003)	Arable (new farm woodland, set-aside, field margins)	43:1	Askew et al. (2007)
Trapping — MNA (data from 2004)	Arable (new farm woodland, set-aside, field margins)	69:1	Askew et al. (2007)
Trapping — MNA	Road verges	73:1	Bellamy et al. (2000)
Trapping — MNA	New farm woodland	1:1	Moore et al. (2003)
Trapping — MNA	Farmland	9:1	Moore et al. (2003)
Trapping — MNA	Hedgerows	61:1	Moore et al. (2003)
Barn owl pellet analysis	Unknown	15: 1	Love et al. (2000)
Cat predation questionnaires	Unknown	11: 1	Woods et al. (2003)
Meta-analysis of methods below:	Unknown	27:1	Harris et al. (1995)
Barn owl pellets	*Unknown*	*9:1*	*Harris et al. (1995)*
Short eared owl pellets	*Unknown*	*81:1*	*Harris et al. (1995)*
Bottle samples	*Unknown*	*37:1*	*Harris et al. (1995)*
Trapping samples	*Unknown*	*58:1*	*Harris et al. (1995)*
Mean ratio		**34:1 (SE 9)**	

Table 7.10b Area of occurrence (geographical range as defined by alpha shape) for wood mice, and density for wood mice and harvest mice. For wood mice, density was calculated by dividing the total population size (see Table 7.11b) by area. For harvest mice, density was calculated by dividing the density of wood mice (this table) by the ratio of wood mice to harvest mice (Table 7.10a).

	Wood mouse				Harvest Mouse		
Country	**Area (ha)**	**Density (ha^{-1})**	**-95CI**	**+95CI**	**Density (ha^{-1})**	**-95CI**	**+95CI**
England	12,759,000	1.45	0.67	2.32	0.04	0.02	0.07
Wales	2,005,000	1.95	0.84	3.21	0.06	0.02	0.09

Table 7.10c Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 7.10b by the area of distribution.

Country	**Area of distribution (ha)**	**Population size**	**-95%CI**	**+95%CI**
England	10,164,000	[532,000]	[272,000]	[879,000]
Wales	504,000	[34,000]	[16,600]	[55,700]
Britain	**10,668,000**	**[566,000]**	**[288,000]**	**[934,000]**

Critique

Harvest mouse population estimates are extremely difficult to make with any level of certainty. Only one pre-breeding population density estimate has been published since 1995 (Clapham, 2011), with very few estimates prior to this. Populations are thought to have a clumped distribution (Harris, 1979b), with large seasonal fluctuations (Gosling and Baker, 2008). Evidence of harvest mouse presence (nests) is easily overlooked in surveys, and the species is difficult to trap in spring and summer as it is rarely found at ground level at that time of year (Harris et al., 1995). Consequently, any direct estimate of population size would be subject to considerable error.

Our estimate is based on the mean ratio of wood mice to harvest mice. Yet the primary habitats for wood mice do not necessarily correspond with those for harvest mice (the former being highly dependent on woodland and the latter on long grass). Our estimation of harvest mouse numbers makes the assumption that the ratios shown in Table 7.10a are representative across the geographical range. No account is taken of the differing areas of each habitat. Therefore, rarer habitats may be over-represented in the mean ratio. Harvest mice also occur in some habitats, such as fenland, for which no ratios are available. The ratios in many habitats are likely to be unreliable because data are sparse (e.g., for road verges); even so, information for all habitats has been weighted equally. A reliability assessment is shown in Table 7.10d.

Table 7.10d Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All habitats
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**0**
		Overall reliability score	**0**

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated the population size of harvest mice to be 1,425,000, with 1,415,000 in England and 10,000 in Wales. These estimates, however, were not adjusted to take into account the smaller distribution of harvest mice compared to wood mice. Rather, the population size in Britain was calculated by dividing the population size of wood mice by the ratio of wood mice to harvest mice, and proportions of this value were assigned to countries *post hoc*. This is likely to have overestimated the total population size, as wood mice are present in a larger area of Britain than harvest mice. Reassessment of the data from Harris et al. (1995) using the method presented here (i.e. (wood mouse population size/current wood mouse distribution area/ratio of wood mice to harvest mice) * area of harvest mouse distribution)), suggests a total population size in Britain of 793,000, although this method assumes that the species' range has remained constant since 1995.

The reassessed population size from 1995 falls within the confidence limits of the current estimate. Further surveys are therefore suggested to improve the precision of population size estimates and to allow for an assessment of trends.

Other evidence of changes through time

Information about harvest mouse density is very sparse, and no other reports of temporal trends are currently available. A summary of trends in population size and range is provided in Table 7.10e.

Table 7.10e Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				England Wales

Drivers of change

Table 7.10f Drivers of population change for harvest mice between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Environmental conditions.	Increase in wet summers limits range expansion.	Harris et al. (1995) Sutton and Dong (2012)	Negative
	Warmer climate may increase fecundity.	Harris (1979a)	Positive
Habitat quality — decline.	Changes in agricultural practice, such increased use of winter-sown crops which are harvested earlier in the summer, likely to result in loss of nests and young.	Perrow and Jowitt (1995) Boatman et al. (2007)	Negative

Data deficiencies

Table 7.10g Areas where further research is required to improve the reliability of population size estimates for harvest mice.

Data deficiencies	Habitat	Details
No density estimates for key habitat.	Hedgerows, woodland edges, reed beds, rough grassland	
Limited density estimates for key habitat.	Arable land	
No occupancy data.	Arable land, hedgerows, woodland edges, reed beds, rough grassland	

Future prospects

Table 7.10h An assessment of the future prospects of the harvest mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Stable
Habitat	Decline

7.11 Wood mouse *Apodemus sylvaticus*

Habitat preferences

The wood mouse is highly adaptable and is found in most habitats, including woodland, arable land, rough grassland, heather, blanket bog, sand dunes, urban areas and hedgerows (Kotzageorgis and Mason, 1997; Marsh and Harris, 2000; Flowerdew and Ellwood, 2001; Tattersall et al., 2001). Population densities in woodland vary with successional stage: mid-level regeneration of 5- to 10-year-old vegetation supports a higher density of wood mice than either ungrazed fields or 10-year-old regeneration (Marsh and Harris, 2000).

Hedgerows, including those distant from woodlands, are an important habitat for the wood mouse, and can support resident populations (Gelling et al., 2007). Population densities in hedgerows are high after arable crops are harvested, and also when grass swards are short because of grazing or cutting (Tew and Macdonald, 1993; Garratt et al., 2012). At these times, the wood mouse, like other small mammals, makes preferential use of boundary features rather than in-field areas (Tattersall et al., 2001). Similarly, hedgerows provide cover and food sources during autumn and winter (Kotzageorgis and Mason, 1997; Liu et al., 2013).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.11a. Gaps in the mainland species' distribution are likely to reflect a local lack of survey effort, rather than true absences. Further surveys are therefore recommended in these areas.

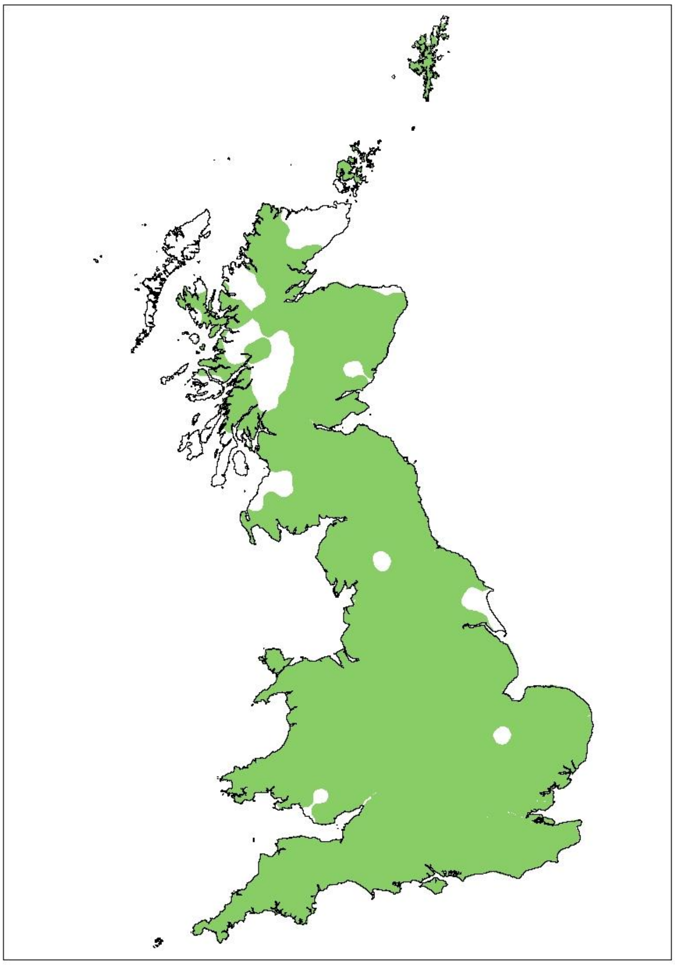


Figure 7.11a Current range of the wood mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Wood mouse populations fluctuate seasonally, with peaks in autumn and early winter and troughs in spring and summer (Flowerdew, 1985; Harris et al., 1995). Density estimates were therefore derived only from studies which took place between March and June. As wood mice in arable land primarily use field margins, with incorporation of crop fields into home ranges only before the harvest (Tattersall et al., 2001), we assume that the in-field population will be included within the estimate for hedgerows. Arable land is therefore excluded from the analysis.

No recent density estimates were available for fen, marsh and swamp habitat. Had the estimates from Harris et al. (1995) for this habitat been included in the present review, confidence intervals would not have been calculable as none were provided in the original paper. In addition, the exclusion of fen, marsh and swamp altered the population estimate by <100,000 (<1%). This habitat class was therefore excluded.

Results

Eighteen relevant papers were identified by the literature search. Of these, five provided an estimate of pre-breeding population density, four provided a relative measure of abundance (captures per trap night), and one contained percentage occupancy for hedgerows (Gelling et al., 2007). The remainder provided post-breeding density estimates, explored the relationship of habitat variables to abundance but gave no effect size, or provided descriptive data only. Population density estimates are shown in Table 7.11a, and population size estimates in Table 7.11b.

Table 7.11a Median density estimates for wood mice, with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Hedgerows are divided into those managed or not through an agri-environment scheme (AES and non-AES). Hedgerow length and density per hedgerow are presented in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	1,270,000	8.64	7.41	9.9	Attuquayefio et al. (1986)	1	n/a
					Flowerdew et al. (2004)	162	
					Malo et al. (2012)	1	
					Marsh and Harris (2000)	19	
					Montgomery (1989)	158	
Sand dunes	21,800	1.2	0.8	3.5	Attuquayefio et al. (1986)	1	n/a
					Gorman and Ahmad (1993)	8	
Urban and gardens	1,360,000	4	0.05	8	expert opinion	2	n/a
Coniferous woodland	1,250,000	6.75	5.6	7.95	expert opinion	2	n/a
Dwarf shrub heath	1,540,000	1.5	0.1	3	expert opinion	1	n/a
Improved grassland	6,760,000	0.1	0	0.5	expert opinion	2	n/a
Unimproved grassland	1,060,000	6.5	0	13	expert opinion	2	n/a
Hedgerows (AES)	142,000 (km)	3	0	12	Broughton et al. (2014)	11	93.9
					Flowerdew et al. (2004)	9	
Hedgerows (Non-AES)	320,000 (km)	14.62	12.31	24.5	Kotzageorgis and Mason (1997)	6	93.9
					Shore et al. (2005)	23	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.11b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 7.11a with the area of habitat within the species' distribution, and adjusting for occupancy where known.

Country	Area of suitable habitat (ha)*	Population size	-95%CI	+95%CI
England	7,260,000	22,700,000	11,600,000	37,800,000
Scotland	4,240,000	12,300,000	6,510,000	18,800,000
Wales	1,770,000	4,600,000	2,240,000	7,680,000
Britain	**13,300,000**	**39,600,000**	**20,400,000**	**64,300,000**

* The lengths of hedgerows are 393,000km in England, 17,600km in Scotland, and 51,000km in Wales.

Critique

No percentage occupancy data were available for most habitats; the population size is therefore overestimated. The population estimate is largely derived from broadleaved (28%) and coniferous (21%) woodland (Figure 7.11b), where population density estimates are supported by five (n=311) and four (n=49) references, respectively. Given the abundance of evidence relative to most other species in this review, sensitivity analyses were not conducted for these habitats.

Improved grassland forms 49% of the habitat within the geographical range of the wood mouse (Figure 7.11b), yet because of low population density ($0.1ha^{-1}$), it contributes just 2% of the estimated population. The density used for improved grassland is the median value from estimates provided by three expert opinions: given the extent of this habitat, field validation of the values would considerably improve the precision of the population estimate. A reliability assessment is shown in Table 7.11c. For density estimates based on expert opinion, a conservative score of 1 has been applied to the 'location of study sites' section.

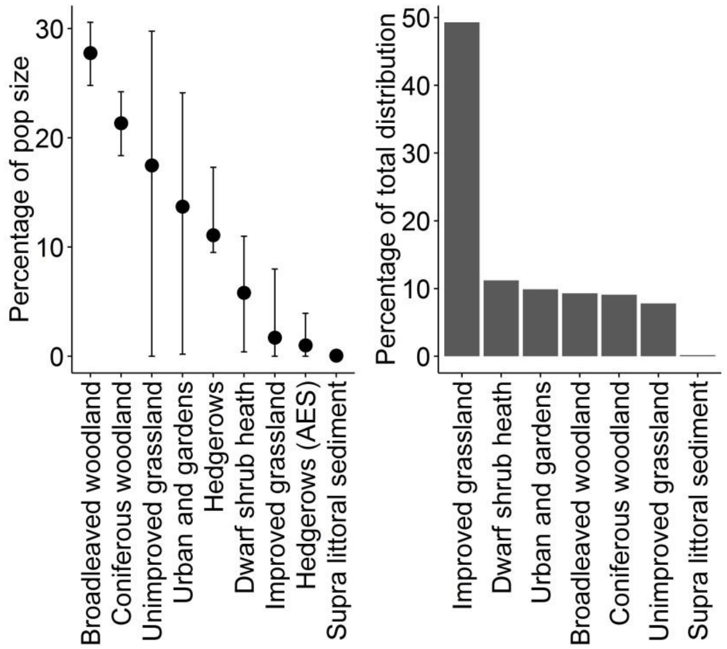


Figure 7.11b Left: The percentage of the total population of wood mice accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.11c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Broadleaved woodland	**Coniferous woodland**	**Improved grassland**
Location of study sites	0	Estimates from one location			
	1	Estimates restricted		1	1
	2	Estimates widespread	2		
Sample size	0	<10 density estimates		0	0
	1	10-30 density estimates			
	2	>30 density estimates	2		
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**4**	**1**	**1**
		Overall reliability score	**2**		

Changes through time

Comparison to Harris et al. (1995)

Population estimates from Harris et al. (1995) were derived from densities reported in the literature and by experts. The habitat classes were equivalent to those used in the current review. Population size was estimated as 38,000,000 in total, with 19,500,000 in England, 15,000,000 in Scotland and 3,500,000 in Wales. These figures are all within our confidence limits, so there is no evidence of a significant change in population size since 1995.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size that differs from the original estimate by only 5%, and that falls within the confidence limits of the original. It is therefore concluded that methodological differences have no material impact on the comparisons between the two time periods.

Other evidence of changes through time

No other references to a temporal trend in population size were found in the literature. A summary of trends in population size and range is provided in Table 7.11d.

Table 7.11d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable		All countries		
	Decrease				
	Data deficient				

Drivers of change

Table 7.11e Drivers of population change for wood mice between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Unknown			

Data deficiencies

Table 7.11f Areas where further research is required to improve the reliability of population size estimates for wood mice.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Urban and gardens Coniferous woodland Dwarf shrub heath Improved grassland Unimproved grassland	Estimates are based on expert opinion.
No occupancy data.	All habitats except hedgerows	
Density estimates are more than 10 years old.	Sand dunes Hedgerows	The most recent density estimates are from 1993 and 2005, respectively.
Limited density estimates are available for specified habitat.	Sand dunes	Nine individual density estimates are available.

Future prospects

Table 7.11g An assessment of the future prospects of the wood mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

7.12 Yellow-necked mouse *Apodemus flavicollis*

Habitat preferences

The yellow-necked mouse is found primarily in mature broadleaved woodlands (Marsh et al. 2008), particularly ancient coppiced woodlands, where they favour older established compartments, rather than recent coppice (Gurnell et al., 1992; Capizzi and Luiselli, 1996). Hedgerows also provide an important habitat for the species in Britain, with telemetry studies showing that individuals can reside solely within a linear hedgerow habitat (Montgomery, 1978). The availability and diversity of tree seeds is an important predictor of yellow-necked mouse density (Marsh et al., 2001). Britain is at the western edge of the species' European range, possibly owing to the impact of low summer temperatures on tree seed abundance. There is some potential for misidentification of this species with the wood mouse, particularly at the edges of its geographical range where abundance may be relatively low.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: n/a; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

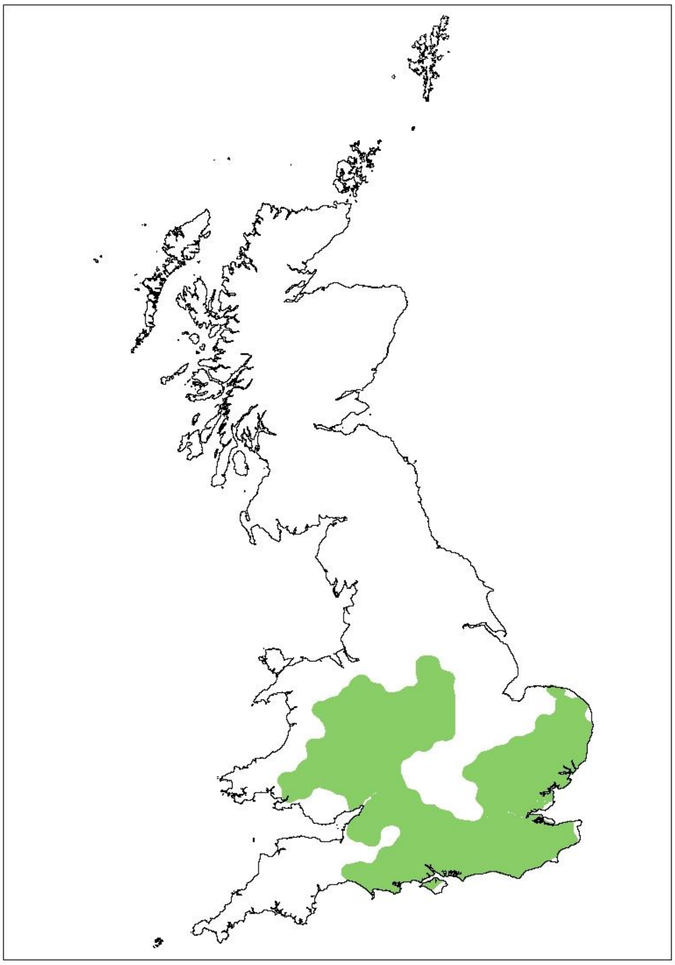


Figure 7.12a Current range of the yellow-necked mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

The percentage of occupied sites for broadleaved woodland was taken from Marsh et al. (2001) where 80 of 146 sites (55%) within the species' range were occupied. Only non-zero population density estimates were included to avoid accounting for occupancy twice. The total area of broadleaved woodland within the species' distribution was multiplied by the percentage of occupied sites. The occupied area of broadleaved woodland was then used for all subsequent calculations of population size. In the absence of occupancy data for coniferous woodlands and urban areas, the same value (55%) was applied to these habitats. For hedgerows, percentage occupancy was taken from Gelling et al. (2007), where 180 hedgerows on 12 dairy farms in 4 geographical areas within the species' range were surveyed. Occupancy (75%) was provided for one of these areas, but in the other areas very few yellow-necked mice were captured. To provide a more representative value for the species throughout its range, an average value of percentage occupancy was taken across all four areas, where the number of sites surveyed across all areas was assumed to be equal, and percentage occupancy at the remaining three sites was assumed to be roughly zero. As two of the sites were located towards the edge of the species' range where densities might be expected to be lower, the resulting value may be a slight underestimate.

Results

Six papers contained useful information for yellow-necked mice. Three of these reported pre-breeding population density (Montgomery, 1980; Kotzageorgis and Mason, 1997; Marsh and Harris, 2000), one gave occupancy for hedgerows (Gelling et al., 2007), and one gave occupancy for broadleaved woodland (Marsh et al., 2001). The remaining papers reported only post-breeding density estimates (Marsh et al., 2001; Moro and Gadal, 2007). Population density estimates are shown in Table 7.12a, and population size estimates in Table 7.12b.

Table 7.12a Median density estimates for yellow-necked mice with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015. Median population density from Marsh et al. (2001) was taken from positive sites only. Hedgerow length and density per hedgerow are presented in km and km^{-1}, respectively.

Habitat	Area within range (ha)	Density (per ha^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	592,000	3.12	0.57	11	Montgomery (1980) Marsh and Harris (2000)	12 13	55%
Urban and gardens	720,000	0.38	0.12	0.8	expert opinion	2	55%
Coniferous woodland	160,000	0.16	0.08	0.52	expert opinion	2	55%
Hedgerows	200,000 (km)	8.5	6	11	Kotzageorgis and Mason (1997)	6	19%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.12b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying density estimates in Table 4.1a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (ha)	Population size	-95%CI	+95%CI
England	1,330,000	1,360,000	426,000	3,940,000
Wales	146,000	140,000	40,600	423,000
Britain	**1,470,000**	**1,500,000**	**467,000**	**4,360,000**

* The lengths of hedgerows are 185,000km in England and 15,300km in Wales.

Critique

68% of the population estimate for the yellow-necked mouse is derived from broadleaved woodland, a habitat which forms 40% of the species' range (Figure 4.1a). The density values applied for broadleaved woodland are based on 25 separate estimates from two papers, although the confidence limits are wide (0.57-11ha^{-1}), reflecting highly variable densities across the geographical range. Further surveys, specifically designed to include a representative sample, would improve confidence in the estimate.

Hedgerows contribute 21% of the estimated population, and there are 200,000km available within the species' range (Table 7.12a, Figure 7.12b). The abundance estimate for hedgerows is derived from 6 estimates from one paper, with the resulting median density estimate being substantially higher than that in broadleaved woodland. Re-calculation of population size following stepwise removal and replacement of the individual density estimates for hedgerows did not result in a significant change in population size. Reliability scores are shown in Table 7.12c. For the purposes of this assessment, we consider the population density estimates of experts to be representative of a restricted area of the species' range.

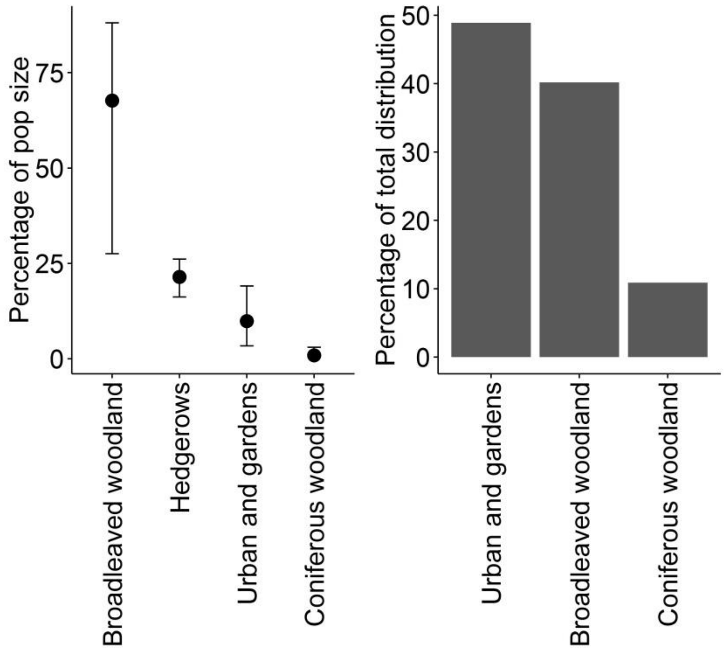


Figure 7.12b Left: The percentage of the total population of the yellow-necked mouse accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type. Linear features (hedgerows) have been omitted.

Table 7.12c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Broadleaved woodland	Urban and gardens
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 density estimates		0
	1	10-30 density estimates	1	
	2	>30 density estimates		
Occupancy data available?	0	No		
	1	Yes	1	1
		Habitat score	**3**	**2**
		Overall reliability score	**2.5**	

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated the total population size for Britain as 750,000, with 662,500 in England and 87,500 in Wales. The British estimate was based on an estimate of 450,000 for ancient woodlands, with the remaining 300,000 added to allow for yellow-necked mice in other habitats. The current review uses broadleaved woodland (adjusted for 55% occupancy), rather than ancient woodland.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, coniferous woodland and improved grassland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a population size that differs from the original estimate by only 5%, and that falls within the confidence limits of the original. It is therefore concluded that methodological differences have no material impact on the comparisons between the two time periods.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is provided in Table 7.12d.

Table 7.12d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	England* Wales			

* Limited expansion into the Midlands.

Drivers of change

Table 7.12e Drivers of population change for yellow-necked mice between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Decline in habitat quality.	Change in the management of ancient/coppiced woodlands, although evidence for the effect on yellow-necked mice is sparse.	Harris et al. (1995)	Negative
Climate change, causing a rise in summer temperature.	Range expansion: the availability of food (tree seeds) is linked to high summer temperature.	Marsh et al. (2001)	Positive

Data deficiencies

Table 7.12f Areas where further research is required to improve the reliability of population size estimates for yellow-necked mice.

Data deficiencies	Habitat	Details
No density estimates for the specified habitat.	Urban areas and gardens Coniferous woodland	Density estimates are based on expert opinion only.
Limited density estimates for key habitat.	Hedgerows	Fewer than 10 density estimates available.
Density estimates are more than 10 years old, highly variable, and possibly not representative of the species' geographical range.	Broadleaved woodlands Hedgerows	Estimates are from 2000.

Future prospects

Table 7.12g An assessment of the future prospects of the yellow-necked mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Potential increase
Range	Potential increase
Habitat	Stable

7.13 House mouse *Mus musculus*

Habitat preferences

The house mouse lives commensally with humans: its movement patterns and current widespread distribution are attributed to this relationship (Searle et al., 2009). Although listed as native to Britain by IUCN, the best available evidence suggests the species arrived in western Europe in the Bronze age and is recorded in Britain by the Iron age (Yalden, 1999). The species is therefore not subject to many of the environmental factors that regulate the population sizes of most other small mammals. It is, however, sensitive to human activities, including the alteration of buildings and the deployment of rodenticide (Pocock et al., 2004). The decline in urban infestations in the 1970s is likely to be the result of increased rodenticide efficacy (Richards, 1989; Harris et al., 1995). Detailed analyses of the habitat preferences of the house mouse are, however, lacking.

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: LC; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.13a. Gaps in the distribution in England and south Wales are likely to reflect a lack of survey effort, rather than true absences. It is unclear whether larger gaps elsewhere in Wales, northern England and Scotland reflect a lack of recording or true absences, so further survey effort is recommended in these regions. There is also potential for misidentification, particularly in the winter when wood mice and yellow-necked mice make greater use of buildings, and this may have produced inaccuracies in the mapped range.

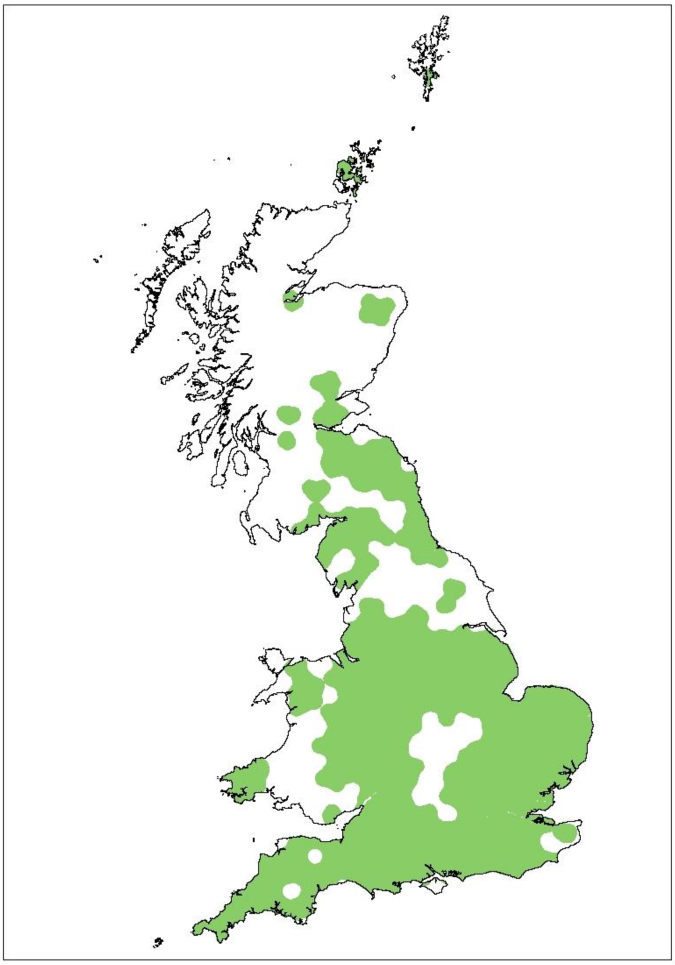


Figure 7.13a Current range of the house mouse in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Although house mice are found in habitats such as field margins and woodland, they are poor competitors with other rodents, particularly wood mice (Berry and Tricker, 1969;

Tattersall et al., 1997). In one study in pastoral farmland in Ireland, although 10% of the rodents captured in field margins in summer were house mice, no house mice were captured in field margins in the spring (Montgomery and Dowie, 1993). No population density estimates were identified in the literature review for habitats other than buildings, so the population estimate is based on buildings only.

The number of farm holdings per country was taken from the Agriculture category in the UK 2015 key statistics dataset (Office for National Statistics). The number of dwellings per country was derived from the 2014-2015 dwelling stock reports from the English, Scottish and Welsh governments. The numbers of dwellings considered ‘urban’ and ‘rural’ were calculated using percentage of residences that are classed as urban and rural from the 2011 census analysis (Office for National Statistics, 2013). Rural residences were divided into farms (farm holdings per country) and other rural dwellings (hereafter ‘rural’) by subtracting the number of farms from the total rural dwellings.

For the population of house mice in farm buildings, it was assumed that, on average, each farm contains the same number of buildings as the study farms in Pocock et al. (2004). The total population was calculated as ‘number of occupied dwellings * population size per farm’. For urban and rural buildings, data based on surveys by Rennison and Drummond (1984) were taken from Harris et al. (1995); the number of mice per infestation (4.5) was multiplied by the number of occupied dwellings, where 3.8% of urban and 5.6% of rural buildings were reported as occupied.

There is evidence that population size on farms does not vary by season (Pocock et al., 2004), so results from all seasons were included in the analysis.

Results

Two papers were identified by the literature search, both of which contained population sizes in farm buildings. One paper (Quy et al., 2009) was excluded as populations were artificially maintained and so did not represent natural population sizes. The number of mice estimated per holding, and percentage occupancy, are shown in Table 7.13a. The number of dwellings, adjusted for occupancy, is given in Table 7.13b, and total population sizes in Table 7.13c.

Table 7.13a Median estimates for house mice per holding with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	No. (per holding)	-95%CI	+95%CI	Source*	n**	%Occ†
Farm buildings	27	18.5	31.5	Pocock et al. (2004)	25	5.6
Urban buildings	4.5	-	-	Harris et al. (1995)	-	3.8
Rural buildings	4.5	-	-	Harris et al. (1995)	-	5.6

* Literature sources.
** Number of estimates from each literature source. For Pocock et al. (2004), 25 separate monthly capture sessions at 2 adjacent farms were considered as separate estimates.
† Percentage of this habitat that is occupied within the known range.

Table 7.13b Number of occupied dwellings taken from the Agriculture category in the UK 2015 key statistics dataset, Department of National Statistics (farm buildings), and the 2014-2015 dwelling stock reports from the English, Scottish and Welsh governments (urban and rural buildings).

Country	Building type	Occupied dwellings
England	Farm	5,740
	Urban	696,825
	Rural	233,100
Scotland	Farm	1,949
	Urban	78,478
	Rural	26,252
Wales	Farm	2,929
	Urban	43,358
	Rural	14,504

Table 7.13c Total population size estimates, with 95% confidence intervals, for house mice. Values were obtained by multiplying the number of house mice per holding in Table 7.13a with the number of occupied dwellings in Table 7.13b. It was not possible to calculate confidence intervals as none were available for density estimates from Harris et al. (1995).

Country	Total number of dwellings*	Population size	-95%CI	+95%CI
England	22,602,500	4,340,000	-	-
Scotland	2,568,800	523,900	-	-
Wales	1,452,300	339,000	-	-
Britain	**26,623,600**	**5,203,000**	**-**	**-**

* Total number of urban, rural and farm dwellings. Percentage occupancy is not applied to these figures.

Critique

House mouse populations exhibit boom-and-bust fluctuations, depending largely on resource availability and rodenticide use. It is therefore difficult to make precise population estimates (Harris et al., 1995). The assessment was based on house mouse density in buildings. The values used for farm buildings were taken from a single paper that studied two adjacent farms (Pocock et al., 2004), and the overall value per holding was multiplied by the number of farm dwellings. Adjustments for occupancy were made on the assumption that the proportion of farm buildings occupied by house mice was the same as for rural houses generally (for which some data were available). The figures are therefore likely to provide a reasonable estimate of the numbers of animals across farm buildings of all types. However, the extent to which these farms are typical of those found nationally is unclear.

The house mouse uses habitats other than buildings, such as field margins and woodland. However, no density or occupancy estimates were available for these habitats, so the population size is underestimated. A reliability assessment is shown in Table 7.13d.

Table 7.13d Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Buildings
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 density estimates	
	1	10-30 density estimates	1
	2	>30 density estimates	
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**2**
		Overall reliability score	**2***

* It is likely that most of the population is found in buildings, and therefore, unlike the brown rat (see Table 7.14d), no adjustment was made to the reliability score to reflect the lack of information for these habitats.

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total population size of 5,192,000, comprising 4,535,000 in England, 657,000 in Scotland and 206,000 in Wales. Most of the data used for the current review were the same as those used by Harris et al. (1995), with the following exceptions: Harris et al. (1995) included density estimates for arable and pastoral habitats (based on Montgomery and Dowie (1993) and Rowe et al. (1983)), but these were not included in the current estimate; conversely, evidence on population density in farm buildings (Pocock et al., 2004) was available for inclusion in the current review. The house mouse primarily occupies urban and rural dwellings, so these differences are unlikely to affect the population size estimate significantly. Although the area of urban land in the current analysis differs by 45% compared to the area quoted in Harris et al. (1995), the population of house mice was calculated from the number of dwellings, rather than the total area, so this difference does not affect the conclusions. It was not possible to calculate confidence limits for the current population size estimates, but they differ from those in Harris et al. (1995) by less than 1%.

Other evidence of changes through time

No other evidence of temporal trends was found in the literature. A summary of trends in population size and range is provided in Table 7.13e.

Table 7.13e Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable		England Wales	Scotland*	
	Decrease				
	Data deficient				

* Decline in range size in Scotland may be the result of changes in recording effort.

Drivers of change

Table 7.13f Drivers of population change for house mice between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Management (control).	Pest control measures have resulted in a reduction on urban infestations.	Harris et al. (1995)	Negative

Data deficiencies

Table 7.13g Areas where further research is required to improve the reliability of population size estimates for house mice.

Data deficiencies	Habitat	Details
Density estimates do not represent within-habitat variability.	Urban and rural buildings	No range or confidence intervals were available.
Density estimates are more than 10 years old.	Farm buildings	Density estimates are from 2004.
	Urban and rural buildings	Density estimates are from Harris et al. (1995).
Unclear whether presence data are truly reliable.	All	A high probability of confusion with other small mammals found in buildings. A survey to estimate the proportion of all 'infestation' reports that are actually house mice would resolve this issue.

Future prospects

Table 7.13h An assessment of the future prospects of the house mouse, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

7.14 Brown rat *Rattus norvegicus*

Habitat preferences

The brown or Norway rat is an adaptable and versatile species. It prefers habitats with dense cover, readily available water and an abundance of food resources. Prevalent in rural farm buildings, brown rat populations also occur in other rural habitats, including hedgerows, ditches and riparian environments. Densities here vary before and after harvest. Substantial populations also exist in urban areas, where they are typically associated with refuse tips, urban waterways, warehouses, older sewers, and other areas where human food waste is available, such as the vicinity of markets and fast-food outlets (Channon et al., 2006). In urban environments they inhabit buildings, make use of refuges such as sewers, and also build burrows (e.g., into banks of rivers and canals). Populations independent of humans occur in many coastal habitats, particularly salt marshes, and in grasslands (see Harris and Yalden, 2008).

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC.).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is shown in Figure 7.14a. Gaps in the species' distribution in England and Wales are likely to represent areas lacking survey effort, rather than true absences. It is unclear whether some of the larger gaps in Scotland reflect a lack of recorder effort or true absences, although the range is highly likely to have been underestimated (Tony Mitchell-Jones, pers. comm.). Further survey effort is recommended in these areas to increase confidence in the current distribution.

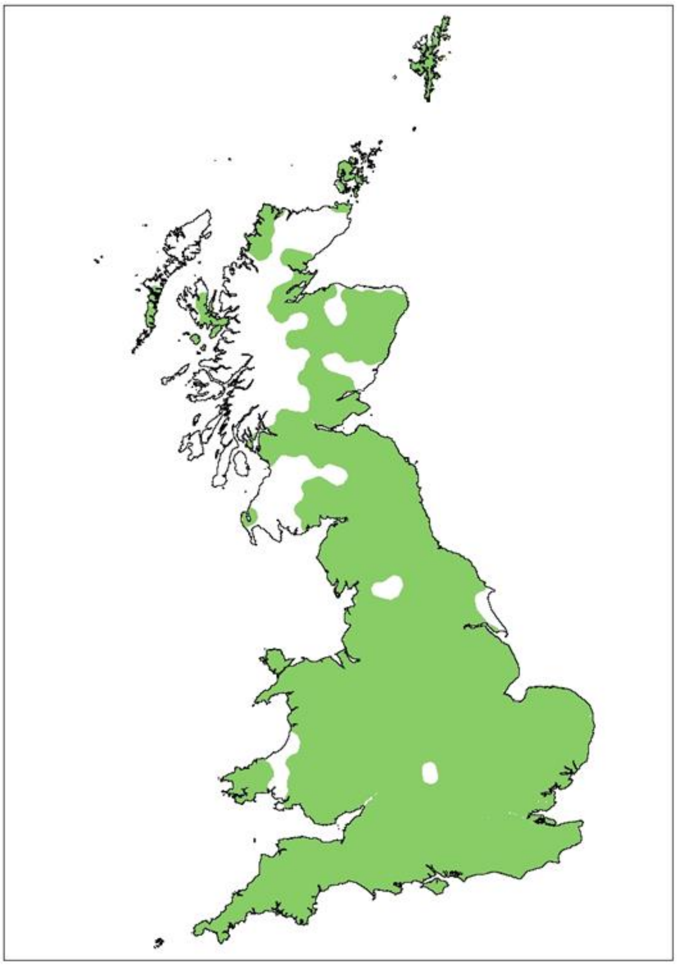


Figure 7.14a Current range of the brown rat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. The areas of absence in Scotland and elsewhere may also reflect under-recording.

Species-specific methods

The total number of farm holdings per country was taken from the Agriculture category in the UK 2015 key statistics dataset (Office for National Statistics). The number of dwellings per country was taken from the 2014-2015 dwelling stock reports from the English, Scottish and Welsh governments. Density and occupancy estimates were available only for two categories of building: ‘farm’ and ‘non-farm’. The availability of non-farm buildings was obtained by subtracting the number of farms from the total number of dwellings.

The percentage of occupied non-farm dwellings was taken from the most recent English House Condition Surveys (EHCS; Department for Communities and Local Department for Communities and Local Government, 2015) as the summed percentage of indoor and outdoor dwellings infested by rats, whilst the percentage of occupied farm buildings was taken from Harris et al. (1995). The population estimate was obtained by multiplying the mean estimate of rats per holding by the availability of buildings and the relevant percentage occupancy.

Results

Four papers were identified by the literature search, none of which contained pre-breeding population density estimates. Two papers contained percentage occupancy values for urban dwellings, including rats present outside as well as inside; one paper contained measures of relative abundance; and one outlined the eradication of brown rats from Lundy. Median estimates of population density are shown in Table 7.14a. The number of dwellings, adjusted for occupancy, is given in Table 7.14b, and total population sizes in Table 7.14c.

Table 7.14a Median estimate of brown rats per holding with 95% confidence intervals, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Estimate (per holding)	-95%CI	+95%CI	Source*	n**	%Occ†
Farm buildings	60	-	-	Harris et al. (1995)	-	7.8
Non-farm buildings	2.2	-	-	Harris et al. (1995)	-	4.0

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 7.14b Median estimate of dwellings occupied by brown rats, calculated using data obtained from a review of the literature from 1995 to 2015.

Country	Building type	Occupied dwellings
England	Farm	46,125
	Non-farm	891,000
Scotland	Farm	13,920
	Non-farm	100,346
Wales	Farm	19,351
	Non-farm	55,440

Table 7.14c Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates, with 95% confidence intervals. Values were obtained by multiplying infestation size estimates in Table 7.14a with the number of occupied dwellings in Table 7.14b. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Number of dwellings*	Population size	-95%CI	+95%CI
England	22,603,000	4,730,000	-	-
Scotland	2,569,000	1,060,000	-	-
Wales	1,452,000	1,280,000	-	-
Britain	**26,600,000**	**[7,070,000]**	**-**	**-**

* Total number of urban, rural and farm dwellings. Percentage occupancy is not applied to these figures.

Critique

Most data, including population density estimates for both urban and farm dwellings and percentage occupancy for farm dwellings, were taken from Harris et al. (1995), which, in turn, was based on very few studies. Percentage occupancy for urban dwellings was taken from the EHCS (Department for Communities and Local Government, 2015). Occupancy is likely to vary between dwellings in different areas (e.g., cities and rural), but this could not be accounted for with current data.

Owing to the lack of data on the size or density of brown rat populations in habitats other than dwellings, the current estimate does not account for populations in other types of man-made structures, or for human-independent populations. This could be a very significant source of underestimation, and one that is not captured by the reliability scores shown below (Table 7.14d). The use of data from Harris et al. (1995), and lack of density estimates across habitat types, mean that a sensitivity analysis is not possible.

There is likely to be some under-recording of the distribution of brown rat because relatively few records are submitted to local biological records centres. The apparent gaps in the distribution in Scotland and elsewhere may therefore be artefacts of recording effort.

For the reliability assessment, we have considered the population density estimates from Harris et al. (1995) to be the expert opinion of the authors and, therefore, to be representative of a restricted area of the species' range (Table 7.14d).

Table 7.14d Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Dwellings
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 density estimates	0
	1	10-30 density estimates	
	2	>30 density estimates	
Occupancy data available?	0	No	
	1	Yes	1
		Total score	**2**
		Overall reliability score	**1****

* Populations may be unstable owing to inter-annual cycles or fluctuations in population size, or as a result of management.
** The overall reliability score is reduced to 1 because although data are only available for dwellings and their immediate environs, a substantial proportion of the population is likely to occur in other habitats, including commercial and farm buildings, riparian habitats and agricultural land.

Changes through time

Comparison to Harris et al. (1995)

The British population size for brown rats was estimated as 6,790,000 by Harris et al. (1995), comprising 5,240,000 in England, 870,000 in Scotland and 680,000 in Wales. The current estimate is largely based on the same data included in Harris et al. (1995), except for numbers of, and occupancy rates for, dwellings. Differences in population size are, therefore, owing to changes in these values. The comparison of the two reviews suggests that the population is approximately stable (7% increase but no confidence limits are available). However, the data are poor, and no information is available for habitats other than dwelling houses.

Differences in the way that landscape composition was measured between the current review and Harris et al. (1995) are not relevant to brown rats because estimates were based on the number of occupied dwellings rather than habitat-specific densities.

Other evidence of changes through time

There are very few studies on the brown rat, and no recent documented trends in population size. A summary of trends in population size and range is provided in Table 7.14e.

Table 7.14e Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		England Wales	Scotland	

Drivers of change

Table 7.14f Drivers of population change for brown rats between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Management (control).	Localised suppression, the effect on total population being uncertain.		Negative
Management (control).	Development of resistance to anticoagulant poisons. There has been no national survey to determine the level of resistance, but it is known that resistance compromises control in some local populations, and it is reasonable to infer wider-scale effects.	Buckle (2013)	Positive

Data deficiencies

Table 7.14g Areas where further research is required to improve the reliability of population size estimates for brown rats.

Data deficiencies	Habitat	Details
No population density estimates for specified habitat.	Coastal habitats Salt marsh All agricultural habitats Buildings not used as dwellings	No published density estimates in the recent literature.
Density estimates are more than 10 years old.	Urban and gardens	Density estimates are taken from Harris et al. (1995).
Density estimates do not represent within-habitat variability.	Urban and gardens	No range or confidence limits were available.

Future prospects

Table 7.14h An assessment of the future prospects of the brown rat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Increase
Range	Stable
Habitat	Increase

7.15 Black rat *Rattus rattus*

Habitat preferences

The black rat is a commensal species with an omnivorous diet, although it is notably more vegetarian than the brown rat (Harris and Yalden, 2008). Dietary investigations on the Shiant Islands in the Hebrides provide evidence for the consumption of seabirds during the nesting season (McDonald et al., 1997; Stapp, 2002), but the extent to which these are scavenged rather than actively predated has not been investigated. Also known as the roof rat, the black rat is highly dependent on buildings, and in Great Britain it has tended to live in dockside warehouses and similar structures. However, in some locations, such as the Shiant Islands and Lundy, it has also occupied rocky habitats and cliffs.

Status

Non-native (naturalised).

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

The black rat was present in Britain by Roman times, with well-stratified remains being recorded in Roman sites in London, York and Wroxeter dated from the 3rd to 5th centuries AD (Rackham, 1979; Armitage et al., 1984). The species was common throughout Great Britain until the introduction of the brown rat — which displaced it — in the early 18th century. Its greater dependency on buildings compared with the brown rat meant that it was more susceptible to rodenticide control; in addition, the switch to containerised storage and the use of grain silos reduced food availability in ports and warehouses (Symes and Yalden, 2002). By 1956, it was restricted to major ports, a few inland towns and some islands (Bentley, 1959); it was eradicated from many of these locations by 1961 (Bentley, 1964), and by 1983 permanent colonies were thought to persist only on the Thames, in Lundy and the Shiant Islands, Hebrides. Since then, there are regular though infrequent records, usually from seaports, where the species is presumably reintroduced with shipping consignments.

The species was eradicated from Lundy, together with the brown rat, in 2006 as part of the Seabird Recovery Project (Lock, 2006) In the Shiant Islands, the population was estimated to be 230-400 individuals in 1996 (McDonald et al., 1997). These were eradicated in 2016 by the Shiant Isles Seabird Recovery Project led by the RSPB, largely in efforts to encourage small-bodied burrowing seabirds such as Manx shearwaters and storm petrels to begin nesting on the islands.

Because of the scarcity of records, and their scattered nature, it is not possible to produce a smoothed distribution map. There were 80 positive hectads between 1960 and 1992, 13 between 2000 and 2009, and one (the Shiants, where the black rat has subsequently been eliminated) between 2010 and 2016.

Results

No estimate was made of population size because of the lack of records.

According to current international guidelines (IUCN, 2001), a species may only be declared extinct in the wild when exhaustive searches fail to find even a single individual. The species therefore cannot formally be considered extinct in Great Britain even though this is likely.

Critique

It is plausible that there are still small populations of this species or occasional individuals present: all commensal animals tend to be under-recorded, and there is also high likelihood of confusion with the brown rat. There has been no systematic exhaustive survey of areas likely to retain the species (such as Tilbury or Cardiff).

Changes through time

Comparison to Harris et al. (1995), Arnold (1993)

The population appears to have been reduced to zero, compared with an estimate of approximately 1300 in the previous report (Harris et al., 1995), comprised of 750 in England and 550 in Scotland. Similarly, the distribution across 80 hectads shown by Arnold (1993) has been reduced to zero.

Table 7.15a Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease			All countries*	
	Data deficient				

* The previous population review by Harris et al. (1995) suggested that the species was absent in Wales. However, records are now available showing that the species did persist at that time.

Drivers of Change

Table 7.15b Drivers of population change for the black rat between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Management (control).	Pest control measures have resulted in widespread eradication.	Bentley (1964); (Lock, 2006)	Negative

Data deficiencies

Table 7.15c Areas where further research is required to improve the reliability of population size estimates for the black rat.

Data deficiencies	**Habitat**	**Details**
Lack of information on the location of any remaining individuals.	Built environment	Exhaustive surveys are required in areas where the species was recorded most recently to identify any remaining animals.

Future prospects

Table 7.15d An assessment of the future prospects for the black rat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	**Status**
Population	Unknown
Range	Unknown
Habitat	Unknown

8 CARNIVORA

8.1 Wildcat *Felis silvestris*

Habitat preferences

Within Britain, the wildcat is now only found in Scotland. The species relies on a mosaic of habitat types, with broadleaved or mixed forest being core (Stahl and Leger, 1992), but its range in Scotland also encompasses a high proportion of coniferous woodland, with young plantations, in particular, being used because of lower deer grazing intensity and high prey densities (Kilshaw, 2011). Open areas, such as marginal farmland and grasslands, also provide hunting opportunities (Easterbee et al., 1991; Silva et al., 2013). They are important habitats in parts of the distribution (e.g., the north east of Scotland), but are avoided elsewhere (Kilshaw, 2011). At a fine scale, habitat fragmentation may be beneficial for wildcats: areas with high percentage cover from coniferous forest are avoided, whereas smaller patches of forest next to areas of grassland are used more frequently. Habitat requirements are, however, unlikely to be a limiting factor for wildcats: the main, and increasing, threat is hybridisation with domestic and feral cats (Littlewood et al., 2014; Kilshaw et al., 2016).

Status

Native.

Conservation Status

- IUCN Red List (GB: CR; England: n/a; Scotland: [CR]; Wales: n/a; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. UK: Bad; England: n/a; Scotland: Bad; Wales: n/a).

Species' distribution

Although recent developments have improved the identification of wildcats, hybrid wildcats and feral cats (see Kitchener et al., 2005; Kilshaw et al., 2010), there is a high probability that some of the presence records used to estimate the species' distribution are feral cat or wildcat hybrids.

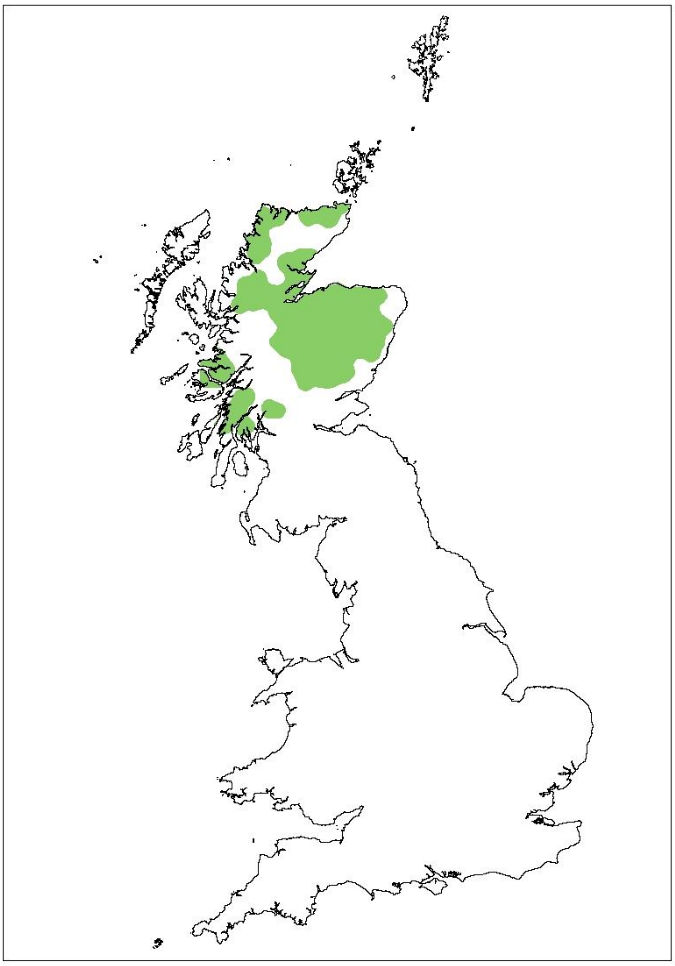


Figure 8.1a Current range of the wildcat in Britain. Range is based on presence data collected between 1995 and 2016. Records verified by local biological recording centres are accepted, together with data from focused surveys (e.g., by Scottish Wildcat Action). Therefore, records defined by both strict and relaxed criteria for pelage characteristics are included. Areas that contain very isolated records may not have been included in the distribution — see Methods, Section 2.5, for details.

Species-specific methods

All of the available density estimates for wildcats were taken over a range of habitats — including heather moorland, coniferous and broadleaved woodland, and rough grazing — that were present within the study areas in differing proportions. Kilshaw et al. (2016) detected wildcats using trail cameras during the winter and found that very few were detected in heather moorland; dwarf shrub heath was, therefore, excluded from the inhabited area. We have assumed equal occupancy of wildcats throughout their known distribution, and applied the same density estimate to broadleaved woodland, coniferous woodland and unimproved grassland, matching the habitats found in the study areas. The percentage occupancy was derived from the observational study by Kilshaw et al. (2016), and the value was applied to all habitats within the study area.

There is considerable difficulty in distinguishing wildcats from domestic cats and hybrids. Whilst classification systems based on pelage characteristics are available (Kitchener et al., 2005), and consistency of application has improved over recent years, extensive interbreeding with domestic cats makes precise identification almost impossible: in a recent survey only 2 of >100 carcasses that appeared to be wildcat on the basis of morphology and pelage were genetically characterised as pure wildcat (Scottish Wildcat Action 2017). It is therefore highly likely that the distribution and population estimates reported here are overly optimistic.

Results

Four papers reporting over-winter estimates of population density were identified by the literature search (Table 8.1a). One of these contained replicate population density values from other papers. Two papers contained assessments of the factors affecting population density, and one — which provides the best data available on wildcat occurrence — provided information on positive and negative sites, and could therefore be used to estimate percentage occupancy (Kilshaw et al., 2016). As all of them considered multiple habitats, the occupancy value is not habitat-specific. Population density estimates and population sizes are shown in Table 8.1a and Table 8.1b, respectively.

Table 8.1a Median density estimates with 95% confidence intervals for wildcats, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	800	0.12	0.02	0.26	Hetherington and Campbell (2012)	2	
Coniferous woodland	4,000				Littlewood et al. (2014)	1	26.7%
						1	
Unimproved grassland	1,500				(Kilshaw, 2015)		

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.1b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.1b with the area of habitat within the species' distribution, and adjusting for occupancy. Small discrepancies between values in the two tables are caused by rounding errors.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
Scotland	6,200	200*	30	430
Britain	**6,200**	**200***	**30**	**430**

* This is highly likely to include some feral cats or hybrid wildcats, and therefore to overestimate population size.

No population size was presented in the 2012 Article 17 Report on wildcat Table 8.1c (Joint Nature Conservation Committee, 2013b). The current review suggests that the geographical range is much smaller than previously estimated (Table 8.1d).

Table 8.1c Article 17 Report on wildcat population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	n/a	n/a
Scotland	n/a	n/a
Wales	n/a	n/a
Britain	**n/a**	**n/a**

Note: maximum and minimum estimates were the same values in the country-level reports.

Table 8.1d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013b).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	0	n/a
Scotland	26,700	n/a
Wales	0	n/a
Britain	**26,700**	**44,130**

Critique

Kitchener et al. (2005) developed a method to distinguish between pure wildcats, hybrids and domestic cats using pelage characteristics. Whilst these characteristics correlate with genetic differentiation (Kilshaw et al., 2010), it is still difficult to classify individuals with certainty, particularly in populations with extensive introgression between wildcats and domestic cats, as is the case throughout the Scottish range (Beaumont et al., 2001; Macdonald et al., 2004a). There is a high probability that some of the density estimates and presence records included in our analysis are from feral cats and hybrid wildcats, particularly since relaxed inclusion criteria tend to be applied to camera-trap records. Population size and distribution are therefore highly likely to be overestimated.

The wildcat makes use of a mosaic of habitat types, so the population density estimates reported in the literature relate to extensive regions, rather than to specific habitats separately. It is therefore not informative to assess the proportion of the population found in each habitat type. All of the surveys that provided population density estimates focused on areas particularly suitable for wildcats, so these densities are likely to be higher than the average for the whole of the species' range. Similarly, the study from which the occupancy data were derived (Kilshaw et al. (2016)) was conducted in areas thought likely to contain wildcats. The density and percentage occupancy values applied in the review are therefore likely to be overly optimistic. A reliability assessment is provided in Table 8.1e.

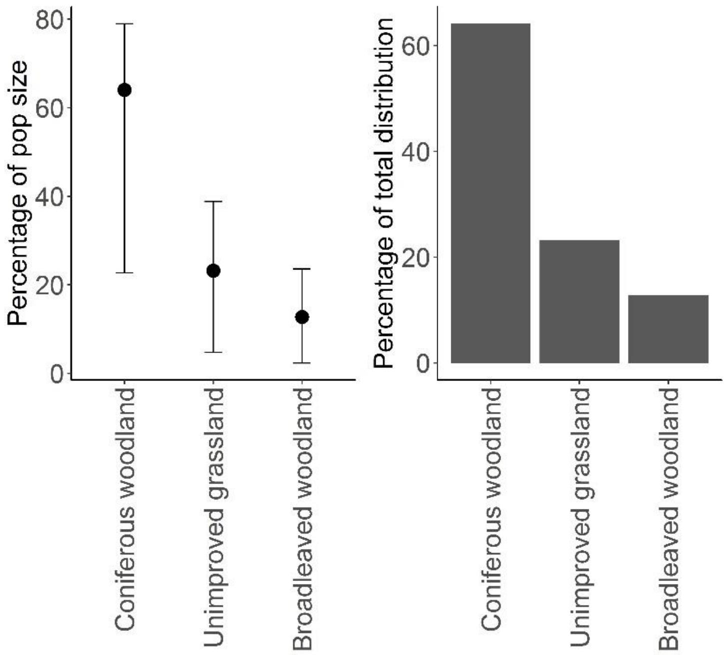


Figure 8.1b Left: The percentage of population estimate derived from each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 8.1e Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All habitats
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	0
	1	10-30 population density estimates	
	2	>30 population density estimates	
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**2**
		Overall reliability score	**2***

* Reliability will be somewhat lower than suggested by this score since uncertainty in the species' identification is high for many of the records.

Changes through time

Comparison to Harris et al. (1995)

Population size was estimated to be 3,500 in Harris et al. (1995). This estimate was produced before the development of the pelage scoring system (Kitchener et al., 2005), however, and so it includes contains hybrid wildcats. This is likely to be the largest source of error in assessing relative trends in wildcat population size. A comparison of methods is, nevertheless, provided for completeness. The population estimate in Harris et al. (1995) was based on two population density estimates ($3km^{-2}$ and $0.8km^{-2}$) assigned to occupied $100km^2$ squares, depending on the frequency of sightings in each square. Both of these density estimates are higher than the median used for the current population size estimate.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (broadleaved woodland and coniferous woodland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a small increase in population size which falls within the confidence limits of the original. Methodological issues relating to changing habitat availability are therefore unlikely to influence the assessment of temporal trends.

Comparison of population sizes between the two reviews shows a sharp decline in population size: the estimate from Harris et al. (1995) is well above the higher confidence interval of the current estimate. This comparison does not, however, take any account of feral hybrids.

Other evidence of changes through time

The population of wildcats in Scotland is widely reported to be under threat of extinction, mostly from hybridisation with feral and domestic cats. This threat has been present for much longer than the last 20 years (Stahl and Leger, 1992) but poses an increasing threat with time (Macdonald et al., 2004a; Kilshaw et al., 2016). A population size of approximately 400 was estimated in the mid-2000s by extrapolating from samples taken from free-living wildcats collected during the 1990s (Macdonald et al., 2004a). A more recent population size of 115-314 wildcats was estimated by Kilshaw (2015), which is in line with the current estimate and represents a decline since 1995 and the mid-2000s. A summary of trends in population size and range is provided in Table 8.1f.

Table 8.1f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease			Scotland	
	Data deficient				

Drivers of change

Table 8.1g Drivers of population change for wildcats between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Hybridisation.	Loss of genetic integrity owing to hybridisation with feral and domestic cats.	Kilshaw et al. (2016) Littlewood et al. (2014)	Negative

Data deficiencies

Table 8.1h Areas where further research is required to improve the reliability of population size estimates for wildcats.

Data deficiencies	Habitat	Details
Population density estimates do not represent within-habitat variability.	All habitats	Density estimates were taken in priority areas for wildcats.
Limited density estimates.	All habitats	Median density is based on four density estimates.
Uncertainty in the degree of hybridisation with domestic cats.	All habitats	This may vary regionally. In addition to being a source of cryptic extinction, the issue may undermine the use of pelage characteristics as a means of identifying the species. Work is needed to assess whether the criteria previously proposed (Kitchener et al., 2005) require revision in the light of increasing hybridisation.

Future prospects

Table 8.1i An assessment of the future prospects for the wildcat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Decline
Habitat	Stable

8.2 Red fox *Vulpes vulpes*

Habitat preferences

The red fox is highly adaptable and versatile. It is most abundant in habitats offering a wide variety of cover and food, but also occurs in montane areas, sand dunes and other open habitats. In urban areas, the species prefers low-density, residential suburbs (Harris and Rayner, 1986), and has territories with greater overlap which drift over time, probably because of the unpredictable nature of human-associated food resources (Doncaster and Macdonald, 1991). In rural areas, the fox often uses unoccupied badger setts as breeding sites (Parrott et al., 2012), and populations can be suppressed by badgers because of competitive exclusion (Macdonald et al., 2004b). Despite long-term attempts to control foxes by hunting, the uncoordinated nature of these interventions means that they appear to have only minor impacts on populations, even where efforts are intensive in the short term (Macdonald et al., 2003; Newsome et al., 2014). In both urban and rural areas, earthworms form a significant proportion of the diet (Doncaster et al., 1990), with the remainder of the diet being comprised of birds, small mammals, rabbits and scavenged items (Macdonald et al., 2015).

The first detailed studies of foxes in urban environments were conducted in the 1970s and 1980s in Oxford and Bristol, and since then the species appears to have colonised increasing numbers of British towns and cities, including Newcastle, Manchester, Brighton, Birmingham and Leeds (Scott et al., 2014). However, occupancy remains patchy, and more systematic surveys are required to assess the relative importance of urban and rural habitats.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [NT]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

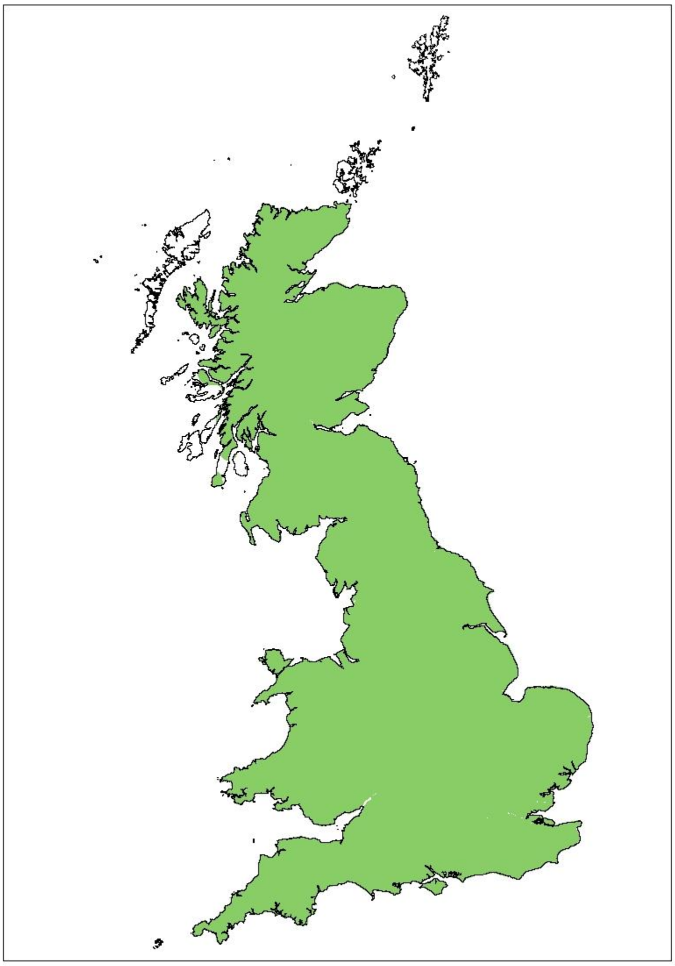


Figure 8.2a Current range of the fox in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

The red fox is a generalist species that includes a range of different habitat types within its home range. Density estimates in the literature therefore often relate to 'urban' or 'rural' habitats generally, rather than to specific categories in the LCM2007. To maximise the number of estimates contributing information to this review, all density estimates were assigned to either 'urban' or 'rural' habitats, where rural habitats encompass all terrestrial habitats other than urban and gardens.

Results

Thirteen papers were identified by the literature search. Five of these contained pre-breeding density estimates, and the remainder reported post-breeding estimates, measures of relative abundance, details of distribution changes, density estimates already included within other studies, or the effects of habitat variables on relative density. Percentage occupancy values were found in two papers, but the values were habitat-specific for just improved grasslands and dwarf shrub heath, and so could not be used in the current analysis. Population density estimates are shown in Table 8.2a, and population size estimates in Table 8.2b.

Table 8.2a Median density estimates with 95% confidence intervals for foxes, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	% Occ†
Urban and gardens	13,800	13.9	1.5	25.8	Soulsbury et al. (2007)	7	n/a
					Scott (*in prep.*)	2	
Rural	210,000	0.79	0.4	1.4	Heydon et al. (2000)	6	n/a
					Webbon et al. (2004)	7	
					Petrovan et al. (2011a)	7	
					Parrott et al. (2012)	7	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.2a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	131,000	255,000	65,200	464,000
Scotland	71,500	74,000	30,100	132,000
Wales	20,800	27,700	9,260	50,000
Britain	**223,000**	**357,000**	**104,000**	**646,000**

Critique

No percentage occupancy data were available; the population size is therefore overestimated. Rural areas form 94% of the species' distribution, but despite this, the high density in urban environments means that similar numbers of foxes are found in urban and rural habitats (Figure 8.2b). The density estimates are based on 9 estimates in urban environments, and 27 in rural environments. Sensitivity analysis was carried out by re-calculating of population size with stepwise removal and replacement of each density estimate for urban areas. The resulting population sizes fell within the confidence limits of the original. Further systematic research to establish density and occupancy is needed in both urban and rural environments: reports of foxes are most likely in places where densities are high, and this may have led to a bias in the literature towards locations with unrepresentative populations.

Webbon et al. (2004) estimated the population of rural foxes to be 225,000, based on faecal count data from 1999-2000. This compares with the current estimate of 168,000 for foxes in rural areas. The current review incorporates the population density estimate from Webbon et al. (2004) ($1km^{-2}$) into the median density estimate. Methodological differences between Webbon et al. (2004) and the current review are likely to explain the divergent population estimates: Webbon et al. (2004) derived density estimates for each British land class (Institute of Terrestrial Ecology), and then extrapolated the findings to all 1km squares within the range, whereas the current review distinguishes only rural and urban habitats.

The main factor that limits fox density is the availability of food, with the highest densities found in rural lowland and urban areas, and much lower densities in the uplands, where food is more scarce (Chadwick et al., 1997). The population size reported here is therefore likely to be an overestimation because of the large areas of upland in Wales and Scotland that are unlikely to be represented accurately by a single median population density applied to all rural areas. A reliability assessment is provided in Table 8.2c.

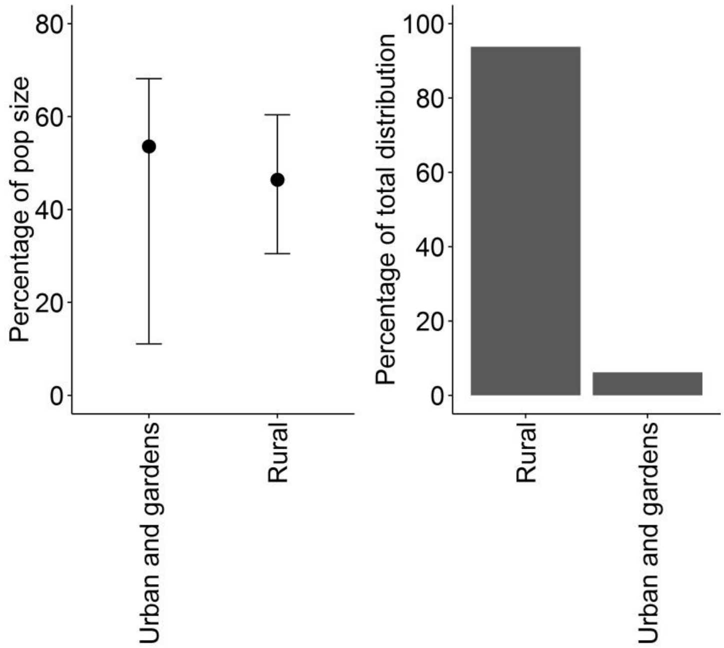


Figure 8.2b Left: The percentage of the total population of foxes accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 8.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Urban and gardens	Rural
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	
	2	Estimates widespread		2
Sample size	0	<10 density estimates		
	1	10-30 density estimates	1	1
	2	>30 density estimates		
Occupancy data available?	0	No	0	0*
	1	Yes		
		Habitat score	**2**	**3**
		Overall reliability score	**2.5**	

*Occupancy data are available for improved grassland and marginal uplands, but were not applicable to the current analysis (Parrott et al., 2012).

Change through time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated that there were 240,000 foxes in Britain, comprising 195,000 in England, 23,000 in Scotland and 22,000 in Wales. These values were derived from separate estimates for 'urban' and 'rural' habitats. The urban estimate was based on a model predicting the density of social groups and a value of 3.4 adults per social group (derived from long-term monitoring of populations in Bristol). Extrapolation was limited to large urban areas (50,000+ residents). The rural estimate was based on social group density (0.04 and 1 social group per km^2, depending on the land class) and 3 adults per social group (Kolb & Hewson (1980); Lloyd (1980)).

The current estimate is also based on population density estimates for urban and rural areas, although density estimates were made directly, rather than via model prediction, and rural areas were not divided into land classes.

Other evidence of changes through time

Relative trends in fox numbers are measured by the BTO as part of the Breeding Bird Survey (BBS), and by the Game and Wildlife Conservancy Trust through the National Gamebag Census (NGB). Between 1995 and 2009, no significant change in relative abundance had been detected by the NGB (-8%), whilst a small but significant increase was reported in the BBS (1%, 95%CI = 1%-21%) (Risely et al., 2010). However, the most recent BBS report indicates a decline of 34% in the numbers of foxes culled (95%CI = 48%-23%) between 1996 and 2014 (Harris et al., 2016). Survey effort is not quantified for the NGB survey, and it is not known whether the observed differences reflect changes in population size. The extent of any decline in the fox population is therefore unclear. A summary of trends in population size and range is shown in Table 8.2d.

Table 8.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	Scotland*	England Wales		

* Increase in the range size may be owing to increased recorder effort.

Drivers of change

Table 8.2e Drivers of population change for foxes between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Disease/pesticides.	Local population suppression by multiple outbreaks of mange.	Soulsbury et al. (2007)	Negative
Habitat quality.	Potential increase in urban populations. The mechanism is unknown, although it may be owing to food availability.	(Scott et al., 2014)	Positive
Management (control).	Localised suppression by control measures. The efficacy may, however, be limited.	Rushton et al. (2006) Heydon et al. (2000)	Negative

Data deficiencies

Table 8.2f Areas where further research is required to improve the reliability of population size estimates for foxes.

Data deficiencies	Habitat	Details
Density estimates do not represent within-habitat variability.	All rural habitats	A more detailed analysis would include density estimates from separate habitats within the rural landscape, or undertaken stratified random surveys across large geographical areas, to enable future estimates to account for regional variations.
No occupancy data.	All habitats	

Future prospects

Table 8.2g An assessment of the future prospects for the red fox, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

8.3 Badger *Meles meles*

Habitat preferences

The badger is one of the most commonly-studied British mammals because of its role in the epidemiology of bovine tuberculosis. It is difficult to infer social group size from the size of main and subsidiary setts or from the number of sett entrances. So, with a few exceptions such as the long-term studies at Wytham Woods in Oxford and Woodchester Park in Gloucestershire, most research has focused on the locations of setts. Badgers in Great Britain are highly social and prey primarily on earthworms. Sites conducive to sett construction (e.g., with sandy soils and gently rolling topography (Macdonald et al., 2004b) where cover is available from broadleaved woodland, scrub or nearby hedgerows (Wilson et al., 1997; Newton-Cross et al., 2007), and where earthworms are readily accessible (e.g., pasture (Newton-Cross et al., 2007)), are therefore preferred. Conversely, sett densities are low in upland and montane regions with heather moorland and acid soils. Nevertheless, lowland heath is used, especially if it is adjacent to favourable foraging habitats such as improved grassland. A recent analysis of a large-scale survey of 1614 1km grid squares in England confirmed that sett densities were highest in pastoral landscapes of south west England and south Wales, and in mixed arable agricultural areas of southern England (Judge et al., 2014). However, while such land-class-based approaches are useful at a national level, at a local scale there is considerable variability in sett density, depending on a combination of environmental factors (Macdonald et al., 1996).

Badgers have also made increasing use of urban areas over the last 25 years. In 1984, a survey of 378 English Local Authorities found that very few reported having urban badgers, and those reports were based on small populations largely confined to the urban-rural interface (Harris, 1984). By 2009, approximately 20% of Natural England licence applications relating to badgers came from urban areas (Delahay et al., 2009). Setts tend to be located in gardens rather than amenity grassland (Huck et al., 2008). Many urban badgers are now deliberately provisioned with food by householders, and they also exploit food waste and forage for natural prey such as earthworms in amenity grassland and gardens. The highest sett densities occur at an intermediate level of human population density (Wright et al., 2000; Schley et al., 2004), balancing the anthropogenic food availability against the probability of disturbance (Huck et al., 2008).

Status

Native.

Conservation status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

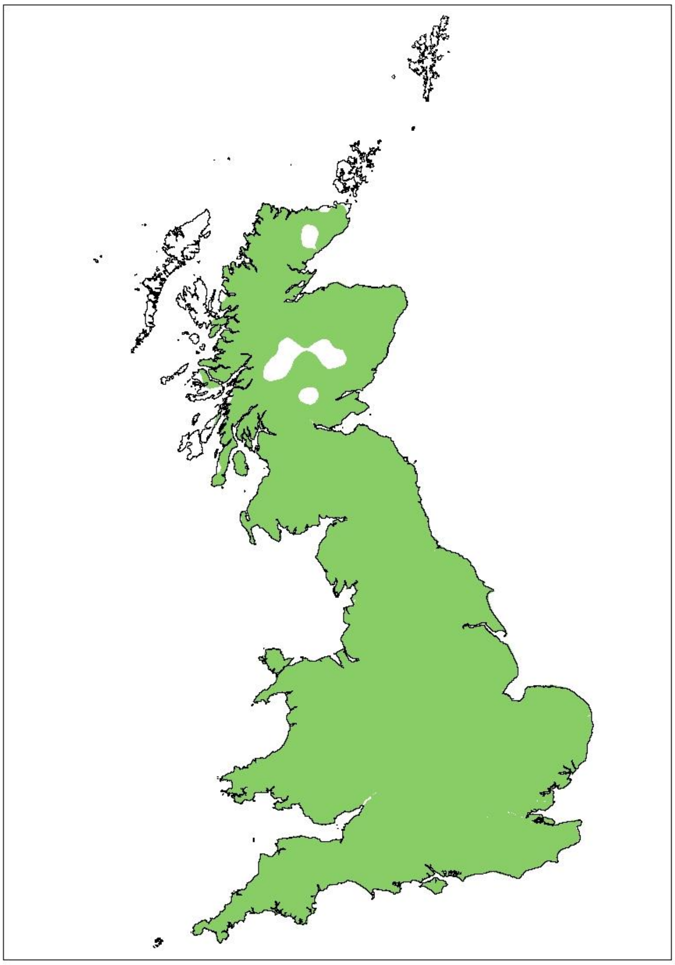


Figure 8.3a Current range of the badger in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

A combination of study types was used, including estimates of individual badger density, sett density and group size. Estimates of sett density were multiplied by the mean value of group size to derive animal densities (although it is recognised that, at the local scales, there is high variability in group size between setts). The mean group size across studies was 4.14. Several of the major studies on badger abundance reported sett densities by landscape character (e.g., 'pastoral habitat' in 'Hunt Countries'). Conversions were therefore made to the nearest broad habitat type. In the case of the paper by Judge et al. (2014), which used land classes, 'marginal upland 6' was taken to equate to 'unimproved grassland', and all 'pastoral' categories were taken to represent 'improved grassland'.

Confidence intervals could not be estimated for unimproved grassland. To permit the computation of confidence intervals for the whole population (across all habitats), the median value for unimproved grassland was substituted for the upper and lower limits.

Because of the way in which predominantly grassland habitats were described in the original research papers, encompassing broad areas which potentially included woodlands, it was unclear whether broadleaved woodland should be included as a separate category, or whether this was already accounted for in the grassland estimates. Therefore, data are presented for both scenarios. The only density data for broadleaved woodland were derived from a population generally considered to be one of unusually high density (Wytham Woods, Oxford (Macdonald and Newman, 2002)). For this habitat only, expert opinion was used in addition to the published literature. Population sizes were then calculated using the general methods outlined at the start of the report.

Results

Thirteen papers were identified during the literature search. Of these, two provided estimates of pre-breeding badger density (Heydon et al., 2000; Parrott et al., 2012), six gave sett density estimates (Micol et al., 1994; Rogers et al., 1997; Macdonald and Newman, 2002; Macdonald et al., 2004b; Huck et al., 2008; Judge et al., 2014), and two estimated group size (Macdonald et al., 1996; Huck et al., 2008). One paper contained post-breeding estimates of social group size, and three contained relative measures of density or temporal trends. Population density estimates are provided in Table 8.3a, and population size estimates in Table 8.3b.

Table 8.3a Median density estimates with 95% confidence intervals for badgers, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	62,500	1.42	1.05	3.58	Heydon et al. (2000) Judge et al. (2014)	2 3	n/a
Urban and gardens	13,700	9.32	4.76	12.91	Huck et al. (2008) Rogers et al. (1997)	4 1 1	n/a
Improved grassland	71,300	4.68	3.91	6.42	Heydon et al. (2000) Judge et al. (2014) Micol et al. (1994) Parrott et al. (2012)	1 2 6 5	n/a
Unimproved grassland	11,237	1.25	n/a	n/a	Judge et al. (2014)	1	n/a
Broadleaved woodland	13,052	15.0	8	38	Macdonald and Newman (2002) expert opinion	2	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.3b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.3a with the area of habitat within the species' distribution.

		Including broadleaved woodland			Excluding broadleaved woodland		
Country	**Area of suitable habitat (km^2)**	**Popn. size**	**-95%CI***	**+95%CI***	**Popn. size**	**-95%CI***	**+95%CI***
England	120,000	519,000	350,000	961,000	384,000	259,000	711,000
Scotland	34,500	156,000	115,000	267,000	115,000	85,000	198,000
Wales	17,400	85,000	63,000	140,000	62,900	47,000	104,000
Britain	**172,000**	**760,000**	**528,000**	**1,370,000**	**562,000**	**391,000**	**1,014,000**

* No confidence intervals were available for the density estimate in unimproved grassland. Therefore, uncertainty in this habitat estimate is not incorporated in the confidence intervals shown here. The contribution of this habitat to the total population estimate is small, so the confidence limits shown above are likely to be reasonable.

Critique

No percentage occupancy data were available; the population size for this species is therefore overestimated. When considering all habitat types (including broadleaved woodland), most of the population size estimate for badgers is derived from improved grassland (61%; Figure 8.3b), and more than 44% of the species' range consists of this habitat type. The population density used for improved grassland was based on 14 esimates from four studies. Broadleaved woodland contributed 26% of the population estimate, but the densities used are extremely uncertain, relying on an unusual population in a single location (at two time points) and expert opinion. National surveys of broadleaved woodland would therefore substantially improve the estimate.

Judge et al. (2017) recently estimated a population size of 424,000 in England and 61,000 in Wales (485,000 total for England and Wales; 95%CI = 391,000-581,000). They used a molecular estimation of social group size (mean 6.7, SE 0.63) using data from 120 main setts, and combined it with land class-specific estimates of sett density and an average social group size of 6.74 (SE 0.63). The current estimate for England and Wales is somewhat higher than the estimate of Judge et al. (2017) when broadleaved woodland is included, but is very similar when it is excluded.

Judge et al. (2017) did not present data for Scotland, but assuming a social group size of 4.14 four badgers (as suggested by this review) and multiplying this by the estimated of number of main setts (7,300-11,200 (Rainey et al., 2009)) gives a population estimate of

30,000-45,000. This is substantially lower than either figure provided in the current review. It is unclear which estimate is more reliable, but it is reasonable to assume that the method used in this review overestimated the Scottish population because lower densities of badgers than the national median would be expected at high altitudes and on acid soils.

Several of the density estimates used in the current analysis relied on mean social group size derived from just two studies (Macdonald et al., 2004b; Huck et al., 2008). Social group size is, however, highly variable (Wilson, 2003). In addition, sett density may be a poor predictor of population density (Judge et al., 2014), and sett densities can vary widely at local scales (Macdonald et al., 1996). The application of single values across large landscape areas may therefore be inaccurate. These constraints affect both the current review and the recent estimates based on molecular approaches (Judge et al., 2014; Judge et al., 2017). A reliability assessment is provided in Table 8.3c.

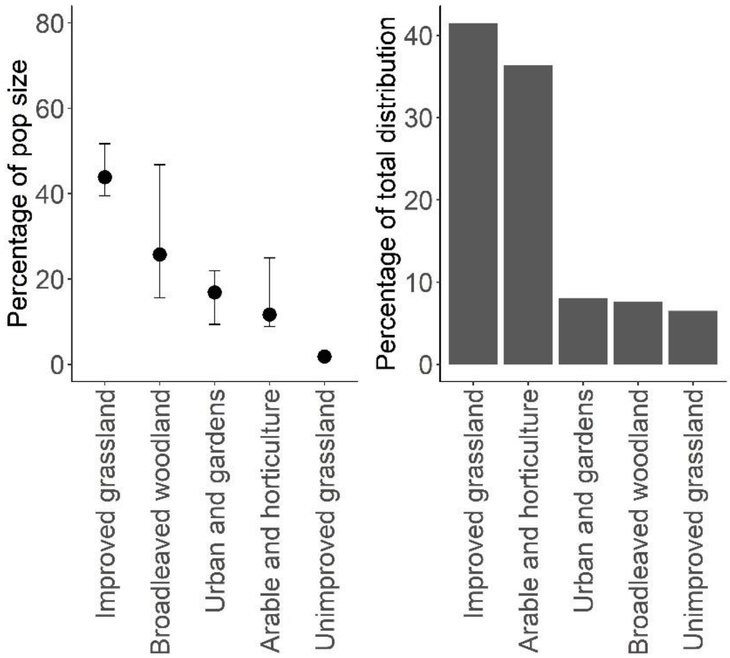


Figure 8.3b Left: The percentage of the total population of badgers accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 8.3c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat		
			Improved grassland	Arable and horticulture	Broadleaved woodland
Location of study sites	0	Estimates from one location			1
	1	Estimates restricted	1	1	
	2	Estimates widespread			
Sample size	0	<10 population density estimates		0	
	1	10-30 population density estimates	1		1
	2	>30 population density estimates			
Occupancy data available?	0	No	0	0	0
	1	Yes			
		Habitat score	**2**	**1**	**1**
		Overall reliability score	**1.3**		
		Revised reliability score*	**4**		

* The reliability score was revised because alternative population estimates based on molecular approaches are available. These corroborate the England and Wales estimate, but are somewhat lower than those obtained by this review for Scotland.

Changes through time

Comparison to Harris et al. (1995)

Population size estimates in (1995) were based on a mean social group size of 6 and a total of 41,894 main setts, giving a population size of 250,000 in Britain. A lower social group size was applied in the current analysis, based on the evidence provided by the literature review, implying that sett density would have had to increase in order to achieve the inferred increased population size. This change in sett density is difficult to verify, however, as the current population size was derived from a combination of sett density and social group size data that were then extrapolated across the geographical range. Judge et al. (2017) have recently estimated social group size as 6.7, based on a genetic assessment of 120 main setts.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (arable land, broadleaved woodland, improved grassland and urban), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). Adjusting the results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a lower estimate that falls just outside the current confidence limits (401,000; 95% CI = 324,000-478,000), implying a smaller, but nevertheless very substantial, population increase (160%).

Population size appears to have increased since 1995, but concerns over the use of social group sizes to infer population size, and differences in the methods used, mean that population size estimates from both time periods may be inaccurate. Comparisons should therefore be drawn with caution.

Other evidence of changes through time

Three nationwide badger surveys were conducted between 1994 and 2013. These surveys suggest a 77% population increase between 1985 and 1997 (Wilson et al., 1997), and a 103% increase in sett density from 1985 to 2010 (Wilson et al., 1997; Judge et al., 2013). Although they provide the best trend data available for badgers, these figures should be regarded with caution, as differences in methodology between surveys may have resulted in an increase in survey effort in the latter years (Battersby and Greenwood, 2004). A summary of trends in population size and range is provided in Table 8.3d.

Table 8.3d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**		All countries*		
	Stable				
	Decrease				
	Data deficient				

* Differences in methodology between surveys mean that trends are uncertain.

Drivers of change

Table 8.3e Drivers of population change for badgers between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Recovery from persecution.	Implementation of the Protection of Badgers Act (1992).		Positive
Management (control).	Legal culls in an attempt to reduce TB transmission resulted in 11,000 badgers killed in 1998-2005, and 4,000 in 2013-2015, in the south west of England. Culls are ongoing, and the number of individuals affected is unclear.		Negative
Anthropogenic impacts.	An estimated 50,000 badgers are killed by vehicle collisions annually.	Harris et al. (1992) Clarke et al. (1998)	Negative

Data deficiencies

Table 8.3f Areas where further research is required to improve the reliability of population size estimates for badgers.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Dwarf shrub heath	Badgers are likely to be found in a variety of habitats, not limited to those for which current density estimates are available.
Density estimates do not represent within-habitat variability.	All habitats	Social group size is highly variable, but one average value was used for the current analysis.
Managed populations.	All habitats	Legal and illegal culling causes localised population suppression.
Limited density estimates for key habitat.	Broadleaved woodland Unimproved grassland Arable land Urban and gardens	Relatively few density estimates are available, particularly for broadleaved woodland which is an important habitat for setts.
No occupancy data.	All habitats	

Future prospects

Table 8.3g An assessment of the future prospects for the badger, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable*
Range	Stable
Habitat	Stable

* Except in intensive cull areas.

8.4 **Otter** *Lutra lutra*

Habitat preferences

The otter is found in freshwater habitats from coast to upland, and is capable of long overland journeys between watersheds. It also exploits marine environments, particularly rocky coasts where there is high food supply, but it is dependent on the availability of fresh water for cleaning salt from its fur (Kruuk et al., 1989). Adult females are highly territorial and defend large home ranges that are overlapped by one or more males. The size of the home range varies from 4km to 50km in length, and depends on the availability of prey and denning resources, as well as on the spatial configuration of aquatic habitats. It is challenging to estimate population densities accurately because the otter is difficult to observe directly, its holts are difficult to find, and spraint abundance (faecal markings) has complex relationships with the numbers of individuals, varying according to sex, season and other factors (Kruuk and Conroy, 1987; Chanin, 2003). Although some new insights are being brought by genetic analysis of non-invasive samples, otter faecal DNA amplifies very poorly compared with many other species (Dallas et al., 2003; O'Neill et al., 2013).

Dependence on water, and aquatic prey, makes the otter vulnerable to river management and to agricultural pollution. Persistent organic pollutants are likely to have caused the historic declines in otter populations. The species has recolonised most of its former range in Great Britain following the banning of these compounds (Chanin, 2003; Kean et al., 2013).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [VU]; Wales: [VU]; Global: NT.).
- National Conservation Status (Article 17 overall assessment 2013. UK: Favourable; England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A distribution map is presented in Figure 8.4a. The gap in the species' distribution in Scotland is likely to represent areas lacking survey effort, rather than true absences. Expert consultation suggests that gaps in the south east of England are more likely to represent true gaps, although the species is beginning to recolonise Kent. Further survey effort is, however, recommended.

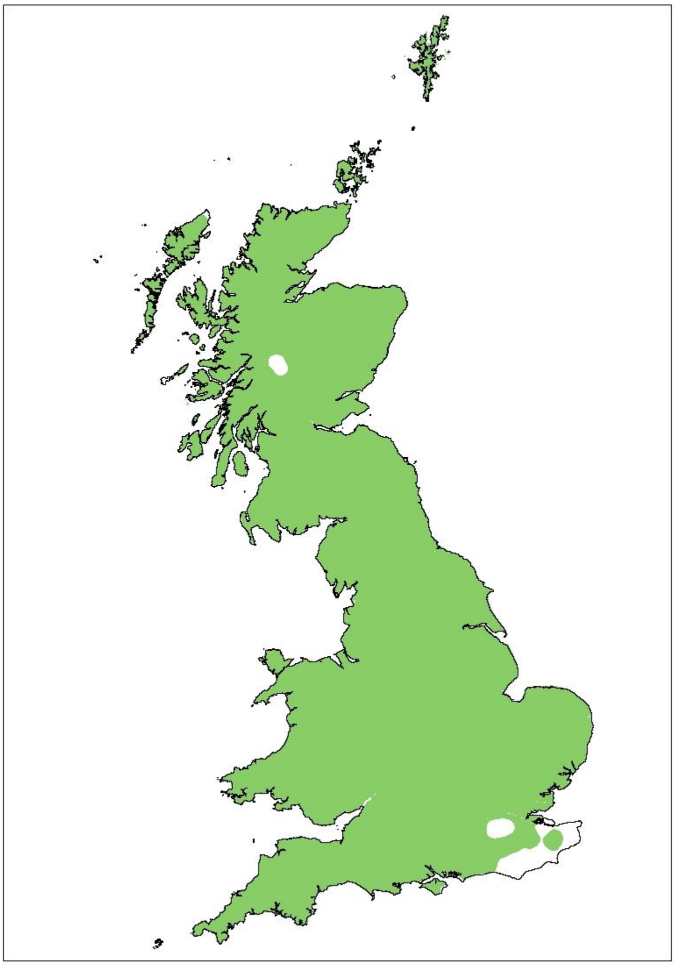


Figure 8.4a Current range of the otter in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

The length of total riparian habitat within the geographical range of the otter in each country was derived by multiplying the data on riparian lengths given in Table 4 of Harris et al. (1995) by the proportion of each country included in the species' distribution. The length of potentially suitable coastline was derived from the report by Jefferies et al. (2003) (Table

10.3 for England and Wales; Table 10.6 for Scotland). These values excluded areas unlikely to be included within the home ranges of otters (e.g., long lengths of sheer cliffs), whereas all riparian habitat was included. Population size was adjusted using the most recent occupancy values for each country. For Scotland, the mean population density values for coastlines in mainland Scotland, the Inner Hebrides, Shetland and Orkney were taken from Table 10.6 of Jefferies et al. (2003). No population density estimates or occupancy values were available for coastlines in England and Wales, so the values for inland populations were applied. This method will provide a conservative estimate of the number of coastal otters in England and Wales, but was judged preferable to applying Scottish coastal values, which are likely to be much higher than those found in England and Wales.

Results

Twelve papers were identified by the literature search. Of these, one reported pre-breeding population density (Jefferies et al., 2003; originally surveyed by Green and Green (1987)), three contained occupancy values (Crawford, 2010; Findlay et al., 2015; Strachan, 2015), and the remainder reported small-scale surveys (no density estimates) and distribution surveys.

The most recent occupancy values for each country were obtained from surveys in 2009-2010 for England (Crawford, 2010), 2009-2010 for Wales (Strachan, 2015) and 2011-2012 for Scotland (Findlay et al., 2015). These percentage occupancy values and population density estimates are shown in Table 8.4a, and population size estimates in Table 8.4b.

Table 8.4a Median density estimates, per unit length of habitat, with 95% confidence intervals for otters, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Country	Density (km^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Riparian & coastal	England	0.037	-	-	Green and Green (1987)	1	56%
Riparian	Scotland	0.042	-	-	Green and Green (1987)	1	80%
Coastal	Scotland	0.453	0.258	0.629	Green and Green (1987)	4	
Riparian & coastal	Wales	0.037	-	-	Green and Green (1987)	1	89.9%

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.4b Length of all riparian and potentially suitable coastal habitat (not adjusted for occupancy) within the species' range, and total population size estimates. Values were obtained by multiplying population density from Table 8.4a with the length of habitat within the species' distribution, and adjusting for occupancy. It was not possible to calculate confidence intervals as they were not available for all density estimates in Table 8.4a.

Country	Length of habitat (km)	Population size	-95%CI	+95%CI
England	141,000	[2,900]	-	-
Scotland	151,000	[7,100]	-	-
Wales	29,000	[1,000]	-	-
Britain	**321,000**	**[11,000]**	**-**	**-**

The Article 17 Report on otter population size 2007-2012 is shown in Table 8.4c (Joint Nature Conservation Committee, 2013b) and is similar to that computed in the current review: both reports are based on largely the same underlying information. The geographical ranges are also similar (Table 8.4d).

Table 8.4c Article 17 Report on otter population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	2,790	2,790
Scotland	8,000	8,000
Wales	930	930
Britain	**11,720**	**11,720**

Note: maximum and minimum estimates were the same values in the country-level reports.

Table 8.4d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	125,700	n/a
Scotland	76,500	76,430
Wales	20,600	n/a
Britain	**222,800**	**229,760**

* To permit comparison with the Article 17 Report, this is the total area encompassed within the alpha shape. Lengths (km) of suitable habitat (riparian and coastal) are shown in Table 8.4b.

Critique

Population size for each country is based on a single country-specific population density estimate for riparian habitats (and coastlines in England and Wales), and four population density estimates for coastlines in Scotland. These density estimates are applied to all occupied riparian habitats and coastlines, so no account is taken of habitat heterogeneity. This is a particular problem for coastlines in England and Wales, as the application of riparian density estimates to coastal areas is highly likely to be inaccurate.

Percentage occupancy for Scotland was taken from Findlay et al. (2015). Field conditions during the survey were poor, with high rainfall, which may have increased the chance of obtaining false negatives. Percentage occupancy may, therefore, be higher than estimated in Scotland.

Table 8.4e Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Riparian	Coastal
Location of study sites	0	Estimates from one location	0	
	1	Estimates restricted		
	2	Estimates widespread		
Sample size	0	<10 density estimates	0	
	1	10-30 density estimates		
	2	>30 density estimates		
Occupancy data available?	0	No		
	1	Yes	1	
		Habitat score	**1**	
		Overall reliability score	**1**	

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The total pre-breeding population of 7,350 individuals was estimated by Harris et al. (1995) for the mid-1980s, comprised of 350 in England, 6600 in Scotland (3,600 on the mainland and 3,000 on the islands), and 400 in Wales. The method of calculating total population size was based on calculations from D. J. Jefferies, which were later published in Jefferies et al. (2003), and were used as the basis for the 2013 Article 17 Report (Joint Nature Conservation Committee, 2013a).

The current review employs the same density estimates as Harris et al. (1995). The 49% increase in population size is therefore the consequence changes in occupancy and geographical range compared with Arnold (1993). The geographical range (surface area) is very similar to the values given in the Article 17 Report (Joint Nature Conservation Committee, 2013a).

Other evidence of changes through time

A series of national surveys have been conducted to detect the rate of change in the otter's area of occurrence. These surveys were not, however, designed to provide information on population trends. There has been an increase in the number of occupied 10km squares in all three countries, with an increase from 5.8% in 1977-1979 to 58.8% in 2009-2010 in England (Crawford, 2010); from 38% in 1977-1989 to 72% in 2002-2003 in Wales (Strachan, 2015); and from 57% in 1977-1979 to 80% in 2003-2004 in Scotland (Findlay et al., 2015). A summary of trends in population size and range is shown in Table 8.4f.

Table 8.4f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	All countries			
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 8.4g Drivers of population change for otters between 1995 and the present.

Driver	**Mechanism**	**Source**	**Direction of effect**
Reduction in organochlorine pesticide pollution.	Substances banned in the UK.	Kean et al. (2013)	Positive

Data deficiencies

Table 8.4h Areas where further research is required to improve the reliability of population size estimates for otters.

Data deficiencies	Habitat	Details
Limited density estimates for key habitat.	Riparian	One estimate for riparian habitats.
Density estimates do not represent within-habitat variability.	Riparian	Limited density estimates make it impossible to calculate confidence limits.
Density estimates are more than 10 years old.	Riparian	Density estimate is from Jefferies et al. (2003).

Future prospects

Table 8.4i An assessment of the future prospects for the otter, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

8.5 Pine marten *Martes martes*

Habitat preferences

The pine marten in western Europe is not dependent on closed-canopy woodland, unlike eastern European populations (Pereboom et al., 2008; Mergey et al., 2011), and it occurs in areas with as little as 4% forest cover (Balharry, 1993). In Scotland, the pine marten is adapted to a landscape with low levels of forest cover; the highest recorded population densities occur in areas with intermediate levels of forest fragmentation (Caryl et al., 2012; Kubasiewicz, 2014). It is also recorded in areas with very low levels of forest cover in the north west Highlands (Balharry, 1993), and in non-wooded habitats such as upland montane areas, semi-natural grassland, and heathland in the Cairngorms (Croose et al., 2013; Moll et al., 2016). High pine marten densities are also recorded in the Irish midlands ($3.13 km^{-2}$), where woodland is particularly sparse and fragmented (Sheehy, 2013). In such regions, home ranges are larger to incorporate the resources required for resting and foraging (Balharry, 1993). The species is also adaptable, and may be able to supplement the resources provided by woodlands, such as denning sites and foraging opportunities, with features found in other habitat types (Caryl et al., 2012).

The dietary composition of the pine marten in Scotland varies seasonally according to the availability of different food sources, including small mammals, carrion, berries and insects (Caryl, 2008). There is a strong preference for the field vole as a primary prey item — in contrast to the preference for bank voles displayed by eastern European populations (Caryl, 2008). This preference is reflected in the incorporation of scrub and tussocky grassland into the home range (Pereboom et al., 2008; Caryl et al., 2012). Milder winters and higher availability of rodents has been linked to higher densities of pine martens in mainland Europe (Zalewski and Jedrzejewski, 2006). These factors may affect population density more than the availability of woodland habitat.

Pine martens were once prevalent throughout mainland Britain. However, by the late 19th century, only a few populations in the north west of Scotland survived (Langley and Yalden, 1977; Ritchie, 2015). Some recovery of suitable habitat, followed by legal protection (Wildlife and Countryside Act (1981); protection for the species was enacted in 1988), has led to a partial recolonisation of the Scottish range over the last few decades (Croose et al., 2013; Croose et al., 2014).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [CR]; Scotland: [LC]; Wales: [CR]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. UK: Favourable; England: Bad; Scotland: Favourable; Wales: Bad).

Species' distribution

All verified records are included in the distribution map (Figure 8.5a, left panel). Some highlighted areas, particularly those in England and Wales, represent very occasional records rather than established populations. A map produced by the Vincent Wildlife Trust (Figure 8.5a, right panel) is provided for comparison, with established populations being shown in dark green. This map is largely the result of two recent expansion zone surveys in Scotland (Croose et al., 2013; Croose et al., 2014), and monitoring following population reinforcement in Wales.

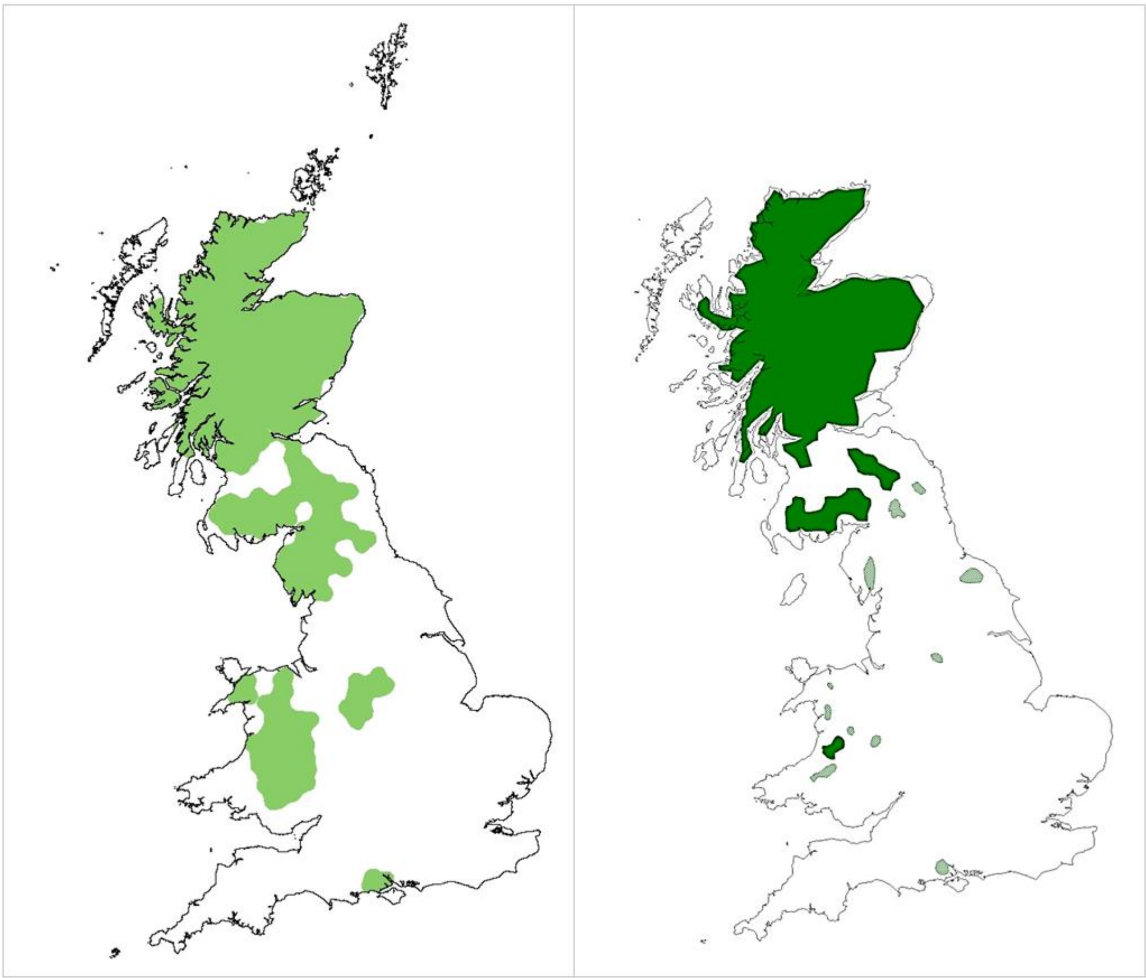


Figure 8.5a Left: Current range of the pine marten in Britain based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. Right: Map showing established populations (dark green) and occasional records (light green) up to 2016 (Vincent Wildlife Trust (Croose et al., 2013; Croose et al., 2014, VWT pers. comm.). There are also verified records for Mull.

Species-specific methods

Estimates of pine marten density and home range size are all taken from sites in Scotland dominated by coniferous forest, including varying degrees of plantation and semi-natural habitat. Sites also contain some broadleaved woodland. All density estimates were applied to both coniferous and broadleaved woodlands. The population estimate therefore only represents individuals associated with woodland, however expert opinion suggests that in Scotland this will be most of the population (Johnny Birks, *pers. comm.*).

Several papers found during the literature search contained estimates of pine marten home range size as opposed to density (Balharry, 1993; Bright and Smithson, 1997; Halliwell, 1997; Caryl et al., 2012). Home range size has previously been used as a proxy for density, with ranges being assumed to be contiguous and without overlap within each sex. This approach was also used for the purpose of the current review.

The population size estimate for Wales is based on the number of animals translocated to Wales from Scotland during the 2015 and 2016 Pine Marten Recovery Project (Vincent Wildlife Trust, *pers. comm.*). Extensive research by the Vincent Wildlife Trust suggests that records in England do not indicate an established population. Therefore, no estimate has been made for England.

Results

Two papers containing population density estimates were identified by the literature search. These reported estimates made between September and November (Kubasiewicz, 2014; Croose et al., 2015). A further four papers reported pine marten home range sizes, based on studies of at least one year (the specific timings of individual capture and tracking were not specified) (Balharry, 1993; Bright and Smithson, 1997; Halliwell, 1997; Caryl et al., 2012). Two papers contained information relevant to occupancy by reporting the percentage of surveyed hectads found to contain pine marten scats in east and central Scotland (25%; Croose et al., 2013) and southern Scotland (4%; (Croose et al., 2014). However, the surveys were conducted with relatively low sampling frequency, and an unusually high proportion of DNA extracted from scats could not be identified to species (48%; Croose et al., 2013). The surveys were also conducted at the edge of the species' range. It is therefore concluded that these reported occupancy rates are unlikely to be representative.

Pine marten kits typically emerge from the natal den in late June and disperse from their mother's territory between September and mid-November (Harris and Yalden, 2008). The calculated population sizes therefore represent means for the year, with some bias towards the post-breeding population. Habitat-specific density estimates per habitat are shown in Table 8.5a, and total population size estimates in Table 8.5b.

Table 8.5a Median density estimates with 95% confidence intervals for pine martens, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ †
Coniferous woodland	9,500	0.3	0.2	0.8	Balharry (1993)	2	n/a
Broadleaved woodland	2,500				Bright and Smithson (1997)	2	
					Halliwell (1997)	1	
					Bright and Smithson (2001)	1	
					Caryl et al. (2012)	1	
					Kubasiewicz (2014)	3	
					(Croose et al., 2015)	1	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.5b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.5b with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size*	-95%CI	+95%CI
Scotland	10,800	3,700	1,600	8,900
Wales	1,300	39	-	-
Britain	**12,100**	**3,700**	**1,600**	**8,900**

* This represents the population in woodlands only. For Wales, the population size is the number of individuals released and monitored during 2015 and 2016 by the Vincent Wildlife Trust (Jenny McPherson, *pers. comm.*).

The Article 17 Report on pine marten population size 2007-2012 is shown in Table 8.5c (Joint Nature Conservation Committee, 2013b). The values fall within the 95% confidence limits of the estimate computed in this review. Geographical ranges sizes, however, are similar in the two reports (Table 8.5d).

Table 8.5c Article 17 Report on pine marten population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	Unknown	Unknown
Scotland	2,237	4,461
Wales	Unknown	Unknown
Britain	**3,500**	**4,461**

Table 8.5d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013b).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	12,400	n/a
Scotland	61,000	n/a
Wales	9,500	n/a
Britain	**82,900**	**70,990**

Critique

Our analysis is restricted to woodland habitats, and this will have tended to underestimate the population size. However, this error is considered unlikely to be serious because most of pine martens in Scotland are thought to incorporate woodland into their home range (Johnny Birks, *pers. comm.*). Potentially more serious is the lack of occupancy data, and the consequent assumption that pine martens are present in all woodlands within the geographical range: the population size is therefore likely to be overestimated.

The density estimates found in the literature (n=11) were applied to all woodlands within the species' range. Although most woodland within the species' distribution is coniferous (75%), deriving separate density estimates for coniferous and broadleaved woodland would be unlikely to improve the estimate materially, because pine martens have large home ranges and use a matrix of different habitats.

The highest densities of pine martens in Scotland were recorded in areas with 20%-35% forest cover (see Kubasiewicz, 2014), but our calculations do not take the importance of local habitat composition into account. Population sizes in areas of high forest cover will therefore tend to be overestimated, and the converse will be true in areas of intermediate cover. Given that average forest cover in Scotland is 17%, these errors are expected to lead to an underestimate of population size. Further surveys to clarify these relationships are recommended, as conclusions are currently based on relatively low sample sizes.

Experts consulted for this report suggested that the population size is most likely to be closer to the upper confidence limit of 8,900 individuals (Laura Kubasiewicz, *pers. obs*.). A reliability assessment is shown in Table 8.5e.

Table 8.5e Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Woodlands
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	
	2	Estimates widespread	2
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	1
	2	>30 population density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**3**
		Overall reliability score	**2***

* The overall reliability score is reduced to 2 because data are only available for woodlands, but the highest densities are found in landscapes with a high proportion of other habitat types, which suggests that these are likely to contribute substantially to the overall population.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The population size given by Harris et al. (1995) was 3,650, comprised of <100 in England, 3,500 in Scotland and <50 in Wales. The estimate for Scotland was based on home range size as a proxy for density: the total area of woodland within the species' distribution was divided by the area of woodland (1.26km^2) found within an average pine marten territory of 4-10km^2. Outside the core range in the Highlands, percentage occupancy of 50% was assumed, although no empirical data were available (Balharry, 1993). The current analysis derived similar population estimates to Harris et al. (1995), but it is unclear whether the population is stable because a different methodology was used: densities measured in woodlands with varying degrees of fragmentation were multiplied by the area of woodland within the range (Scotland only). Although these density estimates are likely to be too high because non-woodland habitats in the home ranges were excluded, the calculations were only applied to woodland. This error is unlikely to be serious provided that woodland forms a core part of the home range of most pine martens, and that detectability is good. However, if much of the population lives independently of woodland, or pine martens have low detectability in woodland, then the estimates will be too low. Further study is needed to distinguish between these two possibilities.

Nationally, there are changes between the two reviews in the estimated availability of key habitats (broadleaved woodland and coniferous woodland), generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — generates a population size that still falls within the confidence limits of the original. Comparisons between the two reviews are therefore unlikely to be affected materially by these methodological issues.

The geographical range estimate for Britain is similar to that reported by the last Article 17 Report (Joint Nature Conservation Committee, 2013a), but is considerably larger than that reported by Arnold (1993). The area of occupancy for England does not represent established populations.

Other evidence of changes through time

Pine martens have continued to increase their range in Scotland in the last 20 years (figure 8.3a; Croose et al., 2013; Croose et al., 2014), and the median density estimate used for the current analysis is larger than both of the estimates provided by Harris et al. (1995). The population is, therefore, highly likely to have increased. An overview of trends in population size and range is provided in Table 8.5f.

Table 8.5f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	Scotland* Wales**			
	Stable				
	Decrease				
	Data deficient				

* Population size increase is currently based on expert opinion and inferred from a significant increase in range size.
** Increase in population and range size is owing to a reinforcement programme.

Drivers of change

Table 8.5g Drivers of population change for pine martens between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Recovery from persecution.	Legal protection under the Wildlife and Countryside Act (1981) has reduced persecution. Populations have continued to recover over the last 20 years.	Croose et al. (2013)	Positive
Habitat availability.	There has been an increase in forest cover from 12% in 1982 to 17% in 2007. Increased habitat availability has enabled range expansion over the last 20 years.	Scottish Natural Heritage (2010) Croose et al. (2013)	Positive
Reinforcement.	39 pine martens have been released into selected sites in Wales as part of a reinforcement project led by the Vincent Wildlife Trust.	Vincent Wildlife Trust (*pers. comm.*)	Positive

Data deficiencies

Table 8.5h Areas where further research is required to improve the reliability of population size estimates for pine martens.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Non-wooded habitats	All density estimates are currently woodland-specific.
No occupancy data.		

Future prospects

Following a feasibility study for the reinforcing of small populations of pine martens in England and Wales through translocation of individuals from Scotland, habitat suitability was assessed, and five regions in Wales and one on the English/Welsh border were identified as potential release sites (MacPherson et al., 2014). Thirty-nine pine martens were translocated successfully to central Wales during 2015 and 2016. The locations selected for release offered large areas of suitable habitat, and had low risks of mortality from road traffic incidents (MacPherson et al., 2014).The prospects for achieving an established population in Wales are therefore positive.

Table 8.5i An assessment of the future prospects for the pine marten, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

8.6 Stoat *Mustela erminea*

Habitat preferences

The stoat is found in most habitats in Britain, and at any altitude, provided that sufficient cover and prey are available. Like other mustelids, there is territory defence against intruders of the same sex (Erlinge, 1977), and the smaller home ranges of females tend to be overlapped by one or more males (Powell, 1979). The species tends to avoid open spaces by travelling along hedgerows, ditches and stone walls. The stoat is a specialised predator of small and medium-sized mammals, and rabbits are a key prey item, particularly for males, forming more than 50% of their diet throughout the year (McDonald et al., 2000). Foraging is therefore concentrated on rabbit warrens, early successional communities favoured by field voles, and brush timber piles that might harbour small mammals (see Harris and Yalden, 2008). Populations appear to fluctuate in response to food supply. They showed marked declines (as measured by the National Gamebag Census) in the 1950s and 1960s, following myxomatosis epidemics (Sumption and Flowerdew, 1985).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [NT]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

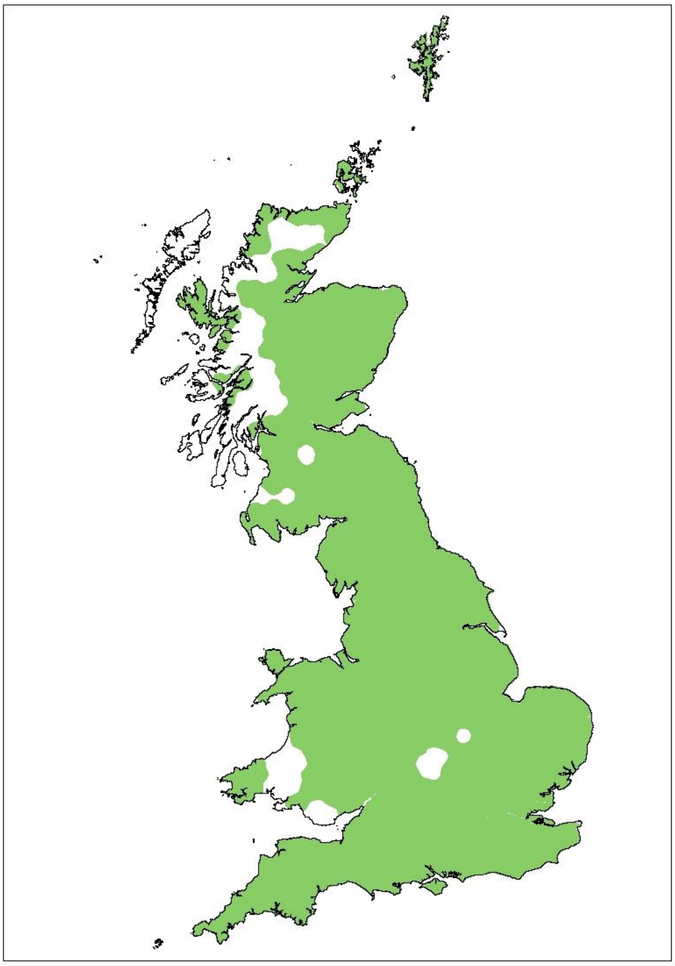


Figure 8.6a Current range of the stoat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

No papers identified by the literature search for stoats reported pre-breeding population density estimates, trends, occupancy, or the effect of environmental variables on relative density. The population density estimates (Table 8.6a) are therefore taken from Harris et al. (1995), but these were not based on any published data for Britain. Population size estimates are shown in Table 8.6b.

Table 8.6a Median density estimates with 95% confidence intervals for stoats, calculated using data obtained from Harris et al. (1995).

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Arable and horticulture	61,200	2	-	-	Harris et al. (1995)	1	n/a
Bog	6,700	2	-	-	Harris et al. (1995)	1	n/a
Broadleaved woodland	12,300	6	-	-	Harris et al. (1995)	1	n/a
Coniferous woodland	11,800	6	-	-	Harris et al. (1995)	1	n/a
Dwarf shrub heath	15,100	2	-	-	Harris et al. (1995)	1	n/a
Improved grassland	65,200	1	-	-	Harris et al. (1995)	1	n/a
Unimproved grassland	10,300	6	-	-	Harris et al. (1995)	1	n/a
Sand dunes	200	2	-	-	Harris et al. (1995)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.6b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.5a with the area of habitat within the species' distribution. It was not possible to calculate confidence intervals, as none were available for density estimates from Harris et al. (1995).

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	116,000	[260,000]	-	-
Scotland	51,600	[140,000]	-	-
Wales	15,600	[37,600]	-	-
Britain	**183,000**	**[438,000]**	**-**	**-**

Critique

No percentage occupancy data were available; the population size is therefore overestimated. The habitat contributing the greatest proportion of the population estimate is arable land (28%), with a further 62% split between broadleaved woodland, coniferous woodland, improved and unimproved grassland (Figure 8.6b). Two of these habitats — improved grassland and arable land — contribute a high proportion of the population size because of the large areas present within the species' distribution (69%). In contrast, the high population densities in the other three habitats — unimproved grassland, coniferous and broadleaved woodland — explain their contribution to the population estimate (Table 8.6b).

Although stoats are present in arable land, they are more likely to use field boundaries and hedgerows. Home ranges in the Swiss Jura mountains tend to be linear and follow boundary features (Debrot and Mermod, 1983). Density estimates from Harris et al. (1995) were based on the authors' expert opinion, rather than empirical data, and it is unclear how this behaviour was taken into account for the density estimates for arable land. In this review, lengths of hedgerows were not included as a separate habitat category in order to avoid double counting.

To assess reliability, we have considered the population density estimates from Harris et al. (1995) to be the expert opinion of the authors and not representative of the entire species' range (Table 8.6c).

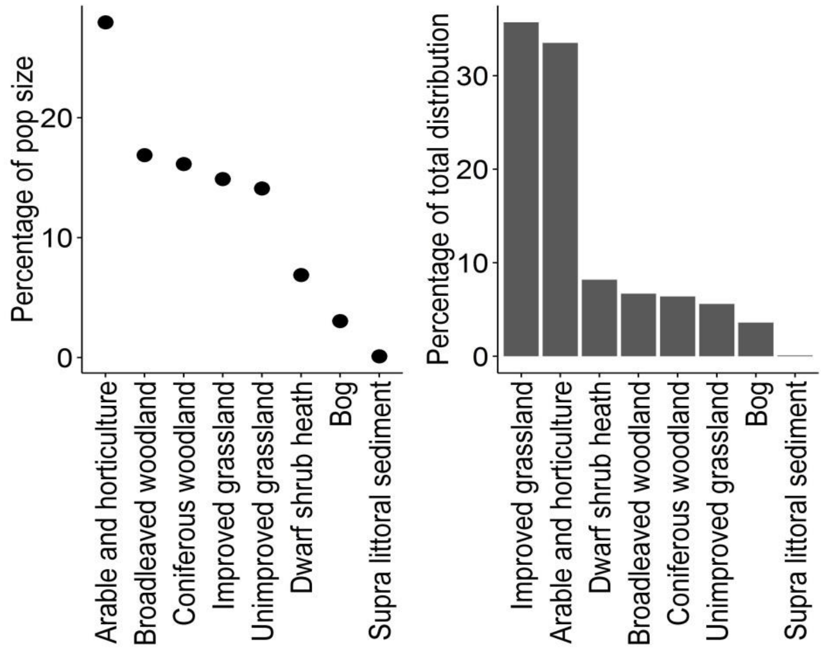


Figure 8.6b Left: The percentage of the total population of stoats accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 8.6c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Arable and horticulture	Improved grassland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 population density estimates	0	0
	1	10-30 population density estimates		
	2	>30 population density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**1**
		Overall reliability score	**1**	

Change through time

Comparison to Harris et al. (1995) and Arnold (1993)

Total population size as estimated by Harris et al. (1995) was 462,000, comprised of 245,000 in England, 180,000 in Scotland and 37,000 in Wales. Population density estimates for all habitat types are based on the authors' expert opinion (Harris et al., 1995). The density estimates in the current review are taken from Harris et al. (1995), so any differences in population size would be the result of divergent measurements of habitat availability.

Nationally, there are changes between the two reviews in the estimated availability of key habitats, generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape — using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008) — produces a new population size estimate of 399,000, which is 9% smaller than the estimate shown in Table 8.6b. The significance of this reduction cannot be assessed because of the lack of confidence limits for either estimate. The original estimate is close to the value presented by Harris et al. (1995), whereas the adjusted estimate is 14% lower.

Other evidence of changes through time

The National Gamebag Census suggests an increase of 28% (95% CI = 12%-42%) in the numbers of stoats culled between 1995 and 2009 (Aebischer et al., 2011), although it is unclear whether this increase is owing to an actual population size increase or an increase in trapping effort. A summary of trends in population size and range is provided inTable 8.6d.

Table 8.6d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries*		

*Aebischer et al. (2011).

Drivers of change

Table 8.6e Drivers of population change for stoats between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Change in prey availability.	A decline in specialist prey species, although the significance is unknown.	Harris and Yalden (2008)	Negative

Data deficiencies

Table 8.6f Areas where further research is required to improve the reliability of population size estimates for stoats.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	All habitats	All density estimates are taken from Harris et al. (1995), which were based the authors' opinions.
Density estimates do not represent within-habitat variability.	All habitats	No range or confidence limits were available from Harris et al. (1995).
Managed populations.	All habitats	The population size has not been adjusted to reflect the number culled.
No occupancy data.	All habitats	

Future prospects

Table 8.6g An assessment of the future prospects for the stoat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Stable

8.7 Weasel *Mustela nivalis*

Habitat preferences

The weasel occupies a wide range of habitats. As with other mustelids, there is territory defence against intruders of the same sex, and the smaller home ranges of females tend to be overlapped by one or more males (Powell, 1979). Home range size, and hence density, is strongly dependent on food availability. The weasel is a specialised predator of voles and mice, but will also take young rabbits, birds, and birds' eggs, particularly in spring when rodent populations are low (McDonald et al., 2000). Common in coniferous woodlands with dense field vole populations, the species is less abundant where small mammals are scarce, such as at high altitudes or in deciduous woodlands with sparse ground cover. On farmland, it is strongly associated with hedgerows, stone walls and other linear features, and rarely ventures into open habitat (see Harris and Yalden, 2008).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

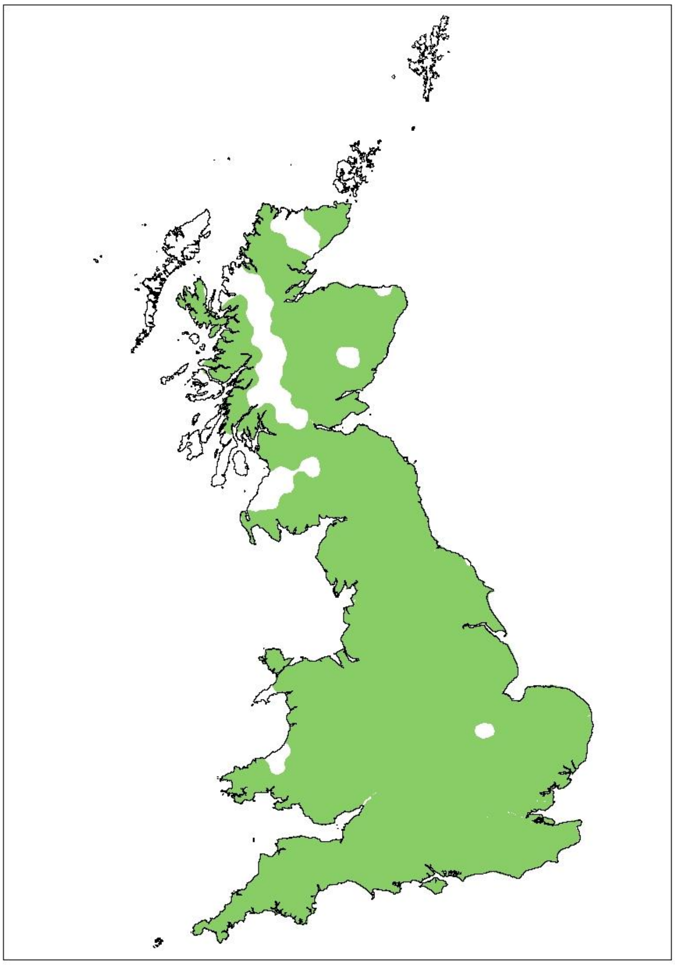


Figure 8.7a Current range of the weasel in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

No papers were identified by the literature search that reported pre- or post-breeding population density estimates. Experts were also unable to provide any further information. Harris et al. (1995) calculated weasel population size based on the ratio of weasels to stoats, which was thought to be 1:1 overall despite the weasel population showing more regional variability. As no new information was available for this review, the estimates previously given by Harris et al. (1995) — a total British population of 450,000, comprised of 308,000 in England, 106,000 in Scotland and 36,000 in Wales — could not be updated, and therefore no reliability assessment was conducted. The original estimate provided by Harris et al. (1995) was scored as extremely unreliable.

Changes through time

The only indicator of trends for weasels is from the GWCT National Gamebag Census. This indicated an increase of 51% (95% CI = 23%-80%), between 1995 and 2009, in the numbers of weasels culled. This suggests a recovery after declines of 37% (95% CI = 52%-22%) in cull rates between 1960 and 2009. The trend, however, may indicate changes in culling effort rather than changes in population size. A summary of trends in population size and range is provided in Table 8.7a.

Table 8.7a Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient	Scotland*	England Wales		

* The increase in range size in Scotland may be because of an altered survey effort.

Drivers of change

Table 8.7b Drivers of population change for weasels between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
N/A*	N/A	N/A	N/A

* Lack of data prevented an assessment of change in population size.

Data deficiencies

Table 8.7c Areas where further research is required to improve the reliability of population size estimates for weasels.

Data deficiencies	Habitat	Details
No population density estimates for specified habitat.	All habitats	No estimates are available pre- or post-1995.
No occupancy data.	All habitats	No occupancy data are available for any habitat.

Future prospects

Table 8.7d An assessment of the future prospects for the weasel, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	**Status**
Population	Unknown
Range	Unknown
Habitat	Stable

8.8 Polecat *Mustela putorius*

Habitat preferences

The polecat is a generalist species in terms of both habitat selection and diet. It tends to prefer woodland edge, farm buildings and field boundaries, and to avoid open fields (Birks and Kitchener, 2008; Birks, 2015). High road casualty rates may prevent the establishment of populations in urban and suburban areas, although it is occasionally found in these places (Birks, 1999). Unlike its counterparts in mainland Europe, the polecat in Britain does not show a preference for riparian habitats, possibly to avoid competition with the American mink. High rabbit abundance throughout the species' range provides an alternative food source outside of riparian habitats. A high proportion of activity is associated with rabbit warrens, and these sites are also frequently used for denning (Birks, 2015). The polecat is less strongly territorial than other small mustelids: territories can be vacated voluntarily and are not necessarily refilled (Blandford, 1986).

Status

Native.

Conservation status

- IUCN (GB: LC; England: [LC]; Scotland: [EN]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. UK: Favourable; England: Favourable; Scotland: Unknown; Wales: Favourable).

Species' distribution

In Scotland, records of true polecats are very sparse (see the 'Critique' section below). The highlighted areas on the distribution map below are therefore most likely to represent occasional individuals, or misidentified ferret-polecat hybrids, rather than an established population of true polecats.

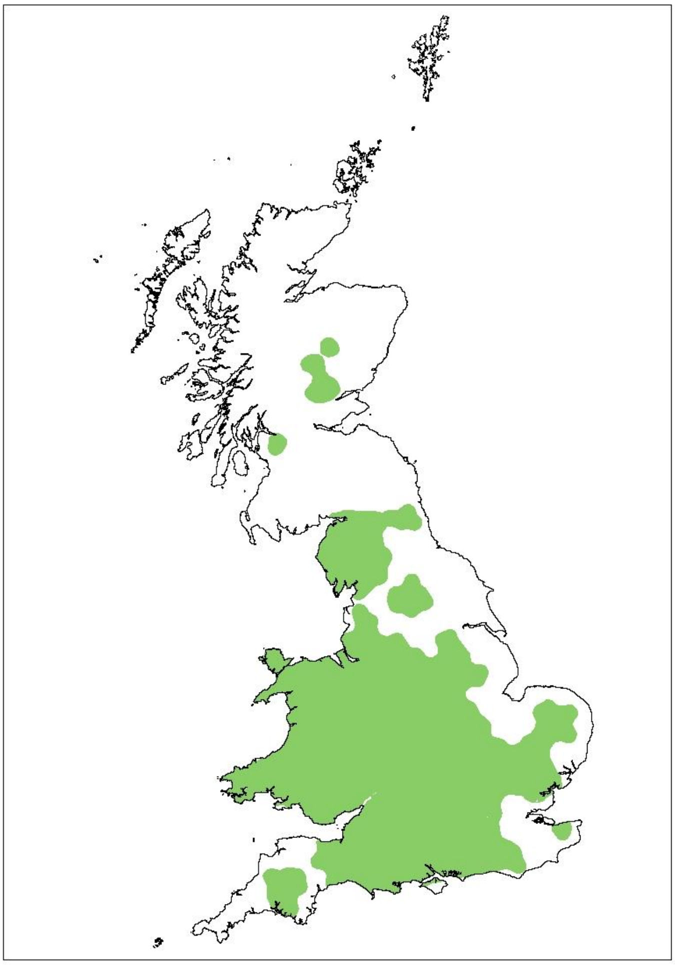


Figure 8.8a Current range of the polecat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Since polecats are generalists and can be found in most habitat types, population density estimates in the literature are not habitat-specific. To permit comparison to previous reports (and in the absence of any other relevant data), population sizes were therefore calculated by multiplying the population density by the total area of the species' distribution. Given that polecats are unlikely to occupy urban areas (Birks, 2015), areas classed as urban in the LCM2007 data were removed from the total distribution area using ArcGIS 10.2.2.

Occupancy data are taken from Birks & Kitchener (1999). In the original reference, occupancy is incorporated within the population density estimates: mean density was calculated as 0.85km^{-2} (95%CI = 0.69km^{-2}-1.01km^{-2}), where 52.3% of 1km squares were occupied (ranging from 56.1% in the centre of the range to 48.5% on the edge). Table 8.8a shows density for occupied squares only (Birks and Kitchener, 1999).

All records from Scotland are thought to be occasional records and/or misidentified ferret-polecat hybrids, so no population size was calculated for this country.

Results

Four relevant papers were identified by the literature search. One paper reported pre-breeding population density estimates and percentage occupancy, one contained estimates of total population size, and two gave details of distribution. Population density estimates are shown in Table 8.8a, and population sizes in Table 8.8b.

Table 8.8a Median density estimates with 95% confidence intervals for polecats, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
All habitats	98,000	1.63	1.32	1.93	Birks and Kitchener (1999)	136	52.3

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.8b Area of suitable habitat (not adjusted for occupancy) within England and Wales, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.8a with the area of habitat within the species' distribution, and adjusting for occupancy.

Country	Area of suitable habitat (km^2)*	Population size	-95%CI	+95%CI
England	78,100	66,400	53,900	79,000
Wales	19,800	16,800	13,700	20,000
Britain	**98,000**	**83,300**	**67,600**	**98,900**

* The area of suitable habitat is the total range size (see Table 8.5a) minus the area of urban and gardens. The area of suitable habitat excludes Scotland.

The Article 17 Report on polecat population size 2007-2012 is shown in Table 8.8c (Joint Nature Conservation Committee, 2013b). The population size estimated in the current review is almost double that reported in the Article 17 Report, though the geographical range sizes are similar (Table 8.8d).

Table 8.8c Article 17 Report on polecat population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	27,990	27,990
Scotland	350	350
Wales	18,450	18,450
Britain	**46,780**	**46,780**

Note: maximum and minimum estimates were the same values in the country-level reports.

Table 8.8d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013b). The extent of the true polecat range in Scotland is very uncertain.

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	85,400	n/a
Scotland	n/a	n/a
Wales	20,600	n/a
Britain	**105,900***	**118,720**

* Totals do not sum because of rounding.

Critique

Population size estimates for polecats were based on 136 individual density estimates from one study. These density estimates are area- rather than habitat-specific, so it is not possible to assess the proportion of the population size or geographical range accounted for by each habitat.

Density estimates and percentage occupancy values in Birks and Kitchener (1999) were taken from areas throughout the species' range in England and Wales. In Scotland, fewer than 85% of records received by the Vincent Wildlife Trust during 2014-2015 were classified as true polecats, as opposed to polecat-ferret hybrids or ferrets (Croose, 2016), and there were fewer than five verified records in the eastern fringe of the distribution. In contrast, most of the species' range in England contained 85% to 95% true-polecat records, and in Wales the value was >95%. It is therefore possible that the current estimate overlooks a small population of polecats in Scotland — Birks and Kitchener (1999) estimated the population in

Scotland to be between 345 and 483 — but this is unlikely to have a major impact on the total figures for Great Britain.

Density estimates are based on the number of sightings per survey. Therefore, they provide a minimum number, rather than a modelled estimate of density. Surveys were conducted between 1997 and 1999, so it would be beneficial to reassess population densities across the species' range, including recently recolonised regions. An assessment of reliability is given in Table 8.8e.

Table 8.8e Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of polecats. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All habitats
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	
	2	>30 population density estimates	2
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**4**
		Overall reliability score	**4**

Changes through time

Comparison to Harris et al. (1995)

Population size was estimated to be 15,000 by Harris et al. (1995), comprised of 2,500 polecats in England and 12,500 in Wales. Population sizes were estimated using more than one method, including applying high ($1km^{-2}$) and low ($0.1km^{-2}$) densities across the species' range. These density estimates resulted in population sizes ranging from approximately 2,000 to 21,000, and are comparable methodologically to the current review. Population sizes have increased significantly between the two reviews. This appears to be entirely driven by an increase in range, although more recent density data would help to verify this conclusion.

Other evidence of changes through time

The current findings concur with the increase in range and population size from 38,000 in 1997 (Birks and Kitchener, 1999) to 47,000 in 2006 (Birks, 2015). A summary of trends in population size and range is shown in Table 8.8f.

Table 8.8f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	England	Wales		
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 8.8g Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Recovery from persecution.	Alleviation of hunting pressure.	Birks (2015)	Positive
Rodenticides.	Increased consumption of rats contaminated with rodenticides may have lethal and sublethal effects.	Shore et al. (2003)	Negative
Hybridisation.	Ferrets and polecat-ferret hybrids are present in a considerable proportion of the true-polecats' range.	Costa et al. (2013)	Negative
Releases.	Releases into Cumbria and Perthshire/Tayside have resulted in the establishment of new populations and increases in the species' range.	Vincent Wildlife Trust (*pers. comm.*)	Positive

Data deficiencies

Table 8.8h Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density estimates are more than 10 years old.	Non-habitat specific	Population density estimates result from surveys conducted from 1994-1995 to 1998-1999.

Future prospects

Table 8.8i An assessment of the future prospects of the polecat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

8.9 Mink *Neovison vison*

Habitat preferences

The American mink became established in Britain following escapes or releases from fur farms in the early 20th century (Macdonald and Harrington, 2003). It is a generalist predator and shows a strong preference for riparian habitats, particularly those with abundant cover, where it feeds on a wide range of prey, including waterfowl, fish and water voles. High population densities are also found in undisturbed rocky coastal areas. Estuaries, urban canals, and habitats away from water may, also, provide sufficient habitat if cover and prey, such as rabbits, are available (Dunstone and Macdonald, 2008). In the Upper Thames region, mink were found to favour areas with tree and scrub cover and to avoid open areas, particularly farmland (Yamaguchi et al., 2003). There is inter-specific competition with the otter: declines in mink signs, and a shift towards a more terrestrial diet and diurnal rather than nocturnal behaviour, have been noted to correlate with the resurgence of otter populations (Bonesi and Macdonald, 2004; Bonesi et al., 2006; McDonald et al., 2007; Harrington et al., 2009)

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 8.9a. Gaps in the species' distribution in the Scottish Borders and Argyll are likely to represent areas lacking survey effort, rather than true absences. The species is known to be present in all areas of mainland Scotland except for the far north (see Fraser et al., 2015b; Gaywood et al., 2016).

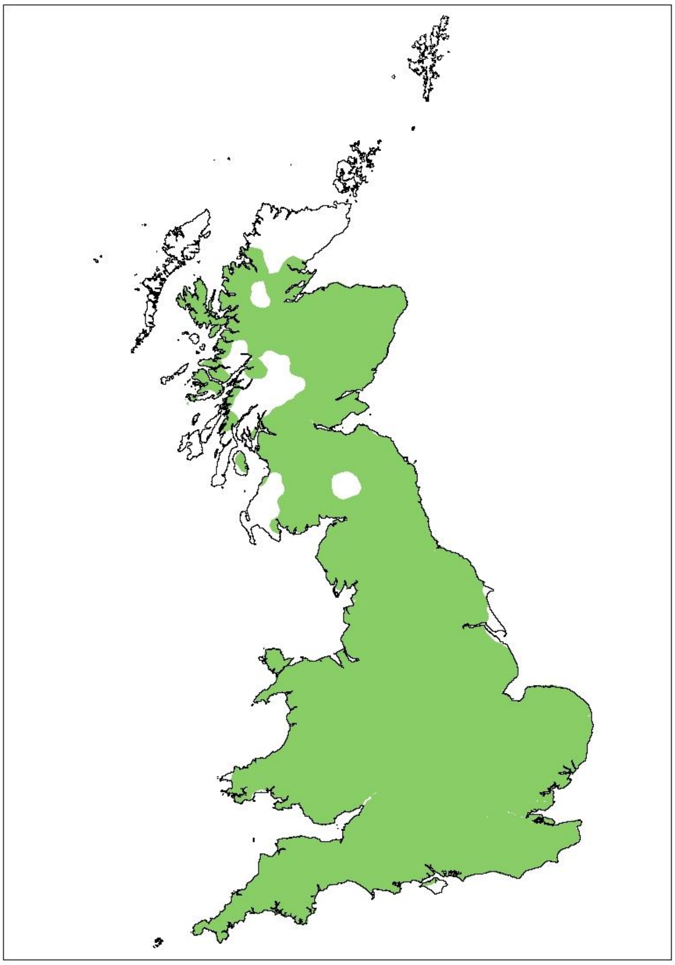


Figure 8.9a Current range of the mink in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. Mink were previously considered abundant in the south west of Scotland, but there are no records from this region since 1995, possibly because of a lack of recorder effort.

Species-specific methods

The length of riparian habitat for Scotland, England and Wales was taken from Table 4 in the previous *Review of British Mammals* (Harris et al., 1995) and multiplied by the percentage of each country included in the species' distribution to give the length of available riparian habitat. Percentage occupancy was taken from Bonesi et al. (2006), using the percentage of sites (n=3188) within 32 50 x 50km squares surveyed during the National Otter Survey of England. The length of suitable coastline was taken from Table 10.3 in Jefferies et al. (2003). As there have been no records of mink in the Outer Hebrides since 1995, these islands were not included. The coastlines of Arran, Skye and Mull were adjusted using the percentage occupancy for Arran (Jefferies et al., 2003).

Results

Eight papers were identified by the literature search. One reported a pre-breeding population density estimate for rivers, and two gave pre-breeding density estimates for coastal populations. One paper contained home range estimates (Males = 1.5 km, females = 1.09 km; Dunstone and Birks, 1985), and one gave presence and occupancy data. The remaining papers contain post-breeding or relative measures of population density. Percentage occupancy was estimated to be 74% by Bonesi et al. (2006) for riparian populations, whilst occupancy for coastal populations was taken from Jefferies et al. (2003): 5.97% in England; 15.15% in the Scottish mainland; 37.5% in Arran (applied to all currently occupied Scottish islands); and 3.57% in Wales. Overall percentage occupancy for coastal populations is given in Table 8.9a. Population density estimates are provided in Table 8.9a and population size estimates are provided in Table 8.9b.

Table 8.9a Median density estimates with 95% confidence intervals for mink, calculated using data obtained from a review of the literature from 1995 to 2015. The literature sources used to estimate each value are listed in 'Source'. The number of estimates obtained from each literature source is given in 'n'.

Habitat	Length within range (km)	Density (km^{-1})	-95%CI	+95%CI	Source*	n**	%Occ†
Riparian	261,000	0.62	-	-	Harrington et al. (2008)	1	74
Coastal	10,600	1.395	1.25	1.54	Jefferies et al. (2003)	1	12.5
					Helyar (2005)	1	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 8.9b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 8.9a with the length of rivers obtained from Table 4 of Harris et al. (1995), and adjusted by the percentage of rivers occupied in Table 8.9a. It was not possible to calculate confidence intervals, as none were available for the density estimate from Harrington et al. (2008).

Country	Length within range (km)	Population size*	-95%CI	+95%CI
England	139,000	[62,400]	-	-
Scotland	104,000	[46,600]	-	-
Wales	29,100	[12,900]	-	-
Britain	**273,000**	**[122,000]**	**-**	**-**

*Across Great Britain, 1.5% of the population is estimated to be in coastal environments.

Critique

Population size in riparian habitats was estimated from a single density estimate. Coastal populations, which account for only 2% of the total population size, were based on two density estimates. The small contribution of coastal areas to the overall population is, in part, owing to the length of available coastline, which is substantially shorter than the length of available riparian habitat. However, there is also a large difference in the percentage occupancy values for the two habitat types. The most recent values for percentage occupancy were used in each case, with the value for coastlines taken from Jefferies et al. (2003) and riparian habitats from Bonesi et al. (2006). Occupancy of riparian habitats was also provided in Jefferies et al (2003) and, as for coastlines, was calculated as the percentage of 10 x 10km squares positive for mink within Water Authority Regions, or longitudinal sections of differing river length and 100km width. In contrast, Bonesi et al. (2006) used the percentage of occupied sites within alternate 50km x 50km squares. Both Jefferies et al. (2003) and Bonesi et al. (2006) reported the same declining temporal trend in occupancy relative to the same measures of occupancy in previous years, but the absolute occupancy values are not comparable between studies. For the older survey of riparian habitats (Jefferies et al., 2003), the values were 13.42% in England, 10.69% in Scotland and 3.74% in Wales. If these values are used in place of those from Bonesi et al. (2006), the population size estimate is reduced considerably to 20,300 in Britain, comprised of 11,600 in England, 8,000 in Scotland and 700 in Wales. A reliability assessment is provided in Table 8.9c.

Table 8.9c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of mink. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and the availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			Riparian
Location of study sites	0	Estimates from one location	0
	1	Estimates restricted	
	2	Estimates widespread	
Sample size	0	<10 population density estimates	0
	1	10-30 population density estimates	
	2	>30 population density estimates	1
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**1**
		Overall reliability score	**1**

Change through time

Comparison to Harris et al. (1995)

The population size estimated by Harris et al. (1995) was at least 110,000 individuals, with 46,750 in England, 52,250 in mainland Scotland and 9,750 in Wales. The authors stated, however, that more information was needed on the coastal and island population to improve the reliability of the estimate, and suggested that the percentage of occupied habitat (which was based on data from the English National Otter Survey and expert opinion) was likely to be underestimated. The same problems persist in the current review, and so the identification of temporal trends has been limited to comparisons within survey types, i.e., The Water Vole and Mink Survey of Britain (Jefferies, 2003), or The National Otter Survey dataset (see Bonesi et al., 2006).

Other evidence of changes through time

Both The Water Vole and Mink Survey of Britain (Jefferies et al., 2003) and The National Otter Survey dataset (see Bonesi et al., 2006) suggest declining trends in percentage occupancy for mink. A comparison of population size calculated using the percentage occupancy for riparian habitats from Jefferies et al. (2003) suggests a 65% decline in population size between 1989-1990 and 1996-1998 (from 105,650 to 36,950), and a 45% decline between 1996-1998 and 2016 (from 36,950 to 20,500), with the largest decline found in Scotland (58%, as opposed to 29% in England and Wales). Conversely, although range size has declined in Scotland since 1993 (Arnold, 1993), it has increased in England and Wales. This trend may be an artefact of more intense recording in England and Wales in more recent years, or it could potentially be the result of animals dispersing more widely in response to control measures.

The GWCT National Gamebag Census for mink suggests a decrease of 41% (95%CI = 49%-33%) in culling rates between 1995 and 2009. However, this trend is not adjusted for effort, which may also vary over time. A summary of trends in population size and range is provided in Table 8.9d.

Table 8.9d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease	England Wales*		Scotland*	
	Data deficient				

* Population decline is inferred from a fall in site occupancy and density within the species' range. Apparent increase in range size may be owing to increased recorder effort since 1995; and there is evidence of recent decline in parts of Scotland following concerted control efforts.

Drivers of change

Table 8.9e Drivers of population change for mink between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Predation/ competition.	Encroachment by competitor (otters).*	Bonesi et al. (2006)	Negative
Management (control).	Localised population suppression.	GWCT, SNH	Negative

* There is no evidence of this behaviour in Scotland.

Data deficiencies

Table 8.9f Areas where further research is required to improve the reliability of population size estimates for mink.

Data deficiencies	Habitat	Details
Limited density estimates for key habitat.	Riparian Coastal	One recent estimate is available for riparian habitat.
Density estimates do not represent within-habitat variability.	Riparian	It was not possible to calculate confidence limits for the riparian density estimate.
Occupancy information out of date	Riparian Coastal	Occupancy data were published in 2006 (Riparian) and 2003 (Coastal).

Future prospects

Table 8.9g An assessment of the future prospects for the American mink, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable/Decline*
Range	Stable/Decline*
Habitat	Stable

*Possible future decline owing to control measures.

9 ARTIODACTYLA

9.1 Wild boar *Sus scrofa*

Habitat preferences

The wild boar incorporates a variety of habitats into its home range, but is mainly associated with woodlands (Spitz and Janeau, 1990; Gerard et al., 1991). The species can cause damage to agricultural crops, particularly where agricultural land borders woodland, although they also make use of linear features such as hedgerows, stone walls and ditches for shelter while moving through the landscape (Thurfjell et al., 2009).

Status

Native (extinct in Britain by the 13th century; current populations derived from unknown sources).

Conservation Status

- IUCN Red List (GB: NT; England: [DD]; Scotland: [DD]; Wales: [DD]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 9.1a. In addition to the areas presented, two established populations are known to exist in Dumfries and Galloway (Campbell and Hartley, 2010). The distribution map does not show the location of feral pigs.

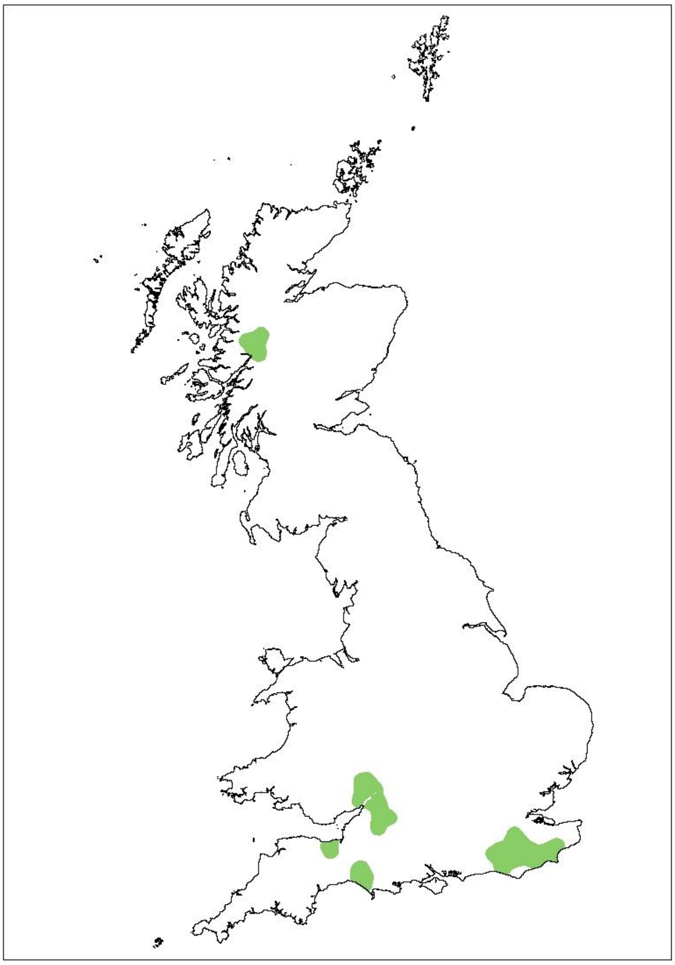


Figure 9.1a Current range of the wild boar in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Results

Three papers were returned from the literature search. One paper contained pre-breeding density estimates, one contained a number for road kill only, and one contained a total count without a defined area. Population density estimates per habitat are provided in Table 9.1a, and total population size estimates in Table 9.1b.

Table 9.1a Median density estimates with 95% confidence intervals for wild boar, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	900	2	0.1	6	Wilson (2003)	8	n/a
					Gill (2014)	40	
Coniferous woodland	400						

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.1b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.1a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	200	500	30	1,500
Scotland	1,000	2,000	100	6,500
Wales	<100	150	<10	500
Britain	**1,300**	**2,600**	**200**	**8,400**

Critique

The population size calculation is based on median densities in woodland habitats only, because data were lacking for other locations. In itself, this is unlikely to have introduced a serious error, as woodland is a core habitat for current populations, but 100% occupancy was assumed in all woodlands across the range, which will have overestimated population size. In addition, the density values, while derived from 48 individual estimates across two studies, came from known strongholds in the Forest of Dean (Gill, 2014) and in Dorset (Wilson, 2003). It is highly likely that densities elsewhere in southern England and in Scotland are lower, again suggesting an overestimate of population size. However, wild boar may be present in additional locations that are not recorded, and this error would act in the opposite direction. The population size for Scotland, in particular, should therefore be viewed with caution, and further surveys to clarify the status of wild boar are advised.

Although wild boar make use of a variety of habitats within their home range, they are primarily associated with woodlands. The population size estimate based on woodlands only is likely to represent the total population. Records of culled animals were included in the data used to estimate the current population size (Wilson, 2003), suggesting a source of overestimation. A reliability assessment is provided in Table 9.1c.

Table 9.1c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of wild boar. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All woodland
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 density estimates	
	1	10-30 density estimates	1
	2	>30 density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Habitat score	**2**
		Overall reliability score	**2**

Changes through time

Comparison to Harris et al. (1995)

Only sporadic records of wild boar, derived from escapes and releases, were present in Britain in 1995, so the population was not reviewed by Harris et al. (1995).

Other evidence of changes through time

Although the wild boar became extinct in Britain in the 13th century, captive animals have been kept in wildlife collections, zoos and farms since the 1980s. Over the last 10 to 15 years, small populations have become established as a result of escapes and deliberate releases. In 1998, there were two populations in Kent and Dorset, consisting of approximately 100 and 12-20 animals, respectively (Goulding et al., 1998; Wilson, 1999). By 2003, the population in Dorset was reported to be well-established and breeding, but culling pressure meant that range expansion was slow and population size had remained small (Wilson, 2003). In the Forest of Dean, a large population has become established as a result of accidental releases from a wild boar farm in the 1990s and an illegal release of around 60 farm-reared boar in 2004 (Dutton et al., 2015). The population has increased significantly since then, with an estimate of just under 1,000 in 2008 for the whole of the UK (DEFRA, 2008), to 1562 (95%CI = 1095-2296) in the Forest of Dean alone in 2016 (Gill and Waeber, 2016). This most recent estimate in the Forest of Dean included non-mature individuals, and approximately a quarter of the population were piglets: it is therefore likely that the total number of mature individuals is closer to 1,000. This increase does, however, suggest that our figure of 500 wild boar in England is a significant underestimate.

Recent sightings in the western Highlands in Scotland, the first of which was reported in 2007, suggest that a small population (estimated at 60 individuals) may have become established in Lochaber (Tony Mitchell-Jones, *pers. comm.*). In addition, two populations are known to have become established in Dumfries and Galloway (Campbell and Hartley, 2010), although further details are not known. The population in Scotland is therefore likely to be higher than estimated in this review. A summary of trends in population size and range is provided in Table 9.1d.

Table 9.1d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	England Scotland			
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 9.1e Drivers of population change for wild boar between 1995 and the present.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Escapees from wildlife collections, zoos and farms have resulted in established populations.	Goulding et al. (1998) Wilson (1999)	Positive
Management (control).	Culling has slowed the increase in population size and range.	Wilson (2003)	Negative

Data deficiencies

Table 9.1f Areas where further research is required to improve the reliability of population size estimates for wild boar.

Data deficiencies	Habitat	Details
Managed populations.	Woodland	Culled individuals are included in density estimates.
No occupancy data.	Woodland	

Future prospects

Table 9.1g An assessment of the future prospects for the wild boar, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

9.2 Red deer *Cervus elaphus*

Habitat preferences

The red deer in Great Britain is most commonly associated with upland open moorland habitats, where it lives in sexually segregated herds. However, there is evidence that it prefers woodland habitats, particularly in other parts of its global range (Clutton-Brock and Albon, 1989): in Britain, there are small herds that live in woodland all year round, apparently benefiting from greater foraging resources. In summer, the open habitat populations feed primarily on graminoids (Latham et al., 1999), and prefer areas with grass or heather rides (Welch et al., 1990). In winter, these populations move to lower ground in search of grazing opportunities and shelter, switching to foraging primarily on heather (Latham et al., 1999). Although afforestation has resulted in a loss of some traditional overwintering habitat, the red deer has become established in plantations. It is managed by culling throughout its range, largely for sport and food, although recently in some areas also to reduce the impact of grazing on plant — and associated animal — biodiversity in woodlands (Trenkel et al., 1998). On some estates in Scotland, high numbers of red deer are promoted by winter feeding and reduced culling in order to increase numbers for sport hunting (Putman and Staines, 2004).

Status

Native, although most populations, including all those in Wales, are relatively recent reintroductions.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

The red deer has a patchy distribution throughout its range, and records in England and Wales are particularly scattered. The process used to create the smoothed distribution map (Figure 9.2a) means that small distribution gaps are not evident. Therefore, the map presented below may overestimate the range. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

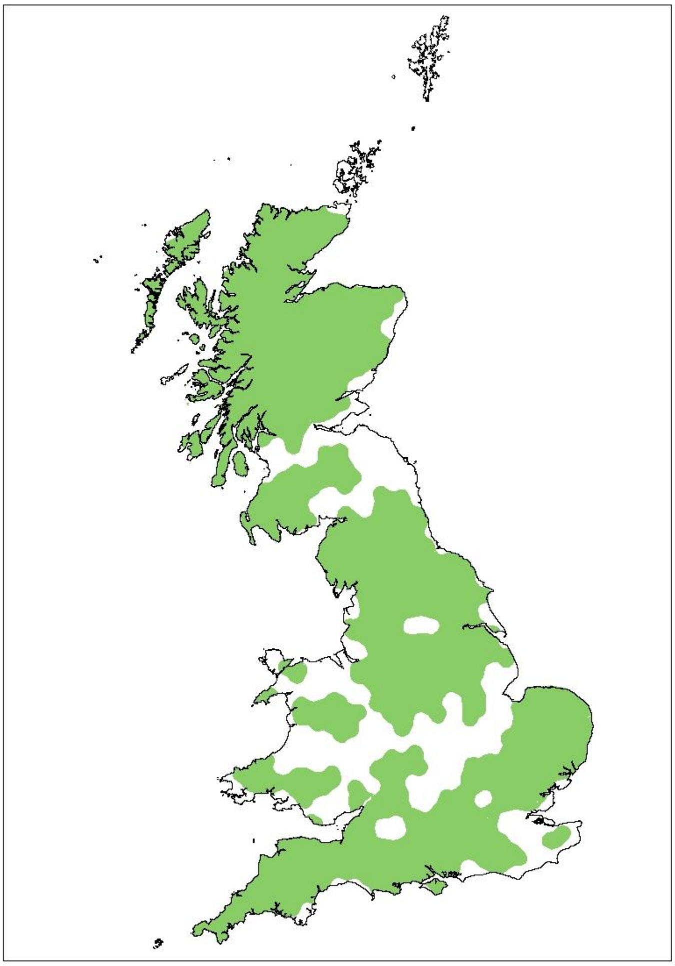


Figure 9.2a Current range of the red deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Although some deer surveys were habitat-specific (e.g., woodlands surveyed by faecal pellet counts (Latham et al., 1996; Mayle, 1996)), many were at landscape scale and included several habitat types (Trenkel et al., 1998; Daniels, 2006). For the latter, the density estimates were included in the assessment for suitable habitats found within the study areas. Suitable habitats were considered to be broadleaved and coniferous woodlands, dwarf shrub heath and montane. Expert opinion estimates were provided for improved grassland. However, it is assumed that any deer counted in improved grassland are transient and will be included within the estimates for the other suitable habitats.

Density estimates vary widely depending on region, and particularly between countries. Population size was calculated for woodlands, within specific regions of England and for Wales, using region-specific density data from Iossa et al. (2009). No data were available on the density of red deer in dwarf shrub and montane habitats for England or Wales, so data from Scottish studies were applied. For Scotland, density estimates were calculated by applying habitat-specific population density estimates. Data specific to England and Wales from Iossa et al. (2009) were excluded from the analysis for Scotland.

Results

Ten papers were identified by the literature search. Five papers reported total counts, post-breeding density estimates, or mean year-round densities. One paper provided a density estimate for females only. The population density estimates are shown in Table 9.2a, and population size estimates in Table 9.2b.

Table 9.2a Median density estimates with 95% confidence intervals for red deer, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Region	Area within range (km^2)	Density (km^2)	-95% CI	+95 %CI	Source*	n**	%Occ†
Broadleaved woodland	Scotland	1,970	12.7	9.2	13.4	Trenkel et al. (1998) Daniels (2006)	8 6	n/a
Coniferous woodland	Scotland	8,680	9.3	6.6	12.3	Latham et al. (1996) Trenkel et al. (1998) Daniels (2006)	20 8 11	n/a
Broadleaved woodland	England/ Wales	8,000	5.3	1.1	7.9	Iossa et al. (2009)	7	n/a
Coniferous woodland	England/ Wales	3,000	5.3	1.1	7.9	Iossa et al. (2009)	7	n/a
Montane habitats	All	4,980	10.9	5	15.0	Expert opinion		n/a
Dwarf shrub heath	All	17,800	7.2	5.6	12.6	Trenkel et al. (1998) Daniels (2006) Perez-Espona et al. (2010)	8 11 13	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.2b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.2a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	13,500	79,700	31,400	124,000
Scotland	29,200	256,000	176,000	376,000
Wales	1,750	10,200	4,110	16,100
Britain	**44,400**	**346,000**	**212,000**	**516,000**

Critique

Populations of red deer have patchy distribution, particularly in England and Wales, but occupancy data with which to refine the estimates were not available. Therefore, population sizes are overestimated, most notably in England and Wales. The woodland density estimates for England and for Wales are based on a single source (Iossa et al., 2009). Whilst these are more realistic than using estimates from surveys in Scotland, and were drawn from 44 locations, the use of a single source nevertheless introduces considerable uncertainty to the estimates. Trends in deer population density vary widely in different parts of Scotland, from 1.9km^{-2} in the Cairngorms and east Loch Lomond to 15.1km^{-2} in Glenelg and Knoydart (Scottish Natural Heritage, 2016). Stratification by region may therefore be advisable for future assessments of population size.

Edwards and Kenyon (2013) reported an estimated population size in Scotland of 400,000 in 2011, based on annual reports from the Deer Commission for Scotland (DCS) (which merged with SNH in 2010) and its predecessor the Red Deer Commission. This estimate is above the upper confidence limit of the current estimate. A more recent estimate of 360,000-400,000 has also been provided (Scottish Natural Heritage, 2016), with the lower end of this range falling just within the current confidence limit.

Most of the red deer population is found in dwarf shrub heath (32%) or coniferous woodland (28%), both of which constitute the majority of habitat in the species' range (Figure 9.2b). The density estimates for these habitats are derived from, respectively, 32 (dwarf shrub heath) and 39 (coniferous woodlands) individual density estimates. A reliability assessment is provided in Table 9.2c.

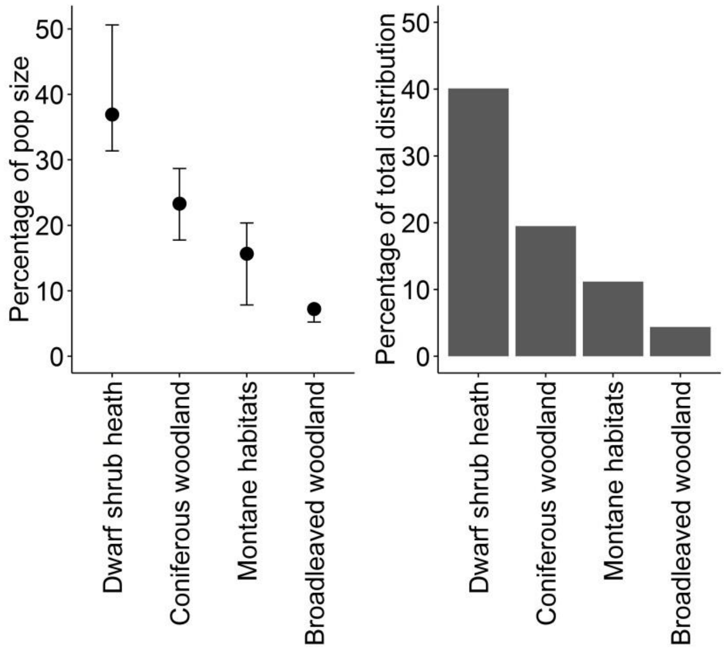


Figure 9.2b Left: The percentage of the estimated red deer population derived from each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 9.2c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of red deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**	
			Dwarf shrub heath	**Coniferous woodland**
Location of study sites	0	Estimates from one location		
	1	Estimates restricted		
	2	Estimates widespread	2	2
Sample size	0	<10 population density estimates		
	1	10-30 population density estimates		
	2	>30 population density estimates	2	2
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**4**	**4**
		Overall reliability score	**4***	

*The reliability score applies to the entire British population, most of which is in Scotland. In England and Wales, although estimates were widespread and came from >30 locations, it is anticipated that the lack of occupancy data will introduce considerable error compared with Scottish fell populations.

Changes through time

Comparison to Harris et al. (1995)

Harris et al. (1995) reported a total of 360,000 red deer, with 12,500 in England, 347,000 in Scotland and fewer than 50 in Wales. The overall figure for Great Britain is very similar to the current report, but the relative abundance across the three countries differs: Harris et al. suggested that there were fewer red deer in England and Wales, and more in Scotland. However, there are significant methodological differences between the two reports that complicate any inference of trends over time. For England, Harris et al. (1995) totalled the sizes of known herds or populations within each region, excluding areas known to consist of red-sika hybrids. This method may have underestimated the total population, as understudied areas or undocumented herds were not included. In contrast, the current report is likely to have overestimated the population, because it assumes 100% occupancy of all suitable habitat within the species' distribution. For Wales, Harris el al. (1995) used records from the National Mammal Atlas. In Scotland, they used calculations by Clutton-Brock and Albon (1989) based on census data from the Red Deer Commission, but included a separate count for woodlands (Staines and Ratcliffe, 1987).

Other evidence of changes through time

Over the last 15 years there has been considerable variability in the numbers of red deer culled in Scotland. The reported numbers culled peaked in 2004-2005, decreased substantially to its lowest level in 2011-2012, but by 2014-2015 had returned to 2004-2005 cull levels (over 68,000 per annum) (Scottish Natural Heritage, 2016). In contrast, the National Gamebag Census indicates that the numbers culled in England have increased by 27% between 1995 and 2014 (95%CI 32% decrease to 296% increase; Nicholas Aebischer, *pers. comm.*). However, this trend does not account for hunting effort, which may also vary over time.

The Deer Management Report (Scottish Natural Heritage, 2016) suggests opposing trends in abundance for different habitats in Scotland: there has been a decline in National Forest Estate woodlands of 12% between 2001 and 2016 compared with an increase (which has plateaued in recent years) in open ground. Over the longer term, population densities increased across Scotland between 1961 and 2000-2001, and have remained roughly stable since then (Scottish Natural Heritage, 2016).

Geographical range size increased by 0.3% per year between 1972 and 2002 (Ward, 2005; Ward et al., 2008), followed by a slower increase from 2007-2011 (Scottish Natural Heritage, 2016). A summary of trends in population size and range is provided in Table 9.2d.

Table 9.2d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	All countries*			
	Stable				
	Decrease				
	Data deficient				

* Despite an increase in population size in Scotland since 1995, rates are thought to have slowed; it is unlikely that the population is still increasing. Trends for England and Wales are based on the assumption that an increase in range (Ward et al., 2008, Scottish Natural Heritage 2016 annex 2) is the result of an increase in population size.

Drivers of change

Table 9.2e Drivers of population change for red deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Hybridisation with sika deer.	Goodman et al. (2001) Senn and Pemberton (2009)	Negative
Management (control).	Deer are culled to reduce the impact of grazing on plant biodiversity and associated animal biodiversity.	Trenkel et al. (1998)	Negative
Management (feeding).	High deer numbers are promoted on Scottish deer estates by winter feeding and insufficient culling.	Putman and Staines (2004)	Positive
Habitat availability.	Increased woodland availability (4.7% increase in broadleaved woodland, and 6.4% increase in coniferous woodland, between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008)	Positive

Data Deficiencies

Table 9.2f Areas where further research is required to improve the reliability of population size estimates for red deer.

Data deficiencies	Habitat	Details
No occupancy data.	All habitats	No occupancy data are available in the current literature.
Limited density estimates for key habitat.	Broadleaved woodland Coniferous woodland	Median density estimates are based on 7 density estimates for both habitats.
No density estimates for specified habitat.	Montane	Density estimates are based on expert opinion.
Managed populations.	All habitats	Management is not taken into account in the population size estimate.

Future prospects

Table 9.2g An assessment of the future prospects for red deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects see Appendix 7.

Trend	Status
Population	Increase*
Range	Increase*
Habitat	Stable

*Increase overall, but population and range have most likely stabilised in Scotland.

9.3 Sika deer *Cervus nippon*

Habitat preferences

The Sika deer is well established in Britain, and is associated with heathland and young pre-thicket- and thicket-stage coniferous woodlands (Horwood and Masters, 1970; Uzal Fernandez, 2010). It is, however, relatively adaptable, and occupies habitats that provide substantially different resources from those found within their native range in East Asia (Mann and Putman, 1989). In the New Forest, it makes extensive use of conifer plantations (44% of transect observations across the year) and oak woodland (42% of transect observations across the year), and little use of agricultural or other open habitats (Mann and Putman, 1989). In contrast, in a study area in Dorset where little broadleaved woodland was available, most animals were recorded in conifer woodland, but they also used heathland, saltmarsh and agricultural fields beyond the forest boundary at night, with almost all feeding activity occurring in these habitats (Mann and Putman, 1989). In both study areas, activity in conifer woodland was focused in young thicket plantation.

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

Records of sika deer in England and Wales are scattered, with a patchy distribution throughout their range. The method used to create the smoothed distribution map (Figure 9.2a) means that small gaps in the distribution are not evident. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

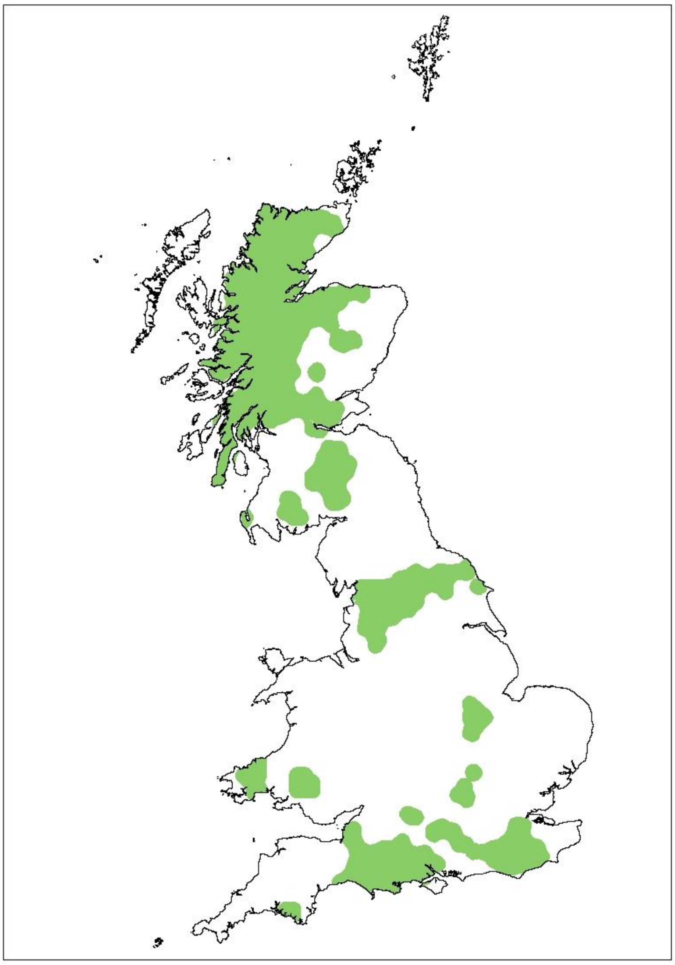


Figure 9.3a Current range of the sika deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Sika deer have extensive home ranges that encompass a range of habitat types, although they are usually associated with woodland and heathland. Data on pre-breeding population density were not available for broadleaved woodland, even though this habitat is used extensively (Mann and Putman, 1989; Uzal Fernandez, 2010), often as part of a mosaic of suitable habitat. Population densities in coniferous woodland were extrapolated to broadleaved woodland, following expert advice. The species also makes extensive use of lowland heath in England, so values were derived from the long-term study by Uzal Fernandez (2010). However, they were not applied to Scotland because the broad habitat category of 'dwarf shrub heath' in that country would include large areas not used by sika deer, and in the absence of occupancy data, this would have introduced a far greater error than by simply excluding the habitat.

Results

Four papers were identified the literature search. One of these reported pre-breeding density estimates, and the remainder gave post-breeding estimates or total counts without a defined area. One extensive study summarised data from 3 years of observation, and was included despite the final density estimates incorporating some information from the post-breeding period (Uzal Fernandez, 2010). The population density estimates are shown in Table 9.3a, and population size estimates in Table 9.3b.

Table 9.3a Median density estimates with 95% confidence intervals for sika deer, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^2)	-95%CI	+95%CI	Source*	n**	%Occ†
Coniferous woodland	7,200	7.08	2.35	19.6	Marques et al. (2001)	8	n/a
Dwarf shrub heath	1,300	18.3‡	0.5	36.0	Uzal Fernandez (2010)	1	n/a
Broadleaved woodland	4,000	7.08	2.35	19.6	expert opinion (extrapolated from coniferous woodland density)	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.
‡ Value set as median of reported ranges.

Table 9.3b Total population size estimates, with 95% confidence intervals, for England, Scotland, Wales and the whole of Britain. Values were obtained by multiplying population density estimates in Table 9.3a with the area of habitat within the species' distribution. Small discrepancies between this calculation and the population sizes presented are owing to rounding errors.

Country	Area of suitable habitat (km^2)	Population size*	-95%CI	+95%CI
England	4,400	45,300	8,200	107,000
Scotland	7,600	54,000	17,900	149,000
Wales	400	3,600	900	9,300
Britain	**12,500**	**103,000**	**27,000**	**266,000**

Critique

Occurrence data were not available, so 100% occupancy of woodlands and dwarf shrub heath (excluding Scotland) within the range was assumed. However, as populations of sika deer, particularly in England and Wales, are known to have a patchy distribution, there may be some overestimation for England and Wales.

The median population density estimate for coniferous woodland is derived from multiple estimates from a single paper. All of these estimates were taken from southern Scotland (Marques et al., 2001), and the limited spatial range means that environmental conditions affecting deer density are unlikely to be reflected in the confidence limits of the estimate. In addition, survey effort in coniferous woodland and dwarf shrub heath was highest in areas of greatest perceived deer density, which may have resulted in a biased assessment of population density. In Arne and Hartland in the south of England, for example, robust estimates made across the mosaic of lowland shrub heath and woodland indicate densities of 118km^{-2} and 27km^{-2} respectively: these values are far higher than any of the habitat-specific density estimates. Given the patchy distribution of the species, further research to establish occupancy and also densities within occupied areas is urgently required. Finally, it should be noted that the density estimates are somewhat dated, particularly for coniferous woodland. A reliability assessment is provided in Table 9.3c.

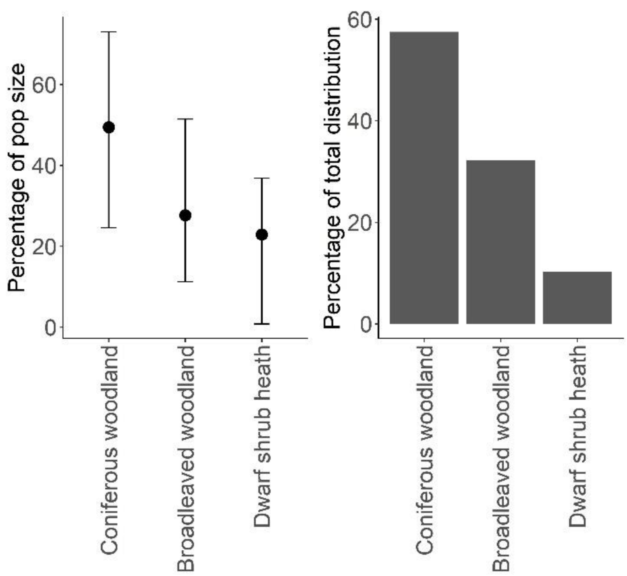


Figure 9.3a Left: The percentage of the total population of red deer accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 9.3c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of sika deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Coniferous woodland	Broadleaved woodland*
Location of study sites	0	Estimates from one location		0
	1	Estimates restricted	1	
	2	Estimates widespread		
Sample size	0	<10 population density estimates	0	0
	1	10-30 population density estimates		
	2	>30 population density estimates		
Occupancy data available?	0	No	0	0
	1	Yes		
		Habitat score	**1**	**0**
		Overall reliability score	**0.5**	

* Expert opinion only.

Changes though time

Comparison to Harris et al. (1995)

Harris et al. (1995) estimated the British population as 11,500, comprised of 2,500 in England and 9,000 in Scotland. Population sizes in England were estimated from total counts of known populations. In Scotland, the estimates were derived from expert opinion on the density (20km^{-2} in suitable habitat), area of colonisation, and an assumed percentage of this area with suitable habitat (25%). The current estimates are based on population densities and the area of suitable habitat within the species' range, so it is difficult to make a direct comparison with the previous review.

Other evidence of changes through time

The GWCT National Gamebag Census found that the number of sika deer culled in Scotland between 1995 and 2014 has increased by 35%, although this trend is not significant (95%CI

8% decrease to 74% increase, Nicholas Aebischer, *pers. comm.*). However, this trend does not account for hunting effort which may also vary over time.

Although the population of sika, fallow and roe deer in Scotland's National Forest Estate has declined by 30% between 2001 and 2016 (Scottish Natural Heritage, 2016), an increase of 46% was found in the Scottish borders between 1998 and 2004. Geographical range size increased by 5.3% per year between 1972 and 2002 (Ward, 2005), followed by a further increase between 2007 and 2011 (Ward et al., 2008; Scottish Natural Heritage, 2016). A summary of trends in population size and range is shown in Table 9.3d.

Table 9.3d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	All countries*			
	Stable				
	Decrease				
	Data deficient				

* Increase in population size is assumed from an increase in range (Ward et al., 2008, Scottish Natural Heritage 2016 annex 2), although trends are not certain.

Drivers of change

Table 9.3e Drivers of population change for sika deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Management (control).	6675 sika deer were reported culled in Scotland in 2014-2015, which is 14% of the estimated population.	National Gamebag Census, GWCT	Negative
Hybridisation.	Hybridisation with red deer has limited the spread of pure sika deer, in addition to the effect on red deer populations.	Senn and Pemberton (2009) Goodman et al. (1999)	Negative
Habitat availability.	Increased woodland availability (4.7% increase in broadleaved woodland and 6.4% increase in coniferous woodland between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008)	Positive

Data Deficiencies

Table 9.3f Areas where further research is required to improve the reliability of population size estimates for sika deer.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Heathland	No density estimates were available from recent literature or expert opinion.
No occupancy data.	All habitats	No occupancy data were available in the current literature.
Limited density estimates for key habitat.	Coniferous woodland	Median density estimates are based on 8 individual densities.
Density estimates are more than 10 years old.	Coniferous woodland	
Managed populations.	All habitats	Management is not taken into account in the population size estimate.

Future prospects

Table 9.3g An assessment of the future prospects for the sika deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

9.4 Fallow deer *Dama dama*

Habitat preferences

The fallow deer is typically associated with woodland, which it uses primarily for shelter. It prefers broadleaved or mixed woodlands with an established understory, but will also colonise coniferous plantations containing some open areas. It frequently forages outside woodland, in grasslands and arable fields, particularly at night. In autumn and winter, greater use is made of woody forage and mast crops. Social groups, usually of fewer than five individuals and comprised of one or two adult females and their young, are common, although larger groups are also sometimes formed. Males live in bachelor groups for most of the year. The species is non-territorial, and home ranges, which are usually between 100ha and 200ha, overlap extensively (see Harris and Yalden, 2008).

Status

Naturalised (extinct in Great Britain by the last Ice Age, then introduced by the Normans).

Conservation status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

The distribution of fallow deer is very patchy. The British Deer Society recently produced a distribution map for fallow deer, showing isolated records in northern Scotland, Islay, Aberdeenshire and the Kintyre peninsula. The method used to produce the smoothed distribution map (see Methods, Section 2.5) is likely to have removed these records, and small gaps in the distribution will not be evident. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

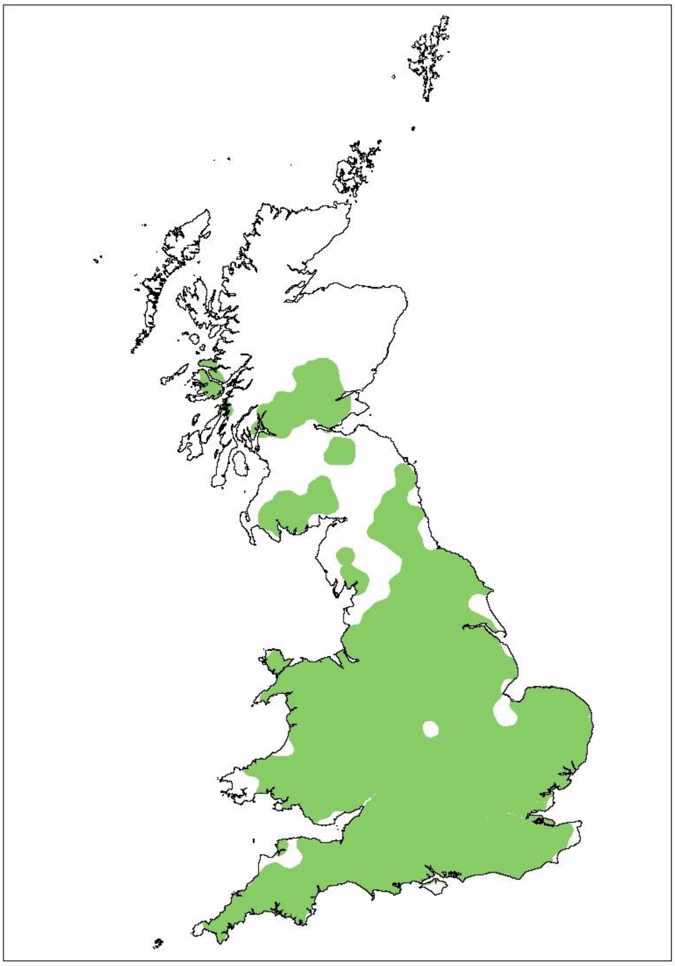


Figure 9.4a Current range of the fallow deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Fallow deer are dependent on woodland, and tend to congregate in wooded areas. Whilst they are frequently seen foraging or commuting through open areas, the core part of the range includes woodland. Population estimates based on woodland are therefore likely to represent the entire population (Jochen Langbein, *pers. comm.*). Population density estimates from Iossa et al. (2009) are taken from raw data supplied by the author. Where habitat type was listed as ‘mixed woodland’ the density estimate was applied to both broadleaved and coniferous woodland.

Percentage occupancy was taken from Gill and Morgan (2009), where 9 out of 15 surveyed woodlands (60%) contained fallow deer. Density estimates were derived from positive sites in these surveys.

Results

Four papers were identified by the literature search. Three of these reported pre-breeding density estimates and one gave post-breeding density estimates. Population density estimates are shown in Table 9.4a, and population size estimates in Table 9.4b.

Table 9.4a Median density estimates with 95% confidence intervals for fallow deer, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^2)	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	10,600	28.4	20.9	37	Thirgood (1996)	4	60%
					Iossa et al. (2009)	62	
Coniferous woodland	4,900				Gill and Morgan (2009)	9	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.4b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.4a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	11,000	188,000	138,000	245,000
Scotland	3,300	56,700	41,700	73,900
Wales	1,100	19,000	14,000	24,800
Britain	**15,500**	**264,000**	**194,000**	**343,000**

Critique

The same density values were applied to both coniferous and broadleaved woodlands, and these were the only habitat types included in the population estimates. Therefore, sensitivity analyses could not be performed.

Fallow deer have a very patchy distribution, and their density is highly variable both within and between habitats. The density estimates in published literature are likely to be derived from high density populations, rather than the averages across the range. In addition, the literature rarely states the timing or extent of any deer control in the region, so it is difficult to determine whether the estimates are representative of the population as a whole. Percentage occupancy was based on a small sample size (surveys from 15 sites). No data were available for either density or occupancy in Scotland; it is therefore impossible to determine whether the values for England and Wales are appropriate across the entire range. A population estimate of 8,000 was proposed in 2013 for Scotland (Scottish Natural Heritage, 2014), with a population size of 15,000 suggested by an expert consulted for this review (James Irvine, *pers. comm.*). These figures differ considerably from those estimated here, emphasising that further evidence is urgently required for this species. A reliability assessment is provided in Table 9.4c.

Table 9.4c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of fallow deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat	
			Coniferous woodland	Broadleaved woodland
Location of study sites	0	Estimates from one location		
	1	Estimates restricted	1	1
	2	Estimates widespread		
Sample size	0	<10 population density estimates		
	1	10-30 population density estimates		
	2	>30 population density estimates	1*	1*
Occupancy data available?	0	No		
	1	Yes	1	1
		Habitat score	**3**	**3**
		Overall reliability score	**3****	

*Although >30 estimates are available, expert opinion suggests these are likely to be based largely on high density populations, so a score of 1 has been assigned to this category.
** Reliability is likely to be lower in Scotland.

Changes through time

Comparison to Harris et al. (1995)

Population size in Great Britain was estimated by Harris et al. (1995) to be 100,000, comprised of 95,000 in England, 4,000 in Scotland and fewer than 1,000 in Wales. These estimates were based on the authors own expert opinion, following assessment of several published estimates of population size. These included non-habitat-specific density estimates that were then multiplied by the species' distribution, and estimates published by the former Red Deer Commission. It is therefore difficult to make direct comparisons with the current estimates, which are habitat-specific and indicate much larger populations for Wales and Scotland.

Other evidence of changes through time

The National Gamebag Census indicates a 45% increase (95%CI 2% decrease to 196% increase) in the number of fallow deer culled in Britain between 1995 and 2014, although this trend is not significant (GWCT, Nicholas Aebischer, *pers. comm.*) and does not account for hunting effort which may also vary over time.

Range size increased by 1.8% per year between 1972 and 2002 (Ward, 2005), followed by a further recorded increase between 2007 and 2011 (Ward et al., 2008; Scottish Natural Heritage, 2016). Most of this expansion was in England and Wales. A summary of trends in population size and range is provided in Table 9.4d.

Table 9.4d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	All countries*			
	Stable				
	Decrease				
	Data deficient				

* Increase in population size is assumed based on an increase in range size (Ward et al., 2008, Scottish Natural Heritage 2016 annex 2), although increases in both population and range are small, and trends are not certain.

Drivers of change

Table 9.4e Drivers of population change for fallow deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Management (control).	Deer are controlled to reduce the impact of grazing on plant biodiversity and associated animal biodiversity.	Trenkel et al. (1998)	Negative
Habitat availability.	Increased woodland availability (4.7% increase in broadleaved woodland and 6.4% increase in coniferous woodland between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008)	Positive

Data Deficiencies

Table 9.4f Areas where further research is required to improve the reliability of population size estimates for fallow deer.

Data deficiencies	Habitat	Details
Limited occupancy data.	All habitats	Occupancy data are based on 15 surveyed sites only.
Managed populations.	All habitats	Management is not taken into account in the population size estimate.

Future prospects

Table 9.4g An assessment of the future prospects for the fallow deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

9.5 Roe deer *Capreolus capreolus*

Habitat preferences

The roe deer is a typical browser, and feeds selectively on only the most digestible plant matter, such as leaves, flower-heads, seedlings and forbs (Latham et al., 1999). Forest habitats with richer food plant biomass such as young stands, forest rides and edges are favoured (Gill et al., 1996), but it will also utilise a wide mosaic of habitats to forage (Danilkin, 1996). It occurs at highest densities in mixed, coniferous or broadleaved woodland, and has benefited from the increase in woodland cover over the last century (see Harris and Yalden, 2008). The behaviour of the roe deer behaviour depends on the fragmentation of woodland habitats. Where woodland patches are numerous and widely distributed, populations are found within these patches; whereas when woodland is clumped and patches are distant, the species takes advantage of open areas instead, congregating in larger herds as distance from woodland increases (Hewison et al., 2001).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

A distribution map is presented in Figure 9.5a. Maps produced by the British Deer Society (2011) suggest a patchy distribution in Wales and in central and south east England. The process used to create the smoothed distribution map (Figure 9.5a) means that small distribution gaps are not evident. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

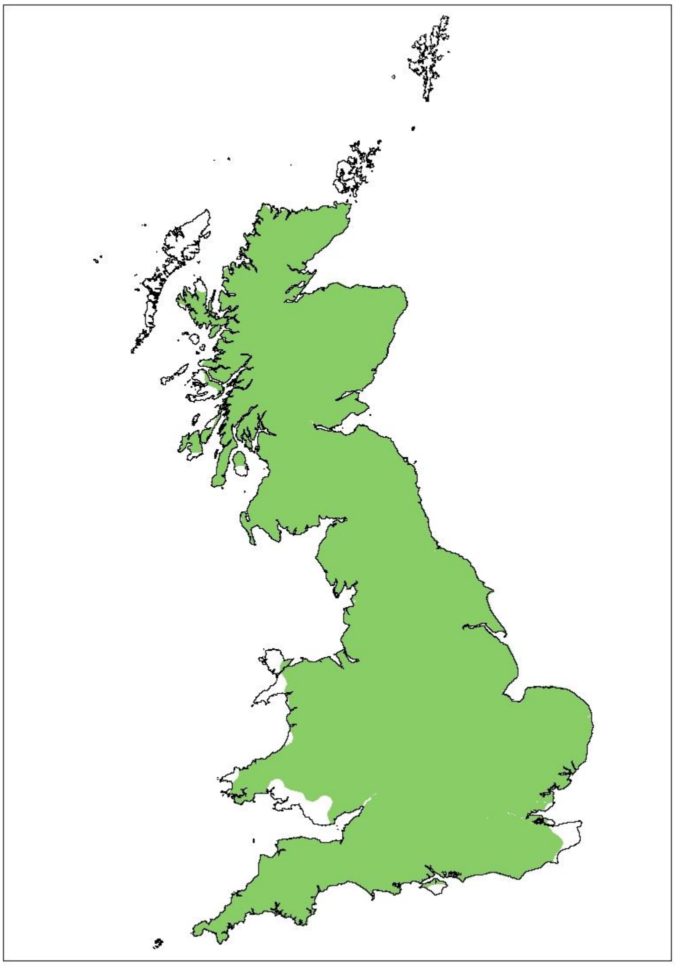


Figure 9.5a Current range of the roe deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Roe deer are dependent on woodland, using them for cover as well as foraging. Therefore, population density estimates were based on woodland — even though animals extend beyond them to forage — in order to avoid double counting.

Percentage occupancy was taken from Gill and Morgan (2009), who reported that 12 out of 15 surveyed woodlands (80%) contained roe deer. Density estimates were derived from positive sites in those surveys.

Results

Twelve papers were identified by the literature search. One paper reported the likely geographical range rather than a specific estimate, three contained a total population size without a specified area, and two gave post-breeding density estimates. Population density estimates are shown in Table 9.5a, and population size estimates in Table 9.5b.

Table 9.5a Median density estimates with 95% confidence intervals for roe deer, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^2)	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	12,700	12.3	10	13.8	McIntosh et al. (1995)	5	80%
					Ward et al. (2004)	3	
					Hemami et al. (2005)	1	
Coniferous woodland	14,200				Hemami et al. (2007)	4	
					Iossa et al. (2009)	7	
					Gill and Morgan (2009)	12	
					Waber and Dolman (2015)	15	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.5b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.5a with the area of habitat within the species' distribution and the occupancy value. Small discrepancies between this calculation and the population sizes presented are owing to rounding errors.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	12,200	120,000	97,900	135,000
Scotland	12,400	122,000	98,900	136,000
Wales	2,300	22,300	18,100	24,900
Britain	**26,900**	**265,000**	**215,000**	**296,000**

Critique

The same density values were applied to both coniferous and broadleaved woodlands, and these were the only habitat types included in the population estimates. Therefore, sensitivity analyses could not be performed.

The density estimates for woodland are based on large numbers of estimates, encompassing a range of woodland types (7 studies with 47 separate estimates). As a result, confidence intervals are relatively small. Study sites were located throughout England (Gill and Morgan, 2009; Iossa et al., 2009), in the east of England (Hemami et al., 2005; Waber and Dolman, 2015), and in North Yorkshire (Ward et al., 2004; Hemami et al., 2007). However, only a single estimate was available for Wales (Iossa et al., 2009), and none for Scotland. It is therefore likely that the population estimates for England are more robust than those for Wales or Scotland. This review assumes that population sizes can be estimated effectively on the basis of woodland habitats only, despite other habitats being used for foraging. It is possible that woodland patches surrounded by favourable resources (such as arable crops) may support higher roe deer densities than would the same size of patch in continuous woodland. If this is the case, then the computed population size will be an underestimate.

Percentage occupancy was based on surveys from 15 sites only, and may not, therefore, accurately represent occupancy throughout the species' range.

Scottish Natural Heritage reported a population size estimate of 200,000 to 350,000 for roe deer in Scotland in the 2014 Review of Scotland's Wild Deer report (Scottish Natural Heritage, 2014), which is significantly higher than our estimate. There is, however, no systematic monitoring of roe deer across different habitats, and estimating roe deer number is difficult as animals seek refuge in sheltered areas (Scottish Natural Heritage, 2016). A reliability assessment is provided in Table 9.5c.

Table 9.5c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population roe deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All woodland
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	
	1	10-30 population density estimates	
	2	>30 population density estimates	2
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**4**
		Overall reliability score	**4**

Changes through time

Comparison with Harris et al. (1995)

Harris et al. (1995) estimated the total population size in Great Britain to be 500,000, comprised of 150,000 in England, 350,000 in Scotland and approximately 50 in Wales. For Scotland, these figures are based on an assumption that the reported number of individuals culled represented 10% of the total population. This assumption was justified on the basis that the population was known to be expanding, so the cull rate had to be lower than 15% — the level that would prevent population growth (Shedden, 1993). The figures for England and Wales were inferred from their relative distribution. The current estimate uses a different methodology and is based on the observed population density in woodlands only, so it is not directly comparable.

Other evidence of changes through time

Between 1995 and 2014, there was a 31% increase across Britain in the number of roe deer culled according to the National Gamebag dataset (95%CI = 18%-54%; Nicholas Aebischer, *pers. comm.*). The trends for England and Scotland are similar to the national trend, and are highly significant. Insufficient data are available from Wales to permit assessment. These time trends are not adjusted for hunting effort, which may also vary over time.

Scottish Natural Heritage (2016) suggests that the population size of roe, sika and fallow deer in the National Forest Estate in Scotland is declining slowly, although woodland populations are difficult to measure with accuracy and individual trends for these three species are not reported. Ward (2005) suggests an increase in overall range size of 2.3% per year (based on data from 1972 and 2002), although the trend from 1995-2016 is not stated specifically, and expert opinion suggests that roe deer are now likely to be at their limit in Scotland (James Irvine, *pers. comm.*). A summary of trends in population size and range is provided in Table 9.5d.

Table 9.5d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

<table>
<tr><td></td><td></td><td colspan="4">Range</td></tr>
<tr><td></td><td></td><td>Increase</td><td>Stable</td><td>Decrease</td><td>Data deficient</td></tr>
<tr><td rowspan="4">Population size</td><td>Increase</td><td>England
Wales*</td><td></td><td></td><td></td></tr>
<tr><td>Stable</td><td></td><td>Scotland**</td><td></td><td></td></tr>
<tr><td>Decrease</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Data deficient</td><td></td><td></td><td></td><td></td></tr>
</table>

* Trends for England and Wales are based on the assumption that an increase in range is the result of an increase in population size (Ward et al., 2008, Scottish Natural Heritage 2016 annex 2). However, the computed population size is only half that reported in Harris et al. (1995), albeit using a different methodology.
**In Scotland, population is assumed to be stable as there has been no change in range since 2002 (Scottish Natural Heritage 2016 annex 2).

Drivers of change

Table 9.5e Drivers of population change for roe deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Competition with muntjac.	Ward (2005)	Negative
Vehicle collisions.	3%-7% of the population is killed annually on roads, but population consequences are unknown. Collision risk reflects the density of roads and traffic rather than deer density.	Langbein (2007)	Negative
Habitat availability.	Increased woodland availability (4.7% increase in broadleaved woodland and 6.4% increase in coniferous woodland between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008)	Positive

Data Deficiencies

Table 9.5f Areas where further research is required to improve the reliability of population size estimates for roe deer.

Data deficiencies	Habitat	Details
Limited occupancy data.	All habitats	Occupancy data are from 15 surveyed sites only.
Managed populations.	All habitats	Management is not taken into account in the population size estimate.

Future prospects

Table 9.5g An assessment of the future prospects for the roe deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Stable
Range	Stable
Habitat	Stable

9.6 Chinese water deer *Hydropotes inermis*

Habitat preferences

The Chinese water deer prefers reed beds, river shores, and woodlands with mixed vegetation. It has a very restricted geographical range (largely Cambridgeshire and Norfolk): the wet fenlands in these areas appear to offer ideal habitat, similar to its native regions. It is occasionally found in arable habitat at low densities, and relies on woody habitats for cover (see Harris and Yalden, 2008). The open parkland and downland around Whipsnade, Bedfordshire, the site of the original escape into the wild in 1929, is also used. However, body weights in that region appear lower than elsewhere, suggesting that the habitat is suboptimal. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: VU.).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

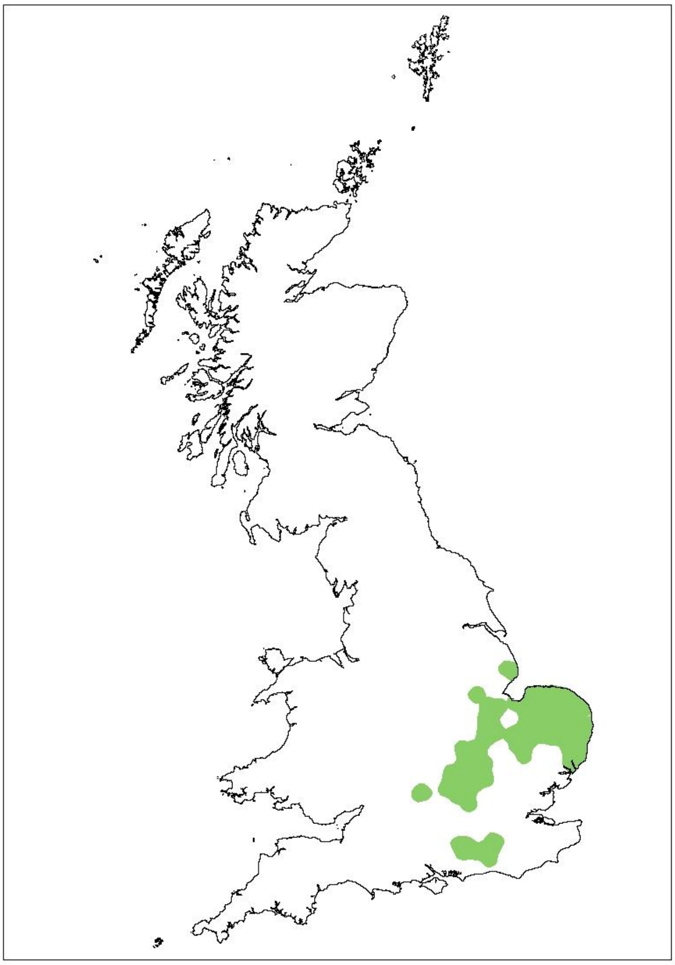


Figure 9.6a Current range of the Chinese water deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Chinese water deer have different habitat requirements from other cervidae species in Britain. They are associated with wetlands, but are also found in a variety of other habitats. Population density estimates are therefore included for habitats suggested by expert consultees.

Results

One paper was identified from the literature search for Chinese water deer. This contained data on presence and distribution but no information on population density. Population density estimates are therefore based on expert opinion only (Table 9.6a). The area of suitable habitat and population size estimates are shown in Table 9.6b.

Table 9.6a Median density estimates with plausible upper and lower range for Chinese water deer, calculated using data obtained from expert opinion consultation.

Habitat	Area within range (km^2)	Density (km^{-2})	Plausible Range Lower	Upper	Source*	n**	%Occ†
Arable and horticulture	10,600	0.1	0	10	expert opinion	1	n/a
Broadleaved woodland	1,400	0.5	0	20	expert opinion	1	n/a
Fen, marsh and swamp	40	40	1	100	expert opinion	1	n/a
Unimproved grassland	500	0.5	0	10	expert opinion	1	n/a

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.6b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.6a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	12,600	3,600	200	143,000
Britain	**12,600**	**3,600**	**200**	**143,000**

Critique

No percentage occupancy data were available, so the population size is likely to be overestimated. The two habitats contributing most to the estimated population are fen, marsh and swamp habitats (44%); and arable and horticulture (30%; Figure 9.6b). The high population density in fen, marsh and swamp accounts for this habitat's contribution to the total population; whereas in arable land, the large contribution is driven by the large area within the species' range (i.e., 84%; Figure 9.6b). As population density estimates are taken from expert opinion, a conservative score of 1 has been applied to the 'location of study sites' section in the reliability assessment (Table 9.6c). The population is thought to be relatively stable.

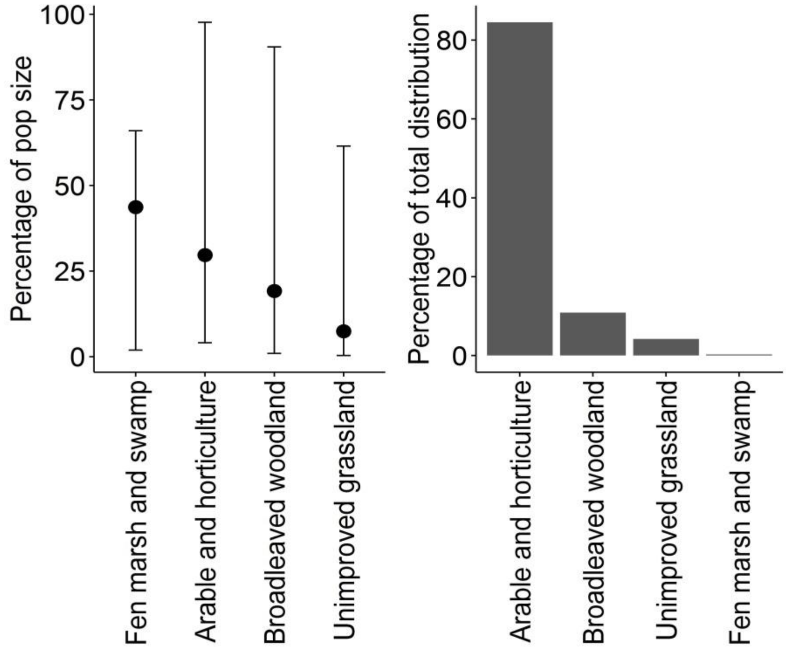


Figure 9.6b Left: The percentage of the total population of Chinese water deer accounted for by each habitat type. Error bars are derived by multiplying the lower and upper confidence limit for density by the area occupied. Right: The percentage of total area within the species' distribution represented by each habitat type.

Table 9.6c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of Chinese water deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	**Score**	**Details**	**Habitat**
			All habitats
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 population density estimates	0
	1	10-30 population density estimates	
	2	>30 population density estimates	
Occupancy data available?	0	No	0
	1	Yes	
		Total score	1
		Overall reliability score	1

* Populations may be unstable owing to inter-annual cycles or documented fluctuations in population size, or as a result of management.

Changes through time

Comparison to Harris et al. (1995)

The population size in Great Britain was estimated by Harris et al. (1995) as 650, all in England. This estimate was based on total counts in areas of known deer populations. Although a different method was used to estimate the current population size, the estimate for Harris et al. (1995) was considered reliable as the population was still relatively restricted and total counts were possible.

Nationally, there are changes between the two reviews in the estimated availability of key habitats, generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape, by using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008), produces a population size estimate that falls within the confidence limits of the original. These methodological issues are therefore unlikely to have a material impact on comparisons between the two reviews.

A previous estimate for Chinese water deer resulted in a population size of 7,000 in 2010 (Cooke, 2011). This estimate was based on the mean number of deer records per occupied tetrad within an extensively surveyed area; this density estimate was then applied to the number of positive hectads in the British Deer Society range map. As a habitat-based approach was not used, a comparison with the current estimate is not possible, but this estimate would indicate an increase in population size since the report by Harris et al (1995).

There has been an increase in the range of Chinese water deer since 1986 (Arnold, 1993), but it is unclear whether the range had stabilised by 1995. The population estimate from Harris et al. (1995) falls towards the lower confidence interval of the new estimate, but these confidence intervals are very wide and reflect the uncertainty of the current estimate.

Other evidence of changes through time

Ward (2005) found an increase in range size of 2% per year between 1972 and 2002. Trends between 1995 and 2016 are less certain.

Table 9.6d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	England			
	Stable				
	Decrease				
	Data deficient				

* Based on the population size in 2010 (Cooke, 2011).

Drivers of change

Table 9.6e Drivers of population change for Chinese water deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Continued expansion into suitable habitat.	Ward (2005)	Positive
Habitat quality.	Changes in land management.	Arnold Cooke (*pers. comm.*)	Positive

Data deficiencies

Table 9.6f Areas where further research is required to improve the reliability of population size estimates for Chinese water deer.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	All habitats	No recent population density estimates are available in the literature
No occupancy data.	All habitats	No recent occupancy data are available in the literature.

Future prospects

Table 9.6g An assessment of the future prospects for the Chinese water deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

9.7 Reeves' muntjac deer *Muntiacus reevesi*

Habitat preferences

The Reeves' muntjac deer primarily select habitats with dense cover and a diverse understory, such as broadleaved, mixed and coniferous woodland and scrub (see Chapman et al., 1994), although grasslands and arable fields, especially those with hedgerows, are also used. Habitat preferences are, however, challenging to assess, because populations formed by colonising dispersers cannot readily be distinguished from those founded by animals that have been deliberately released (Chapman et al., 1994).

The Reeves' muntjac occupies the same broad habitat types as native roe deer, and interspecific competition exists between the two. Nevertheless, there are some differences in local scale habitat preferences. Reeves' muntjac is found in higher densities among older woodland stands and areas of bramble, and shows a greater degree of habitat selection than roe deer (Hemami et al., 2005).

Status

Non-native.

Conservation Status

- IUCN Red List (GB: n/a; England: n/a; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

There is no known wild population in Scotland. Sightings have been reported and followed up, but no evidence of Reeves' muntjac has been found. An established population is therefore unlikely. Reeves' muntjac populations are patchily distributed across the rest of their range. The process used to create the smoothed distribution map (Figure 9.7a) means that small distribution gaps are not evident. For a comparison, the British Deer Society map can be found at https://www.bds.org.uk/index.php/research/deer-distribution-survey.

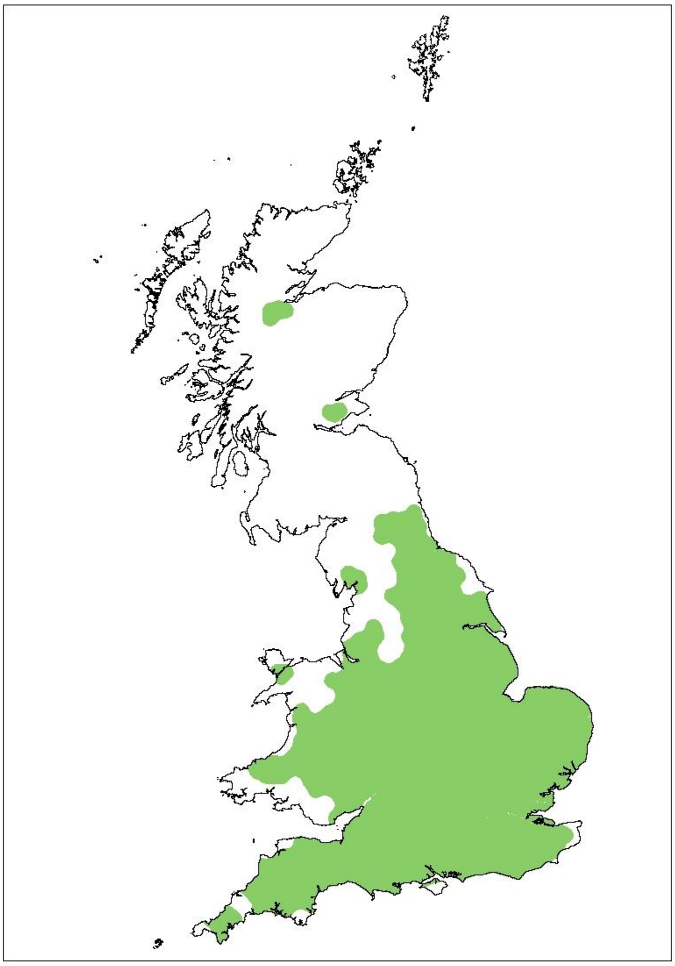


Figure 9.7a Current range of the Reeves' muntjac deer in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

There is a strong consensus that an established population is not present in Scotland, so population size estimates are limited to England and Wales.

It is assumed that woodland forms a core part of the home range for Reeves' muntjac. This assumption may introduce a larger source of error than for other deer, because the species can be found in other habitats such as suburban gardens, small patches of scrub and brownfield sites. However, no additional information was available to enable any account to be taken of these habitats.

Percentage occupancy was drawn from Gill and Morgan (2009), where 7 out of 15 surveyed woodlands (46.7%) contained Reeves' muntjac deer. Density estimates were derived from positive sites in those surveys.

Results

Seven papers were identified by the literature review. Three papers reported pre-breeding density estimates, two contained presence-only data, one gave a total count without a defined area, and one provided details of temporal trends in relative population size. Population density estimates are shown in Table 9.7a, and population size estimates in Table 9.7b.

Table 9.7a Median density estimates with 95% confidence intervals for Reeves' muntjac deer, calculated using data obtained from a review of the literature from 1995 to 2015.

Habitat	Area within range (km^2)	Density (km^{-2})	-95%CI	+95%CI	Source*	n**	%Occ†
Broadleaved woodland	9,100	22.9	20.7	26.3	Hemami et al. (2007)	4	46.7%
					Gill and Morgan (2009)	7	
Coniferous woodland	2,900				Waber and Dolman (2015)	19	

* Literature sources.
** Number of estimates from each literature source.
† Percentage of this habitat that is occupied within the known range.

Table 9.7b Area of suitable habitat (not adjusted for occupancy) within the species' range, and total population size estimates with 95% confidence intervals. Values were obtained by multiplying population density estimates in Table 9.7a with the area of habitat within the species' distribution.

Country	Area of suitable habitat (km^2)	Population size	-95%CI	+95%CI
England	10,500	112,000	100,000	128,000
Wales	1,500	16,300	14,800	18,700
Britain	**12,000**	**128,000**	**115,000**	**147,000**

Critique

The same density values were applied to both coniferous and broadleaved woodlands, and these were the only habitat types included in the population estimates. Therefore, sensitivity analyses could not be performed. The percentage occupancy value was based on surveys from 15 sites only, so it may not accurately represent occupancy throughout the species' range. In addition, although numerous studies of population density were available, only one surveyed both coniferous and broadleaved woodland (Gill and Morgan, 2009); the other references (Hemami et al., 2007; Waber and Dolman, 2015) used estimates from coniferous woodland but applied them to both woodland habitats. As a consequence, it is possible that the population density for broadleaved woodland is inaccurate. The assumption that density estimates derived from woodland represent the entire population may also be less sound for Reeves' muntjac than for other deer species, as it is known that animals can occupy small patches of rough vegetation, hedgerows and ditches away from woodland. A reliability assessment is provided in Table 9.7c.

Table 9.7c Reliability assessment for each habitat representing >25% of the species' distribution, or accounting for >25% of the overall population of Reeves' muntjac deer. Each habitat received a score based on the number of locations in which density was measured, the number of density estimates contributing to the median, and availability of occupancy data. These scores are summed to give a total score per habitat.

Measure	Score	Details	Habitat
			All woodlands
Location of study sites	0	Estimates from one location	
	1	Estimates restricted	1
	2	Estimates widespread	
Sample size	0	<10 density estimates	
	1	10-30 density estimates	
	2	>30 density estimates	2
Occupancy data available?	0	No	
	1	Yes	1
		Habitat score	**4**
		Overall reliability score	**4**

Changes through time

Comparison to Harris et al. (1995)

Population size in Great Britain was estimated by Harris et al. (1995) as approximately 40,000, comprised of 40,000 in England, fewer than 50 in Scotland and fewer than 250 in Wales. These figures were derived from a reported density of 30km^{-2} in optimal habitat, and an assumed density of 15km^{-2} in sub-optimal habitat. The density estimates were applied to counties that were ranked among the top 50% of those contributing records for the species. The resulting figure was then doubled to account for patchy populations of deer elsewhere.

Nationally, there are changes between the two reviews in the estimated availability of key habitats generated by a combination of true change and methodological differences, irrespective of any range change (see Sections 2.3 and 32.3 for further details). The adjusting of results to reflect more probable temporal changes in the composition of the British landscape, using differences between the 1990 and 2007 Countryside Surveys (Carey et al., 2008), produces an 11% decrease in population size to 113,000. This estimate is outside the confidence limits of the original, but is still substantially greater than the estimate in Harris et al. (1995). While some of this difference is accounted for by range expansion, differences in other assumptions make the total population estimates difficult to compare.

Other evidence of changes through time

The National Gamebag dataset reports a 219% increase (95%CI = 152%-325%) in the numbers of Reeves' muntjac culled between 1995 and 2015 (Nichola Aebischer, *pers. comm).* However, these trends do not adjust for hunting effort, which may also vary over time. Based on a comparison of data from 1972 and 2002, Ward (2005) found a net increase in range size of 8% per year. A further increase was recorded between 2007 and 2011 (Ward et al., 2008). A summary of trends in population size and range is provided in Table 9.7d.

Table 9.7d Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	England Wales*			
	Stable				
	Decrease				
	Data deficient				

* Increase in population size is assumed based on an increase in range (Ward et al., 2008).

Drivers of change

Table 9.7e Drivers of population change for Reeves' muntjac deer between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Species introduction.	Continued expansion into suitable habitat.		Positive
Climate change.	Mild winters permit population Growth.	(Pickvance and Chard, 1960)	Positive
Habitat availability.	Increased woodland availability (4.7% increase in broadleaved woodland and 6.4% increase in coniferous woodland between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008)	Positive
Vehicle collisions.	25% of deer collisions between 2003-2005 in England were with Reeves' muntjac.	Langbein (2007)	Negative

Data deficiencies

Table 9.7f Areas where further research is required to improve the reliability of population size estimates for Reeves' muntjac deer.

Data deficiencies	Habitat	Details
No density estimates for specified habitat.	Broadleaved woodland and non-woodland habitats	No recent population density estimates are available in the literature.
Managed populations.	All habitats	Management is not taken into account in the population size estimate.
Limited occupancy data.	All habitats	Occupancy data are based on 15 surveyed sites only.

Future prospects

Table 9.7g An assessment of the future prospects for the Reeves' muntjac deer, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

10 CHIROPTERA

10.1 Greater horseshoe bat *Rhinolophus ferrumequinum*

Habitat preferences

The greater horseshoe bat forages in open areas such as pasture and parkland, preferring landscapes with numerous large trees, tall hedgerows and woodland patches. These offer shelter, accumulations of insect prey, and opportunities for perch-hunting. The species is highly dependent on pasture grazed by livestock, particularly cattle (Ransome, 1996). There is concern about the potential impact of agricultural intensification, conversion to arable production, and the use of avermectins (antiparasitic agents) on dung fauna, since dung beetles and other Coleoptera form a high proportion of the diet during the breeding season (Duverge and Jones, 1994). At other times of year, Lepidoptera (moths), Tipulids (crane flies), and other species comprise varying proportions of the diet.

Traditionally cave-dwellers, in Britain the greater horseshoe bat now tends to roost in buildings during the summer. Warmer roost conditions are linked with improved breeding success (Ransome, 1998), and roost modifications to improve thermal gain have resulted in substantial increases in some key maternity roosts in the south west of England. Cave sites and other underground locations are used for hibernation, and may contribute to the limited distribution of the species.

Mating roosts are usually situated in underground sites such as cellars, tunnels and small caves, which are defended by solitary males. Occasionally, such roosts may have two males, but only when divided into separately defensible areas (Fiona Mathews, *pers. obs.*). The males may be present from spring until autumn, and may even stay throughout the winter. In late summer and autumn, groups of related females visit these sites to mate (Rossiter et al., 2000; Rossiter et al., 2005). Even at this time, fewer than 7 bats are usually present at once. However, ringing records show that over the course of the mating season, large numbers of females can pass through the sites, often visiting a series of males (Fiona Mathews, *pers. obs.*; Ransome and Hutson 2000). Genetic analysis has shown that females are likely to mate with the same male in a series of years (Rossiter et al., 2005). One male can mate with multiple females, whereas others may achieve no reproductive success. Nevertheless, many mating sites will be required by the population. It is therefore of considerable concern that very few mating sites are known. As maternity colonies are virtually closed, they are the locations at which gene flow in the population occurs (Rossiter et al., 2000). Moreover, outbreeding is the main predictor of adult survival and reproductive success, and is more important than more conventionally measured parameters such as mass or arm length (Rossiter et al., 2001).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England:[LC]; Scotland: n/a; Wales: [NT]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex II and IV; UK: Favourable; England: Favourable; Scotland: n/a; Wales: Favourable).

Species' distribution

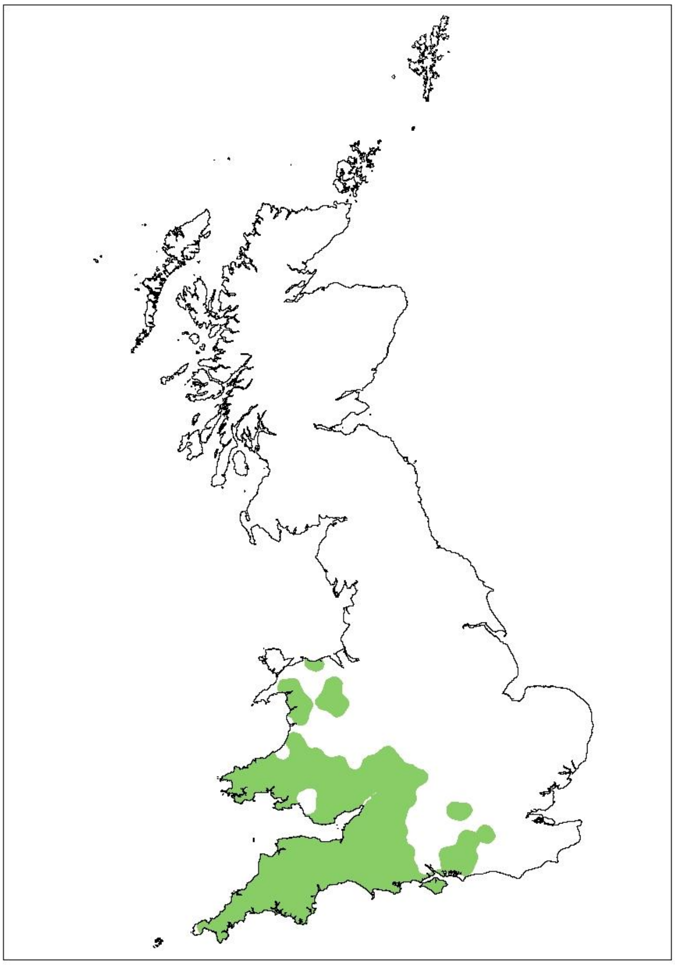


Figure 10.1a Current range of the greater horseshoe bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

It is assumed that most maternity roosts of this species are known. This assumption is based on the fact that most maternity roosts are in buildings where the bats are conspicuous, and that intensive conservation efforts have been undertaken by both Statutory Nature Conservation Bodies and Non-Governmental Organisations over the last 25 years. The population size calculations are therefore based on direct counts of individuals, together with estimates of maternity colony sex ratio, and do not use inferences from habitat associations.

Total numbers of individuals at known greater horseshoe maternity roosts (pre-breeding) were counted. The peak count in the most recently available year was used, and 57 sites were included (among them, 26 sites monitored as part of the NBMP (Bat Conservation

Trust, 2016)). The peak count pre-breeding in the latest available year was used for the analyses, unless the count had fewer than 30 individuals, in which case the next available year when the count was ≥30 was used. If no counts with ≥30 bats were available, then the site was excluded on the grounds that it was unlikely to be used as a maternity roost for this species. After excluding small roosts, the estimates were based on 33 sites. Estimates of the total population size, and upper and lower plausible intervals (PIs), were then derived by adjusting for the sex ratio in the maternity sites pre-breeding.

Expert opinions were provided by 7 individuals. Only one of these was able to estimate the colony sex ratio (70% female). This value corresponds to the 70%-75% figure, derived from a different expert opinion, used in the JNCC Article 17 Reports (2013) for England and Wales (Joint Nature Conservation Committee, 2013b) to provide minimum and maximum population estimates.

The estimate provided here follows the expert opinion that 70% of the individuals in maternity colonies are female. The lower plausible interval (PI) uses a conservative assumption of 50% females, meaning that the entire population is counted at maternity sites; whereas the upper plausible interval assumes that the maternity site contains only females, so the true population is double the number of animals observed at the maternity sites. It has been assumed that there are equal numbers of male and female bats in the population overall, given the lack of any contrary evidence in the literature or from expert opinion.

Habitable area was defined as all area within the range. Given that the species uses a mosaic of habitats, and usage of one habitat depends on the configuration and extent of other habitats, more precise definition of suitable habitat was not possible for this review.

Results

The median number of bats per roost was 50 (95%CI = 20-147). If only roosts with ≥30 bats were included, which is considered a much more realistic approach for maternity colonies of this species, the median was 162 (95%CI = 125-211). Unlike for the non-horseshoe bat species, these median values were not used in estimating the population size because better estimates were available from direct counts.

Table 10.1a Total population size estimates, with plausible intervals, for England, Scotland, Wales, and the whole of Britain.

Country	No. roosts	Observed individuals	Population size	Plausible intervals	
				Lower	Upper
England	28	7,270	10,200	7,280	14,600
Scotland	0	0	0	0	0
Wales	5	1,930	2,700	1,930	3,850
Britain	**33**	**9,200**	**12,900**	**9,210**	**18,500**

The estimates proved relatively insensitive to the removal of counts with <30 bats. When the analyses were repeated using the latest available peak count obtained prior to July, regardless of size (which increased the number of sites to 57), the population estimate for Britain was 13,200 (PIs = 9,400-18,900).

These estimates are compatible with the Article 17 Report on greater horseshoe bat status 2007-2012 for Wales, but are slightly higher than the estimates for England (Table 10.1b; (Joint Nature Conservation Committee, 2013b)).

Table 10.1b Article 17 Report on greater horseshoe bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	4,750	7,120
Scotland	0	0
Wales	1,480	2,220
Britain	**6,230**	**9,340**

The current geographical range of the species, based on known records since 1995, is shown in Table 10.1c.

Table 10.1c Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	29,600	n/a
Scotland	0	0
Wales	13,200	n/a
Britain	**42,800**	**55,100**

Critique

The greater horseshoe bat is one of the most intensively studied species in the UK. Although originally a cave-breeding species, it is now highly dependent on buildings, with only a small number of maternity roosts being found in underground sites. Because of this close dependency on people, the size of the maternity colonies, and the visibility of the bats when roosting, it is likely that a high proportion of its colonies are known. This does not necessarily imply that all roost owners choose to share information with biological recording centres, so concerted efforts are still being made to identify new maternity sites within the main strongholds. Nevertheless, confidence in the estimate is high.

Comparison with expert opinion

Seven experts provided their opinion on this species, but only one was able to give an overall population estimate. This estimate of 7,500 animals for Britain (PIs = 6,500-9,000) is lower than that calculated here.

Table 10.1d Reliability assessment for the greater horseshoe bat. Scores are based on the availability of roost location data, roost count data and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost location data	0	Low proportion of roosts considered to be known (<25%)	
	1	Moderate proportion of roosts considered to be known (25-75%)	
	2	Most roosts considered to be known (>75%)	2
Roost count data availability	0	Low proportion of known roosts (<25%)	
	1	Moderate proportion of known roosts (25-75%)	
	2	High proportion of known roosts (>75%)	2
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**4**

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Harris et al. (1995) estimated a pre-breeding population in Great Britain of at least 4,000, and possibly nearer 6,600, comprised of approximately 3,650 in England and 350 in Wales. These figures are likely to have been an underestimate, as some maternity roosts, each containing several hundred individuals, have been discovered since the publication of the 1995 report. Harris et al. (1995) incorporated counts of hibernacula into expert opinion estimates, whereas the current assessment is based on maternity sites only.

The range is similar to that described by Arnold (1993), except with an expansion into mid and north Wales. The Article 17 Report concluded that the range appeared to be stable (Joint Nature Conservation Committee, 2013b), with slight changes being mainly the result of better data rather than a true range shift. However there have been sightings of the species in mid and north Wales since the early 1990s, including in sites that had been systematically monitored — without yielding any records of greater horseshoe bats — since at least the early 1980s. The discovery of small numbers of greater horseshoes breeding in the Tanat Valley and areas of Herefordshire over the past decade also suggests a recent northward expansion of the population.

Other evidence of changes through time

The species has undoubtedly undergone a severe contraction of its range and population size over the last 100 years. The concentration of females into large maternity sites makes the population vulnerable, and incidents of fire or pesticide use have historically resulted in the loss of several hundred individuals at single locations. Nevertheless, there is long-standing debate about the extent of historical population collapses (see Harris et al., 1995).

The National Bat Monitoring Programme (NBMP) includes 32 maternity sites that have been surveyed between 1997 and 2015. The index is now 126% higher than the base-level established in 1999, and the increases have been consistent throughout the monitoring period. However, a small number of sites where colony sizes have increased dramatically contribute a high proportion of the total monitored population (notably those in south Devon, which includes the largest known roost in a building in central and western Europe). Elsewhere, there is concern for smaller colonies, which appear particularly vulnerable to the impact of adverse weather conditions on reproductive output.

Hibernation data since 1990 are also collated from 231 sites as part of the National Bat Monitoring Programme. The Article 17 Report (Joint Nature Conservation Committee, 2013b) used the hibernation data as their primary index of population trends. There have been increases in both hibernation and maternity roost counts, although the increase hibernation counts appears to be plateauing in recent years.

If the 4.8% annual increase observed for Great Britain in the hibernation counts were applied to a starting population of 5,300 animals (the mid-point of the range given in Harris et al. (1995)), then a current population of 13,536 bats would be expected. This is very close to the estimate of population size made here.

Table 10.1e Population trends for the greater horseshoe bat from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Summer sites were not available from which to measure a population trend in Wales. Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Hibernation**	**91**	**1997**	**124.5***	**5.2**
	Summer	32	1997	102.7*	4.5
Wales	**Hibernation**	**175**	**1990**	**77.9***	**3.7**
	Summer	n/a	n/a	n/a	n/a
Britain	**Hibernation**	**231**	**1990**	**113***	**4.8**
	Summer	32	1997	126*	5.2

* Indicates that the trend is significant ($p<0.05$).
† Percentage change since the 1999 baseline.

Table 10.1f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	All countries			
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 10.1g Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Agricultural intensification/decline of pastoral farming/use of anthelmintics.	Reduction in prey availability.	Ransome (1996) McCracken (1993)	Negative
Inbreeding.	Loss of mating roosts.	Rossiter et al. (2001)	Negative
Climate change and weather fluctuations.	Mild winters permit population growth.	Ransome (1989)	Positive
Vehicle collisions.	Low-flying species, so likely to be vulnerable to collision.	Fensome and Mathews (2016)	Negative
Artificial night lighting.	Species is extremely light-shy. Lighting potentially severs commuting routes and delays emergence time.	Jones and Rydell (1994) Stone et al. (2009)	Negative
Protection and improvement of maternity roosts.	Legislative protection of maternity roosts in particular, to prevent destruction and disturbance. Interventions to provide better thermal conditions, improving reproductive success.	Ransome (1998)	Positive
Disturbance of hibernation roosts.	Legislative protection has improved gating of underground sites. However, in some areas there are increases in recreational activity and other kinds of exploitation of underground sites.		Positive/ Negative

Data deficiencies

Table 10.1h Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Effects of cumulative pressures of land use change, lighting, etc., on local populations.	Pastoral	No data available.
Extent of loss of formation roosts, mating roosts, night roosts, and hibernation sites. Impact of such losses on the population structure and stability.	n/a	Underground sites (used as hibernation, mating, and night roosts) are very vulnerable to disturbance. This is owing to increased recreational use, and severe habitat fragmentation in some areas (e.g., in Wiltshire and Avon, many sites are disused stone quarries in urban areas). In recent years, very mild winters are likely to have reduced dependency on underground sites, with animals spending longer periods in summer roosts. However, predicted increases in extreme weather make it likely that the loss of these sites will be very important to populations. Night roosts are frequently unrecognised and may be poorly protected by current legislation and/or lost inadvertently.
Road casualty rates and impact on local populations.		No data available.
Effectiveness of current planning and licensing systems in securing the viability of SAC site populations through the protection of commuting and foraging areas.		No data available.

Future prospects

Table 10.1i An assessment of the future prospects for the greater horseshoe bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

10.2 Lesser horseshoe bat *Rhinolophus hipposideros*

Habitat preferences

The lesser horseshoe bat forages largely in broadleaved woodland and in wooded riparian corridors, as well as along mature treelines and hedgerows. Here it feeds within or below the canopy, taking small flying insects including Diptera (flies including midges, gnats and dung flies), Tipulids (crane flies) and Lepidoptera (moths). Semi- or unimproved wet pasture bounded by hedgerows is used as the main foraging area for one of the largest European colonies at Glynllifon in Gwynedd (Billington and Rawlingson, 2006). Most activity occurs within a 2.5km radius of its day roost in summer (Bontadina et al., 2002), often within 600m (Boye and Dietz, 2005), and within 1.2km of its hibernaculum in winter (Williams, 2001).

The lesser horseshoe bat has specific roosting requirements, favouring undisturbed sites with large entrances that permit uninterrupted flight into the roost. Old buildings, particularly those with slate roofs, tend to be used in the summer, and underground sites including caves, quarries and cellars are used in the winter. Night roosts appear fundamental to the conservation of the species, particularly during pregnancy and lactation (Schofield, 1996; Knight and Jones, 2009). Whilst occasional long-distance movements are known, hibernation sites are normally situated within 5km of the summer roost (the maximum known distance being 32km). Feeding areas and alternative roosts are accessed by flying in close proximity to mature hedgerows and treelines; for this reason, the lesser horseshoe bat requires a mosaic of habitats. Their stringent requirements, in terms of roosting, foraging and commuting habitats, are likely to restrict the distribution of the species across Great Britain.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: n/a; Wales: [LC]: Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex II and IV; UK: Favourable; England: Favourable; Scotland: n/a; Wales: Favourable).

Species' distribution

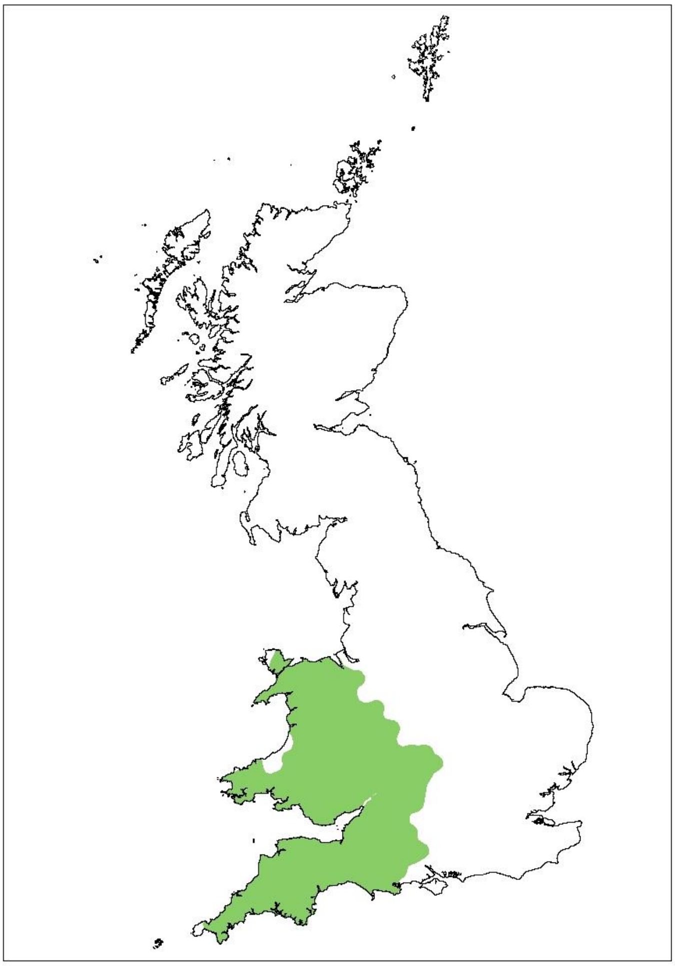


Figure 10.2a Current range of the lesser horseshoe bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

It is assumed that most maternity roosts of this species are known. The assumption is based on the fact that maternity roosts are in buildings, and that intensive conservation efforts have been undertaken by both Statutory Nature Conservation Bodies and Non-Governmental Organisations over the last 25 years. The population size calculations are therefore based on direct observations and estimates of colony sex ratio, and do not use inferences from habitat associations.

The numbers of bats at 312 known lesser horseshoe maternity roosts (pre-breeding) were counted. The peak count in the most recently available year was used for the analyses unless the count had fewer than 30 individuals, in which case the next available year where the count was ≥30 was used. If no counts with >30 bats were available, then the roost was excluded on the grounds that it was unlikely to be used for breeding, and its inclusion would therefore risk double counting the same individuals when they moved to a maternity site. Excluding these small roosts gave a sample size of 260 sites. Estimates of the total population size, together with upper and lower plausible intervals (PIs), were then derived by adjusting for the sex ratio in the maternity sites pre-breeding.

Expert opinions were provided by 8 individuals. Four of these respondents provided information on colony sex ratios. These values were: 70%; 70%-90%; 90%; and >90% female. The estimate of 70% female corresponds with the assumption used in the JNCC Article 17 Report (Joint Nature Conservation Committee, 2013b), and this is adopted in the current review as the basis for computing population size. The lower plausible interval is based on a conservative assumption of 50% females, which would mean that the entire population is counted at maternity sites; whereas the upper interval assumes that the maternity site contains only females, so the true population is double the number observed at maternity sites. It has been assumed that there are equal numbers of male and female bats in the population overall, given the lack of any contrary evidence in the literature or from expert opinion.

Habitable area was defined as all area within the range. Given that the species uses a mosaic of habitats, and the importance of one habitat depends on the configuration and extent of other habitats, more precise definition of suitable habitat was not possible for this review.

Results

The median number of bats per roost is 98 (95%CI = 88-109) based on the 261 sites where colony counts included ≥30 animals. If all sites were included, rather than just those with ≥30 individuals, the median number of bats per roost would be 82 (95%CI = 71-91). Unlike for the non-horseshoe bat species, these median values were not used in estimating the population size because better estimates were available from direct counts.

Table 10.2a Population size estimates, with plausible intervals, for England, Scotland, Wales, and the whole of Britain.

Country	No. roosts	Observed individuals	Population estimate	Plausible interval	
				Lower	Upper
England	147	22,100	19,400	13,900	27,700
Scotland	0	0	0	0	0
Wales	114	13,900	30,900	22,000	44,100
Britain	**261**	**36,000**	**50,400**	**36,000**	**72,000**

The estimates proved relatively insensitive to the removal of counts with <30 bats when computing the median number of bats per roost. When the analyses were repeated using the latest available peak count obtained prior to July, regardless of size (thereby increasing the number of sites to 312), the population estimate was 50,000 (PIs = 35,300-70,600).

These estimates are compatible with the Article 17 Report on lesser horseshoe status 2007-2012 for Great Britain, but are slightly lower than the estimates for England and slightly higher than the estimates for Wales (Table 10.2b; (Joint Nature Conservation Committee, 2013b)).

Table 10.2b Article 17 Report on the lesser horseshoe bat population sizes 2007-2012.

Country	Minimum	Maximum
England	21,500	23,400
Scotland	0	0
Wales	26,600	28,500
Britain	**48,100**	**51,900**

The current distribution of the species, based on known records since 1995, is shown in Table 10.2c.

Table 10.2c Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	33,500	n/a
Scotland	0	0
Wales	19,500	n/a
Britain	**53,000**	**61,500**

Critique

The lesser horseshoe bat is one of the most intensively studied bat species in the UK, second only to the greater horseshoe bat. Its maternity sites are well recorded because it is highly dependent on buildings, it is easily visible when roosting, and maternity colonies frequently contain large numbers of individuals (c.30-500 animals). Long-term monitoring has been conducted at many maternity roosts, and all those in this report were included in the National Bat Monitoring Programme. Although hibernacula are also monitored, the numbers recorded in known maternity roosts frequently far exceed those observed in hibernacula. This is likely to be because many hibernacula contain only small numbers of individuals (<5; Williams, 2001) and so are not monitored routinely. Hibernation data have therefore not been used in this report to generate population estimates.

Eight experts provided their opinion on this species, but only one was able to give an overall population estimate (55,000 animals for Great Britain), and this was close to the value derived above.

Two main sources of error are identified. Firstly, estimates are derived from observed numbers of bats at 260 maternity sites: it is highly likely that there are unrecorded roosts, which would mean that the population size is underestimated. This is probably a more significant issue than for greater horseshoe bats because the species is more widespread. Secondly, little information is available on the sex ratio within maternity colonies pre-breeding. The overall estimate is based on a single expert opinion of 70% of the colony being female, with other experts indicating that they had no additional directly measured data. Unpublished data from recent research conducted using genotyping at 6 roosts in the Republic of Ireland indicate that the proportion of adult males within a colony ranges from 7% to 72% (median 37%) (Andrew Harrington and Catherine O'Reilly, *pers. comm.*). This means that the median proportion of females would be expected to be 63% (range 28% to 93%). If applicable in Great Britain, this figure would reduce the estimated size of the population. Recent genotyping work at 19 colonies northern France also indicates the presence of significant numbers of adult males within pre-breeding colonies, but in that study the median value was 25.8%, with only 5 sites having values greater than the expert opinion used in the current review (Zarzoso-Lacoste et al., 2017). One of these was a large colony with >200 individuals, which implies that it is not just small or suboptimal colonies that may have large proportions of males. Given the large effect on the total population size, further research is therefore urgently required to examine this issue in Great Britain.

Table 10.2d Reliability assessment for the lesser horseshoe bat. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost location data	0	Low proportion of roosts considered to be known (<25%)	
	1	Moderate proportion of roosts considered to be known (25-75%)	
	2	Most roosts considered to be known (>75%)	2
Roost count data availability	0	Low proportion of known roosts (<25%)	
	1	Moderate proportion of known roosts (25-75%)	1
	2	High proportion of known roosts (>75%)	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**3**

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Harris et al. (1995) estimated a total pre-breeding population of 14,000 individuals. Of these, it was thought that there were 7,000 in England, 7,000 in Wales and none in Scotland. The previous report incorporated counts of hibernacula into expert opinion estimates in areas where no breeding colonies were known, whereas the current assessment is based on maternity sites only. Mitchell-Jones (*op. cit.* Harris et al., 1995) made an estimate based on peak numbers of bats recorded England (381 sites) and Wales (273 sites) since 1981, and derived an estimate of 6,947 in England and 6,747 in Wales.

The geographical range of this species appears similar to previous estimates, although there appear to be increasing numbers of records of hibernating individuals in the north of England and the Midlands.

Other evidence of changes through time

The association of the species with woodland means that, over historical time, it is likely to have declined in abundance and/or suffered range contraction (Yalden, 1992). In the early 20th century, it was reported as being abundant in some localities, but being not common (Thorburn, 1920).

The National Bat Monitoring Programme (NBMP) includes 289 maternity sites and 308 hibernation roosts that have been monitored since 1990 and 1993 respectively. Increases have been seen in both the maternity and hibernation indices (see Table 10.2e), and these changes have been consistent throughout the monitoring period. The Article 17 Report (Joint Nature Conservation Committee, 2013b) notes the increase in maternity colony counts over time.

If the 3.6% annual increase observed at maternity sites in the NBMP is applied for the 20 years since the previous estimate of 14,000 individuals (Harris et al., 1995), then the expected population would be 28,400. This is somewhat below the lower limit of the current estimate. The difference is likely to be largely owing to the discovery of new roosts since the 1995 review.

Table 10.2e Population trends for the lesser horseshoe bat from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Hibernation**	**133**	**1997**	**104.4***	**4.6**
	Summer	118	1995	110.6*	4.8
Wales	**Hibernation**	**175**	**1990**	**146.6***	**5.8**
	Summer	171	1993	64.7*	3.2
Britain	**Hibernation**	**308**	**1990**	**138.1***	**5.6**
	Summer	289	1993	76.2*	3.6

* Indicates that the trend is significant ($p<0.05$).
† Percentage changes since the baseline year 1999.

Table 10.2f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**	England Wales			
	Stable				
	Decrease				
	Data deficient				

Drivers of change

Table 10.2g Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Agricultural intensification/decline of pastoral farming/use of anthelminthics.	Reduction in prey availability.	Ransome (1996) McCracken (1993)	Negative
Climate change.	Mild winters permit population growth.	Ransome (1989)	Positive
Vehicle collisions.	Low-flying species, likely to be vulnerable to collision.	Fensome and Mathews (2016)	Negative
Artificial night lighting.	Species is extremely light-shy. Lighting potentially severs commuting routes and delays emergence time.	Jones and Rydell (1994) Stone et al. (2009)	Negative
Protection and improvement of maternity roosts.	Legislative protection of maternity roosts in particular, to prevent destruction and disturbance. Interventions to improve thermal conditions, increasing reproductive success.	Schofield and Barker (2008)	Positive
Disturbance of hibernation roosts.	Legislative protection has improved gating of underground sites. However, in some areas there are increases in recreational activity and other kinds of exploitation of underground sites.		Positive/ Negative

Data deficiencies

Table 10.2h Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Sex ratio of adults in maternity colonies pre-breeding.	n/a	Based on 1 expert opinion (same constraint applies to estimate provided in JNCC Article 17 Report).
Effects of cumulative pressures of land use change, lighting, etc., on local populations.	Pastoral	No data available.
Extent of loss of formation roosts, mating roosts and night roosts, and the impact of such losses on population structure and stability.	n/a	No data available.
Road casualty rates, and the impact on local populations.		No data available.
Effectiveness of current planning and licensing systems in securing the viability of SAC site populations through the protection of commuting and foraging areas.		No data available.
Impact of an increased woodland area and changes in management over the past 20 years.	Woodland	No data available.

Future prospects

Table 10.2i An assessment of the future prospects for the lesser horseshoe bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Increase
Range	Increase
Habitat	Stable

10.3 **Alcathoe bat** *Myotis alcathoe*

Introductory note

The Alcathoe bat (*Myotis alcathoe*), whiskered bat (*M. mystacinus)*, and Brandt's bat (*M. brandtii*) are cryptic species, similar in morphology, flight pattern and habitat, despite the whiskered and the Brandt's bat being only distantly related (Ruedi and Mayer, 2001). The Brandt's bat was first recognised as a separate species in the UK in 1970; and the Alcathoe bat, first described in 2001 (Von Helversen et al., 2001), was only identified in Britain in 2010 (Jan et al., 2010). It remains likely that the species are still frequently confused. They can roost in the same buildings as the much more common *Pipistrellus spp*. (Dietz and Keifer, 2016), and may be overlooked as a consequence. In addition, there is considerable overlap in their echolocation parameters. When recorded in cluttered environments — which they commonly frequent — there is also a high degree of similarity with the calls of other members of the *Myotis* genus (Russ, 2012). Therefore, confidence in the correct species identification when using acoustic records alone is low. Genotyping has even revealed errors in identification of species in the hand, highlighting the difficulties of monitoring this group of small *Myotis* (Brown, 2016).

Habitat preferences

The Alcathoe bat appears to be very patchily distributed across Europe (Dietz and Keifer, 2016), and is only known in a few regions of Great Britain — Sussex, Surrey, Kent and North Yorkshire. Whilst some of this patchy distribution may reflect misidentification or a lack of survey effort, intensive monitoring effort at 108 locations across England (largely swarming sites but also woodlands) in 2014, with subsequent molecular analysis of 140 faecal samples, did not identify any further locations outside Sussex and Surrey (Jan et al., 2010; Brown, 2016).

There is no information on the diet of the Alcathoe bat in Great Britain. Elsewhere in Europe, it is reported to feed mainly on small Lepidoptera (moths) and Diptera (flies, particularly mosquitoes), but it takes a range of prey, with Formicidae (ants) being very important in some areas (Lučan et al., 2009; Danko et al., 2010).

There is little evidence on its habitat preferences in Great Britain. However, the species is usually captured in areas with extensive semi-ancient woodland ((Jan et al., 2010; Daniel Whitby, *pers. comm.*); Daniel Whitby, *pers. comm.*). Evidence from elsewhere in Europe suggests a preference for old woodlands, structured edges of broadleaved woodland, and riparian habitats with large trees. Limited radio-tracking data show that it forages both in the crowns of trees and over water. Hunting areas are usually within 3km of the roost, although individuals are recorded travelling up to 6km (Lučan et al., 2009; Danko et al., 2010).

The roosting habitats of the species are also poorly characterised. However, it appears to roost almost exclusively in trees during the active season, particularly in oaks. A single record of a roost beneath sarking board in a large mansion in England appears to be the only known building roost known across Europe (Daniel Whitby, *pers. comm.*). As with many other tree-dwelling bats, the colonies regularly fragment into smaller units, and roosts are switched very frequently (Dietz and Keifer, 2016; Daniel Whitby, *pers. comm*.): in the Czech Republic, a study of 10 summer roosts found that the median roost count was 8 individuals (range 1-83) (Lučan et al., 2009). Although some individuals have been found hibernating in underground sites in France, Belgium and Germany, it seems likely that most animals hibernate in trees (Dietz and Keifer, 2016). The distances travelled between summer and winter roots are not known. The species has been identified during swarming surveys at several underground sites in England.

Status

Native.

Conservation Status

- IUCN Red List (GB: DD; England: [DD]; Scotland: [DD]; Wales: [DD]; Global: DD.).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Unknown; England: Unknown; Scotland: n/a; Wales: n/a).

Species' distribution

The method used to produce the smoothed distribution map (see Methods, Section 2.5) removes isolated records. Locations known in North Yorkshire are therefore not shown on the map (Figure 10.3a).

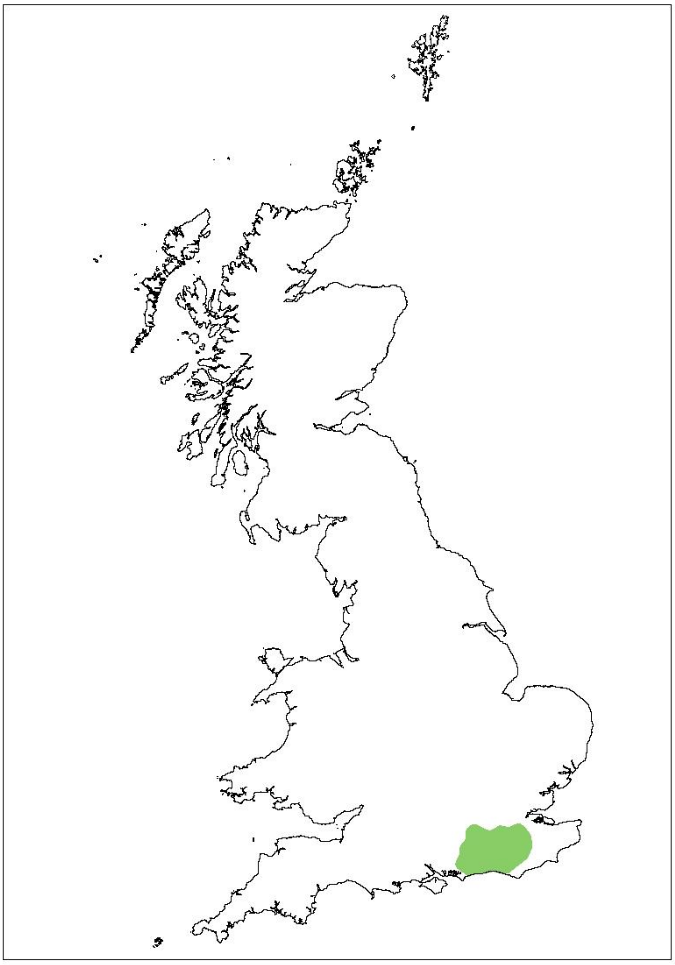


Figure 10.3a Current range of the Alcathoe bat in Britain. Range is based on presence data collected between 2010 and 2016. Areas that contain very isolated records, including those in Yorkshire, have not been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

There has been very little research on this species in the Great Britain. The assessment is therefore based largely on unpublished data provided by two experts.

Maternity colonies in the UK have been found to include 15-100 individuals (Anita Glover and Daniel Whitby, *pers. comm.*), but these frequently fragment so that typical roost sizes are usually small, corresponding with reports elsewhere in Europe (Daniel Whitby, *pers. comm.*). Maternity colonies appear to be comprised almost entirely of female bats, corresponding with the available evidence from continental Europe.. No information on root density is available either from experts or from the literature.

Habitable area was defined as all area within the geographical range. Given that the species uses a mosaic of habitats, and the value of one habitat depends on the configuration and extent of other habitats, a more precise definition of suitable habitat was not possible for this review.

Results

Population estimation and range

The lack of information on roost (or colony) density makes population estimation extremely difficult. Given that at least 8 maternity colonies have been identified, and small numbers of individuals are also captured at swarming and other surveys in Yorkshire and the south east of England, the minimum population is likely to be at least 2,000 individuals: one expert suggests 6,000-8,000 bats (Daniel Whitby, *pers. comm.*). However, the evidence is extremely poor: further systematic surveys, including molecular confirmation of species identity, are urgently required.

No estimation of Alcathoe bat population sizes was made for the last Article 17 Report (Joint Nature Conservation Committee, 2013b).

The current range of the species, based on known records of Alcathoe bats since 1995, is shown in Table 10.3a.

Table 10.3a Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	5,040	800
Scotland	0	n/a
Wales	0	n/a
Britain	**5,040**	**800**

The cluster of records in the south east of England is separated from those in Yorkshire by approximately 350km. Although it is possible that this is an artefact of survey effort and/or misidentification, the gap was not filled during surveys that use genetic confirmation of species identity (Brown, 2016).

Table 10.3b Article 17 Report on the Alcathoe bat population size and range 2006-2011.

Country	Minimum	Maximum
England	n/a	n/a
Scotland	n/a	n/a
Wales	n/a	n/a
Britain	**n/a**	**n/a**

Critique

The estimates provided are extremely poor and rely on expert opinion alone.

Table 10.3c Reliability assessment for the Alcathoe bat. Scores are based on the availability of data on roost location, roost count, and sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or expert opinion with high confidence scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The species was not identified at the time of the Harris and Arnold Reports, so comparisons are not possible. The range is somewhat larger than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b), mainly because of an increase in the number of known sites revealed during intensive specialist surveys in Surrey, Kent and Sussex.

Drivers of change

Table 10.3d Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Unknown.*			

* There are insufficient data on population change to permit drivers of change to be identified.

Data deficiencies

Table 10.3e Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Distribution.	Woodland	Much more intensive survey effort is needed to gain a better understanding of whether this species truly has the restricted distribution so far identified. This should include surveys of woodland and swarming sites. Genetic confirmation of species is required.
Density of roosts.	n/a	Further efforts are needed to identify maternity colonies and identify habitat preferences (e.g., some colonies in England have been found very distant from water, unlike most in continental Europe).
Connectivity of populations.	Woodland	There is an urgent need to establish the extent of connectivity between populations, the extent of inbreeding, and whether the populations are expanding or contracting. This should be done using population genetics in conjunction with assessments of landscape-scale habitat connectivity, and should assess whether urban encroachment and/or infrastructure such as roads present a significant threat. Elsewhere in Europe, high numbers of road casualties are recorded despite the low abundance of the species (Dietz & Kiefer 2016).

Table 10.3f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				England

Future prospects

Table 10.3g An assessment of the future prospects for the Alcathoe bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Unknown

10.4 Whiskered bat *Myotis mystacinus*

Introductory note

The whiskered bat, Alcathoe bat, and Brandt's bat are cryptic species, similar in morphology, flight pattern and habitat, despite the whiskered and the Brandt's bat being only distantly related (Ruedi and Mayer, 2001). The Brandt's bat was first recognised as a separate species in the UK in 1970; and the Alcathoe bat, first described in 2001 (Von Helversen et al., 2001), was only identified in Britain in 2010 (Jan et al., 2010). It remains likely that the species are still frequently confused. They can roost in the same buildings as the much more common *Pipistrellus spp*. (Dietz and Keifer, 2016), and may be overlooked as a consequence. In addition, there is considerable overlap in their echolocation parameters. When recorded in cluttered environments — which they commonly frequent — there is also a high degree of similarity with the calls of other members of the *Myotis* genus (Russ, 2012).

Therefore, confidence in the correct species identification when using acoustic records alone is low. Genotyping has even revealed errors in identification of species in the hand, highlighting the difficulties of monitoring this group of small *Myotis* (Brown, 2016).

Habitat preferences

With echolocation and morphological characteristics suggesting adaptation to foraging in cluttered environments, the whiskered bat is an agile flyer (Norberg and Rayner, 1987b; Holderied et al., 2006). It feeds mainly on small Lepidoptera (moths) and Diptera (flies), including dung flies, houseflies, bluebottles and brown lacewings (Vaughan, 1997; Berge, 2007), which are caught and eaten on the wing. However, it is also capable of gleaning from vegetation, with dietary analysis revealing the presence of diurnal Diptera and Araneida (spiders). There can be considerable differences in prey selection between colonies, suggesting that the species can adapt its diet according to prey availability (Rindle and Zahn, 1997).

Data on the foraging habitat preferences of the whiskered bat are very limited. One radio-tracking study of 27 individuals in Yorkshire (Aegerter, 2003) indicated a preference for farm woodlands, hedgerows, and wetlands; and a further radio-tracking study of 9 bats in south west England indicated a preference for woodland and grassland habitats (particularly cattle-grazed pasture with hedgerows), and avoidance urban and arable habitats (Berge, 2007). Elsewhere in Europe, the species uses a diversity of habitats, including forests, gardens, orchards, riparian corridors and open areas, and can also forage within the crowns of trees (Dietz and Keifer, 2016). It is frequently captured in mist nets placed along linear features such as tall hedgerows, woodland edges and small waterways enclosed by trees (Fiona Mathews, *pers. obs.*).

Maternity roosts are usually located in buildings, although they are sometimes also found in trees and bat boxes (Schober and Grimmberger, 1989). Foraging distances of up to 2.3km (Berge, 2007) and 3.5km (Aegerter, 2003) from maternity roosts have been recorded. As with other *Myotis* species, the whiskered bat frequently visits swarming sites such as cave entrances in the autumn (Parsons et al., 2003a; Glover and Altringham, 2008). While the precise function of swarming is unknown, it is likely to play a role in social communication and mating display, and therefore to be important to species' conservation. Hibernation sites include underground tunnels, ice-houses and caves (Jones, 1991). The species is generally considered to be sedentary across Europe (Dietz and Keifer, 2016), and no long-distance movements have been recorded in Great Britain.

Status

Native.

Conservation Status

- IUCN Red List (GB: DD; England: [DD]; Scotland: [DD]; Wales: [DD]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Unknown; Scotland: Unknown; Wales: Unknown).

Species' distribution

Because of the high probability of misidentification, a joint species' range was derived using all available data for whiskered and Brandt's bats combined. However, records from both swarming sites and roosts are patchier for Brandt's than for whiskered bats. The estimated range is therefore likely to represent more closely the true range for whiskered than Brandt's bats. The precise degree of overlap of the distributions of the species is unknown, but genotyping of bats captured at swarming sites across England (Brown, 2016) confirms the previously reported general pattern of the ratio of Brandt's:whiskered bats increasing from west to east and from south to north in Britain (Richardson, 2000). Expert opinion suggests that there is a ratio of approximately 10:1 of captures of whiskered compared with Brandt's bats at swarming sites, woodland and hedgerows, but this overall ratio is likely to vary locally because the distribution of Brandt's bats appears to be more irregular than that of whiskered bats.

The method used to produce the smoothed distribution map (see Methods, Section 2.5) removes isolated records. Locations known in the Scottish Central Belt are therefore not shown on the distribution map (Figure 10.4a).

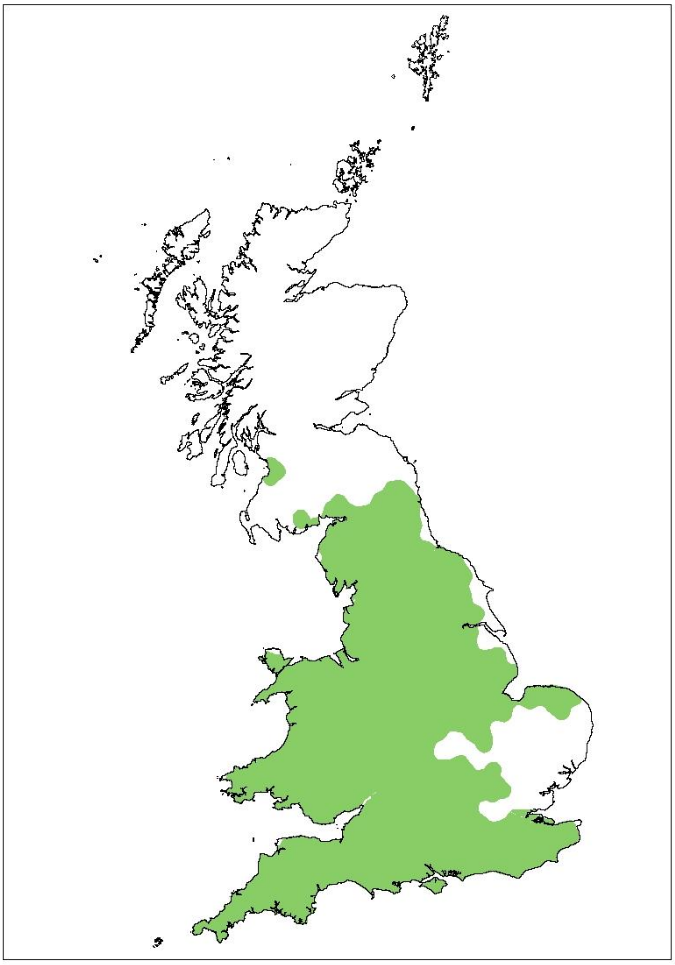


Figure 10.4a Current range of the whiskered/Brandt's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details. The range is therefore likely to be more extensive in Scotland than shown on the map.

Species-specific methods

Estimating population sizes for the whiskered, Brandt's and Alcathoe bats is extremely challenging. In the absence of evidence of genotyping, or of examination of bats in the hand, the veracity of most roost records is unclear. Acoustic surveys cannot be used to provide density information because it is not possible to infer bat numbers from the number of calls recorded. Capture records also cannot readily be used to estimate density because capture success is not proportional to abundance in the environment, and efforts to trap bats tend to be focused on particular sites with a high probability of capture success, such as swarming sites.

Expert opinion was obtained from 4 individuals. A further 2 experts responded to requests for input on this species but were unable to provide information on the parameters needed.

No expert had information on the sex ratio of the population, or on the typical sex ratio of maternity roosts pre-breeding. This information was also not presented by Harris et al. (1995).

Information was available from 465 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses. The median pre-breeding roost size for whiskered/Brandt's bats derived from the available datasets was 14 (95%CI = 6-25, range 1-225, n=27 sites).

The roost density previously reported in Jones et al. (1996) of 0.066 roosts/km^2 was considered too unreliable for further use. It was based on an assumption that the foraging area of each roost was the 5km x 5km grid square in which the roost was located: if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded (Speakman et al., 1991). However, no data on whiskered bats were available to verify this assumption. The estimates also used data collected over several years and took no account of potential roost switching within or between years. The 100km^2 study area monitored during the Cotswold Water Park Bat Initiative (Harris, 2014) had no records of either whiskered or whiskered/Brandt's maternity roosts, despite the fact that other records from trapping and bat boxes indicated that the species was present in the area.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

Results

The values used to derive the density estimates are shown in Table 10.4a.

Table 10.4a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	14 (6-25)
Sex ratio	n/a
Maternity roost density	n/a

Population estimation and range

Given the absence of data on roost density, it was not possible to calculate a population estimate. As it is considered unlikely that most maternity roosts in Britain are known, it was also not possible to make a total count. No population genetics study has been conducted, and therefore no alternative metrics of population size are available. The Article 17 Report on whiskered bat status 2007-2012 is shown below in Table 10.4c (Joint Nature Conservation Committee, 2013b).

Table 10.4b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower limits for England, Scotland, Wales, and the whole of Britain.

Country	**Area within range (km^2)**	**Bat density (adults/km^2)**			**Adult population size**		
		Estimate	**Plausible interval**		**Estimate**	**Plausible interval**	
			Lower	**Upper**		**Lower**	**Upper**
England	109,000	n/a			n/a		
Scotland	2,010	n/a			n/a		
Wales	20,500	n/a			n/a		
Britain	**131,500**	**n/a**			**n/a**		

Table 10.4c Article 17 Report on whiskered bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	30,500	30,500
Scotland	1,500	1,500
Wales	8,000	8,000
Britain	**40,000**	**40,000**

Note: maximum and minimum estimates were the same values for this species.

The current geographical range of the species, based on records of whiskered/Brandt's bats since 1995, is shown in Table 10.4d. The Article 17 Report (Joint Nature Conservation Committee, 2013b) is based on records described as whiskered bats only, whereas the current estimate uses both species combined owing to the difficulties of identification.

Table 10.4d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	109,000	n/a
Scotland	2,010	n/a
Wales	20,500	n/a
Britain	**131,500**	**164,000**

Critique

There is no basis for making a population estimate for this species.

Very few roosts are known, and it is highly likely that there is considerable misidentification of the species. The only available estimate of roost size from the literature gave a mean value of 23.3 individuals based on 15 maternity roosts (Jones et al., 1996), which falls within the confidence limits of our estimates. Both the estimate derived for the current review, and that used by Jones et al. (1996), are considerably smaller than the typical roost size of 20-60 bats reported for other parts of Europe (Dietz and Keifer, 2016).

Experts were unable to provide estimates of roost density. Four experts provided information on roost size, whilst the others were unable to give any additional information. Their estimates of roost counts (usual size 61; typical range 12-99, n=16 roosts) is larger than that derived here. However, they are close to the published data from elsewhere in Europe (Dietz and Keifer, 2016).

Several sources of error are identified. The density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. No expert was able to provide estimates, and it is likely that the species is frequently misidentified. There is also uncertainty about roost sizes, and this is compounded by potential misidentification of the species. No roost counts or density estimates are available for tree roosts. Finally, the ratio of building:tree roosts is unknown, so the scale of bias introduced by basing estimates primarily on data from buildings is unquantifiable.

Table 10.4e Reliability assessment for the whiskered bat. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 40,000 individuals was given in Harris et al. (1995) (England 30,500; Scotland 1,500; Wales 8,000), this estimate was graded as having very poor reliability. Given that there is no basis for deriving a current population estimate, comparison with Harris et al. (1995) was not attempted.

The distribution is similar to that reported by Arnold (1993), which showed the species as being virtually absent from most of Scotland. The current range maps also show the species as being present throughout Wales. This is likely to be a reflection of greater observer effort rather than true range expansion. However, it is also possible that some of the new acoustic records are owing to misidentification. The range is slightly smaller than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b); this difference is likely to reflect the slightly different methodologies.

Other evidence of changes through time

The National Bat Monitoring Programme hibernation count does not distinguish whiskered and Brandt's bats. It suggests that the populations are stable or increasing slightly. However, sample sizes at each site are relatively low, and there are no field or summer roost data available for comparison.

Table 10.4f Trends in whiskered/Brandt's bat activity from baseline to 2015 as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available to estimate trends for Scotland.

Country	Type of site	No. sites**	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	Hibernation	139	1999	39.7*	2.1
Wales	Hibernation	86	1999	-15.9	-1.1
Britain	Hibernation	227	1999	30.6	1.7

* Indicates that the trend is significant ($p<0.05$).
† Since baseline year 1999.

Table 10.4g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparison of point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of substantial changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.4h Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Unknown.*			

* There are insufficient data on population change to permit drivers of change to be identified.

Data deficiencies

Table 10.4i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Species' range.	n/a	Very limited data are available, and confusion with the Alcathoe and the Brandt's bat means that the range is very poorly defined. Trapping at swarming sites and likely habitats is required, particularly in Scotland.
Density of roosts.	n/a	No data available. Systematic study is urgently required, supported by DNA verification of species identity.
Relative proportions of roosts in trees and buildings.	n/a	No data available. Evidence would help to enable future extrapolations of local population size from roosts identified in buildings.
Size of roosts.	n/a	Very limited data available: formal study is urgently required.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change, lighting, etc., on local populations, particularly through the fragmentation of habitat which may restrict access to core foraging areas.	Woodland edge, riparian corridors	No data available. Impacts need to be assessed through monitoring changes to roost size and density, or alternatively, through comprehensive study based on population genetics.
Access to swarming sites.	Cave systems, underground tunnels, possibly large barns.	The species is known to use swarming sites. No information is available on the importance of these sites, and the degree to which access is being lost through either obstruction of the site or loss of connecting habitat.

Future prospects

Table 10.4j An assessment of the future prospects for the whiskered bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Unknown

10.5 Brandt's bat *Myotis brandtii*

Introductory note

The whiskered bat, Brandt's bat and Alcathoe bat are cryptic species, similar in morphology, flight pattern and habitat, despite the whiskered and the Brandt's bat being only distantly related (Ruedi and Mayer, 2001). The Brandt's bat was recognised as a separate species in the UK in 1970; and the Alcathoe bat, first described in 2001 (Von Helversen et al., 2001), was only identified in Britain in 2010 (Jan et al., 2010). It remains likely that the species are still frequently confused. They can roost in the same buildings as the much more common *Pipistrellus spp*. (Dietz and Keifer, 2016), and may be overlooked as a consequence. In addition, there is considerable overlap in their echolocation parameters. When recorded in cluttered environments — which they commonly frequent — there is also a high degree of similarity with the calls of other members of the *Myotis* genus (Russ, 2012). Therefore, confidence in the correct species identification when using acoustic records alone is low. Genotyping has even revealed errors in identification of species in the hand, highlighting the difficulties of monitoring this group of small *Myotis* (Brown, 2016).

Habitat preferences

The echolocation and morphological characteristics of the Brandt's bat are similar to those of the whiskered bat, suggesting adaptation to foraging in cluttered environments (Norberg and Rayner, 1987b). It is has highly manoeuvrable flight and, like the whiskered bat, has a broad dietary range. It feeds on Diptera (including midges and brown lacewings) and Lepidoptera (moths), but also gleans Araneida (spiders) and diurnal Diptera from vegetation (Vaughan, 1997; Berge, 2007).

Only one detailed radio-tracking study of habitat preferences of whiskered bats and Brandt's bats has been conducted in the UK. Using data on 11 Brandt's bats in south west England, it was concluded that whiskered bats favoured coniferous woodland habitat, followed by mixed woodland and grassland (Berge, 2007). The Brandt's bats is frequently captured in mist nets placed along linear features such as tall hedgerows, forest rides and woodland edges (Fiona Mathews, *pers. obs.*). Elsewhere in Europe, it is associated with woodland, particularly damp areas close to water (Taake, 1984).

Most known maternity roosts are found in buildings, although they are sometimes also situated in trees, bridges and bat boxes (Schober and Grimmberger, 1989). The maximum foraging distance for females at maternity roosts is reported as 3.2km for the only British radio-tracking study (Berger, 2006). As with other *Myotis* species, the Brandt's bat frequents underground swarming sites in the autumn. Hibernation sites include underground tunnels, ice-houses and caves, and Brandt's bats appear to hibernate for longer than whiskered bats (Jones, 1991). The species is generally considered to be sedentary across Europe (Dietz and Keifer, 2016), and no long-distance movements have been recorded in Great Britain.

Status

Native.

Conservation Status

- IUCN Red List (GB: DD; England: [DD]; Scotland: [DD]; Wales: [DD]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Unknown; Scotland: n/a; Wales: Unknown).

Species' distribution

Because of the high probability of misidentification, a joint species' range was derived using all available data for whiskered and Brandt's bats combined (Figure 10.5a). However, records from both swarming sites and roosts are patchier for Brandt's than for whiskered bats. The estimated range is therefore likely to be less reliable for Brandt's bats. The precise degree of overlap of the distributions of the species is unknown, but genotyping of bats captured at swarming sites across England confirms the general pattern of increasing proportions of Brandt's bats being found as one moves from west to east, and from south to north, in Britain (Richardson, 2000).

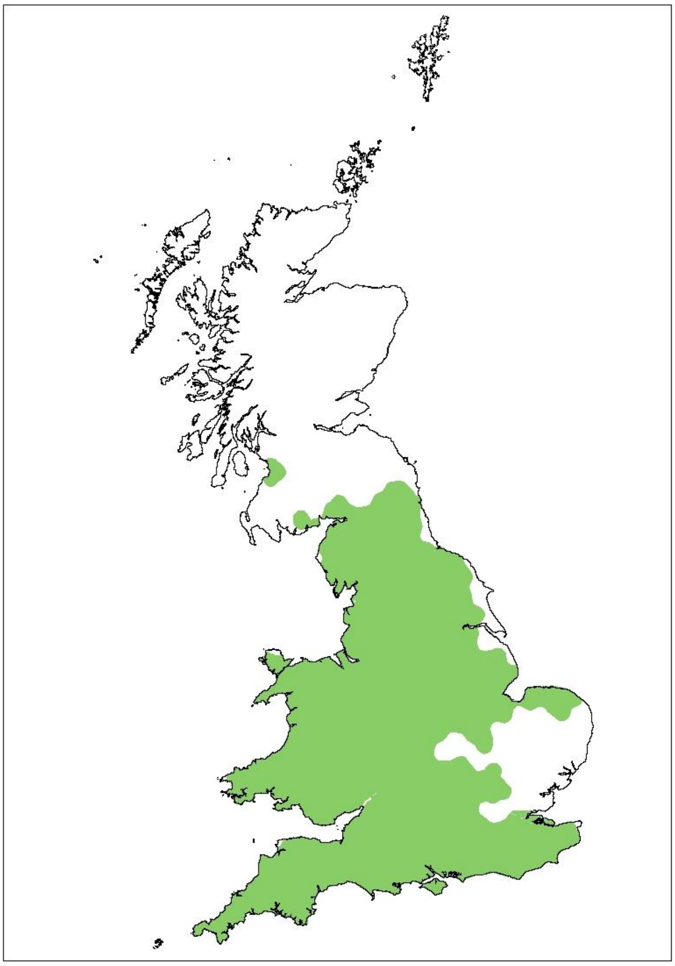


Figure 10.5a Current range of the whiskered/Brandt's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Estimating population sizes for the whiskered, Brandt's and Alcathoe bats is extremely challenging. In the absence of evidence of genotyping, or of examination of bats in the hand, the veracity of most roost records is unclear. Acoustic surveys cannot be used to provide density information because it is not possible to infer bat numbers from the number of calls recorded. Capture records also cannot readily be used to estimate density because capture success is not proportional to abundance in the environment, and efforts to trap bats tend to be focused on particular sites with a high probability of capture success, such as swarming sites.

Because of the high probability of misidentification, a joint species' range was derived using all available data for whiskered and Brandt's bats combined. However, records from both swarming sites and roosts are more patchy for the Brandt's than for the whiskered bat. The estimated range is therefore likely to be less reliable for Brandt's bats.

Information was available from 465 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses. The median pre-breeding roost size for whiskered/Brandt's bats derived from the available datasets was 14 (95%CI = 6-25, range 1-225, n=27 sites).

Expert opinion suggested that there is a ratio of approximately 10:1 of captures of whiskered compared with Brandt's bats at swarming sites, woodland and hedgerows. No expert had information on the sex ratio of the population, or on the density of roosts, and this information was not available from Harris et al. (1995). The roost density previously reported in Jones et al. (1996) of 0.06 roosts/km^2 was considered too unreliable to be used in the current review: it was based on an assumption that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded (Speakman et al., 1991). However, no data on Brandt's bats were available to verify this assumption. The estimates also used data collected over several years and took no account of potential roost switching within or between years. The 100km^2 study area monitored during the Cotswold Water Park Bat Initiative (Harris, 2014) had no records of either whiskered or whiskered/Brandt's maternity roosts despite the fact that records from trapping and bat boxes indicated that the species was present in the area.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

Results

The values used to derive the density estimates are shown in Table 10.5a.

Table 10.5a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	14 (6-25)
Sex ratio	n/a
Maternity roost density	n/a

Population estimation and range

Given the absence of data on roost density, it was not possible to calculate a population estimate. As it is considered unlikely that most maternity roosts in Britain are known, it was also not possible to make a total count. No comprehensive population genetics study has been conducted, and therefore no alternative metrics of population size are available. The Article 17 Report on Brandt's bat population size 2007-2012 is shown below in Table 10.5c (Joint Nature Conservation Committee, 2013b).

Table 10.5b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain. The area within the range is likely to be overestimated because the range is based jointly on the whiskered/Brandt's bats, and the Brandt's bat is generally considered rarer and more patchily distributed.

Country	Area within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	109,000	n/a			n/a		
Scotland	2,010	n/a			n/a		
Wales	20,500	n/a			n/a		
Britain	**131,500**	**n/a**			**n/a**		

Table 10.5c Article 17 Report on Brandt's bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	22,500	22,500
Scotland	0	0
Wales	7,000	7,000
Britain	**29,500**	**29,500**

Note: maximum and minimum estimates were the same values for this species.

The current distribution estimate for the species, based on known records of whiskered/Brandt's bats since 1995, is shown in Table 10.5d. The Article 17 Report (Joint Nature Conservation Committee, 2013b) is based on records described as Brandt's bats only, whereas the current estimate combines both species combined owing to the difficulties of identification.

Table 10.5d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	109,000	n/a
Scotland	2,010	n/a
Wales	20,500	n/a
Britain	**131,500**	**134,000**

Critique

There is no basis for making a population estimate for this species.

Very few roosts are known, and it is highly likely that there is considerable misidentification of the species. The only available estimate of roost size from the literature (as opposed to the available datasets) gives a mean value of 28.5 individuals based on 5 maternity roosts (Jones et al., 1996), which falls outside the confidence limits of our estimates. Both the estimate derived for the current review, and that used by Jones et al. (1996), are considerably smaller than the typical roost size of 20-60 bats reported for other parts of Europe (Dietz and Keifer, 2016).

Experts were unable to provide estimates of roost density. Four experts provided information on roost size, whilst the others were unable to contribute this information. Their estimates of roost counts for whiskered/Brandt's combined (usual size 61; typical range 12-99, n=16 roosts) is larger than that derived here. However, they are close to the published data for elsewhere in Europe (Dietz and Keifer, 2016).

Four main sources of error are identified. The density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. No expert was able to provide estimates, and it is likely that the species is frequently misidentified. There is also considerable uncertainty about roost sizes, and this is compounded by potential misidentification of the species. No roost counts or density estimates are available for tree roosts. Finally, the ratio of building:tree roosts is unknown, so the scale of bias introduced by basing estimates primarily on data from buildings is unquantifiable.

Table 10.5e Reliability assessment for the Brandt's bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
	Overall reliability score		**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 30,000 individuals was given in Harris et al. (1995) (England 22,500; Scotland 500; Wales 7,000), this estimate was graded as having extremely poor reliability. Given that there is no basis for deriving a current population estimate, comparison with Harris et al. (1995) was not attempted.

The distribution is similar to that reported by Arnold (1993), which showed the species as being virtually absent from most of Scotland. The current range maps also show the species as being present throughout Wales. This is likely to be a reflection of greater observer effort rather than true range expansion. However, it is also possible that some of the new acoustic records are owing to misidentification. The range is slightly smaller than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b); this difference is likely to reflect the differing methodologies.

Other evidence of changes through time

The National Bat Monitoring Programme hibernation count does not distinguish whiskered and Brandt's bats. It suggests that the populations are stable or increasing slightly. However, sample sizes at each site are relatively low, and there are no field or summer roost data available for comparison.

Table 10.5f Trends in whiskered/Brandt's bat activity from baseline to 2015 as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available to estimate trends for Scotland.

Country	**Type of site**	**No. sites****	**Start year for monitoring**	**Long-term trend (%)†**	**Mean annual trend (%)**
England	Hibernation	139	1999	39.7*	2.1
Wales	Hibernation	86	1999	-15.9	-1.1
Britain	Hibernation	227	1999	30.6	1.7

* Indicates that the trend is significant ($p<0.05$).
† Since baseline year 1999.

Table 10.5g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				England Wales*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.5h Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Unknown.*			

* There are insufficient data on population change to permit drivers of change to be identified.

Data deficiencies

Table 10.5i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Species' range.	n/a	Very limited data available, and confusion with whiskered bat means that the range is very poorly defined. Trapping at swarming sites and likely habitats is required, particularly in Scotland.
Density of roosts.	n/a	No data available. Systematic study is urgently required, supported by DNA verification of species identity.
Relative proportions of roosts in trees and buildings.	n/a	No data available. The evidence would help to enable future extrapolations of local population size from roosts identified in buildings.
Size of roosts.	n/a	Very limited data available: formal study is urgently required.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change, lighting, etc., on local populations, particularly through the fragmentation of habitat which may restrict access to core foraging areas.	Woodland edge, riparian corridors	No data available. Impacts need to be assessed through monitoring changes to roost size and density, or alternatively, through comprehensive study based on population genetics.
Access to swarming sites.	Cave systems, underground tunnels, possibly large barns.	The species is known to use swarming sites. No information is available on their importance, and the degree to which access is being lost, through either obstruction of the site, or loss of connecting habitat.

Future prospects

Table 10.5j An assessment of the future prospects for the Brandt's bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Unknown

10.6 Bechstein's bat *Myotis bechsteinii*

Habitat preferences

The Bechstein's bat has a diet high in Lepidoptera (moths) and woodland-associated Diptera (flies). There is no detailed dietary study available for the UK except for one unusual colony that roosts in a building. There, the main prey items were ground dwelling arthropods — Chilopoda (centipedes), Dermaptera (earwigs), Coleoptera (ground beetles) and Arachnida (harvestmen) (McAney et al., 1991). More extensive research in Germany indicates that the species predominantly feeds on Lepidoptera, Planipennia (particularly lacewings) and Coleoptera (beetles) (Woltz, 1992). The species forages primarily in and around broadleaved woodland, and uses very wide bandwidth calls to distinguish prey from vegetation clutter when hawking (Siemers and Schnitzler, 2004). As well as catching insects on the wing, it can also perch-feed. Its prey includes diurnal and non-volant species (Vaughan, 1997), and gleaning is used as the main foraging strategy. Like the brown long-eared bat, the Bechstein's bat can use passive listening rather than echolocation to detect prey at close range, and hearing (tympanate) moths are an important dietary component. The strategy may assisting in niche separation from the Natterer's bat (Siemers and Swift, 2006).

Despite being strongly associated with broadleaved woodland, particularly semi-natural ancient woodland with dense structured understorey (Greenaway & Hill 2004), the species also forages along large hedgerows and wooded riparian corridors, and can roost in individual trees found in these environments (Palmer et al., 2013). There is evidence of segregation of the sexes into different woodlands, with males using what appear to be less optimal habitats (Harris and Yalden, 2008; Dietz and Pir, 2011).

Maternity roosts are usually located in trees, most commonly in woodpecker holes and rot holes, but also in other crevices. A wide range of tree species is used, including oak, ash, aspen, London plane, crack-willow, and field maple (Palmer et al., 2013; Chris Damant, *pers. comm.*; Fiona Mathews, *pers. obs.*) In some woodlands, particularly those with few natural tree holes, colonies can make extensive use of bat boxes. Only a single building roost is known in Great Britain (Schofield and Morris, 2000).

Radio-tracking evidence shows that individuals from maternity colonies are very sedentary during the breeding season. For example, the mean distances between the roost and core foraging areas in studies in Dorset and Worcestershire were 620m (range 300m-960m) (Schofield and Morris, 2000) and 726m respectively (range 0-3310m; Palmer et al., 2013). Home ranges are also very small compared with most other British bats; several projects in England and Luxembourg report core areas of less than 5.3ha and frequently even smaller (Schofield and Morris, 2000; Dietz and Pir, 2009; Palmer et al., 2013).The species appears particularly vulnerable to habitat fragmentation: a study of a German population close to a motorway found that no individuals flew over the road, and those that crossed used underpasses. In addition, individual home ranges adjacent to the motorway were small compared with other forest edges (Kerth and Melber, 2009). Bechstein's bat colonies regularly break into smaller units (fission-fusion structure) and can occupy numerous alternative roosts (Kerth and Konig, 1999). However, there is little overlap between the colony home range and that of neighbouring groups, suggesting that colonies are spatially segregated (Dawo et al., 2013). Ringing data indicate little or no interchange of individuals between adjacent colonies (Henry Schofield, *pers. comm.*; Keith Cohen, *pers. comm*.; Kerth et al., 2002.

There is good evidence of high natal philopatry in females, whereas about half of males roosting in close proximity to maternal colonies are immigrants (Kerth et al., 2000; Kerth et al., 2002). Yet local males father fewer than 25% of offspring, and inbreeding is low, implying that females must mix with other males outside the local area, possibly at swarming sites (Kerth et al., 2000; Kerth et al., 2002). Bechstein's bats are regularly captured at some swarming sites in the south of England, although their distribution is patchy. The precise function of these sites is unknown, but as they are likely to be linked with mating activity, they are extremely important for the conservation of the species (Parsons et al., 2003b). Ringed individuals have been recorded to move over 15km to reach swarming sites (Fiona

Mathews, *pers. obs.*), as has also been reported in continental Europe (Rudolph et al., 2004).

The species may be particularly susceptible to habitat loss because of its highly sedentary behaviour. Across Europe, summer and winter roosts are found in close proximity, and the longest recorded movements are 48km and 73km (Dietz and Keifer, 2016). Hibernation sites include tunnels, caves, and probably also tree holes (Dietz and Keifer, 2016).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: n/a; Wales: [EN]; Global: NT.).
- National Conservation Status (Article 17 overall assessment 2013. Annex II and IV; UK: Unknown; England: Unknown; Scotland: n/a; Wales: Unknown).

Species' distribution

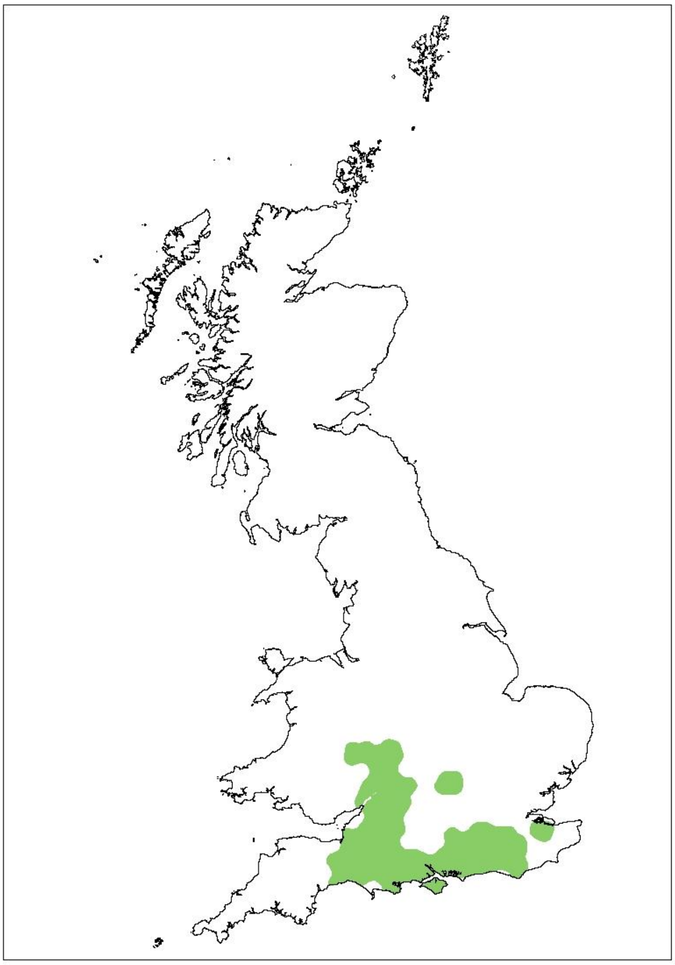


Figure 10.6a Current range of the Bechstein's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Because of the very strong dependency of Bechstein's bats on broadleaved woodland, and the almost complete absence of roosts in buildings, experts were asked to provide information on roost densities within broadleaved woodland only. Given that no national data were available on the extent of broadleaved woodland of different types or qualities, assessments were made for the habitat as a whole.

Expert opinion was obtained from 7 individuals. A further expert responded to requests for input on this species but was unable to the provide information needed for the calculation of population size. Data were also extracted from the report by Palmer et al. (2013). Information was available for more than 75 roosts. Experts generally had a reasonable degree of confidence in their estimates of roost density (median score 6/10; range 4-8), whereas confidence in roost counts was slightly lower (median score 5.5/10; range 3-7), with experts citing as their main constraints the difficulty of seeing all potential roost exits and observing bats that were emerging an hour after sunset.

All available literature and expert opinion suggest that the maternity colonies pre-parturition are exclusively female. Therefore, the proportion female was set as 1. No expert had information on the sex ratio of the population, and this information was also not available from Harris et al. (1995).

Information on typical pre-breeding roost size, and typical upper and lower values, were derived from expert opinion. The median of these values, based on experience at 75 sites were 42.5, 25 and 90 respectively. The only available evidence on roost density (roosts/km^2) was derived from expert opinion, and the median value was used.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: median of expert opinions for typical upper and lower counts.
- Sex ratio: set as 1 for this species as roosts are considered to be exclusively female.
- Roost density: the median value for density in typical habitat was used, together with the median value for typical density in poor quality habitat and the median value for typical density in good quality habitat.

The density of adult bats was calculated as follows:

*Median adult density (bats/km^2) = ((median bats/roost†) * (propn. roost female) * (typical n roosts/typical km^2 broadleaved woodland)) * 2*

*Lower limit = ((lowest plausible n. adults/typical roost) * (propn. roost female) * (plausible n. roosts /typical km^2 poor quality broadleaved woodland))* 2*

*Upper limit = ((upper plausible n. adults/typical roost) * (propn. roost female) * (plausible n. roosts /typical km^2 good quality broadleaved woodland))* 2*

† 'Roost' here means maternity roost in the pre-parturition period.

The estimate of population size was based on adult population density and habitat availability within the range. Habitable area was defined as only broadleaved woodland because of the very strong dependency of maternity colonies on roost locations within woodland. It is acknowledged that there can be maternity roosts in other locations, such as within mature trees in hedgerows.

*Total Adult Population = Median adult density in mixed habitat (bats/km^2) * total habitable area within range (km^2)*
*Lower limit = Lower limit adult density in mixed habitat (bats/km^2) * total habitable area within range (km^2)*
*Upper limit = Upper limit adult density in mixed habitat (bats/km^2) * total habitable area within range (km^2)*

Results

The values used to compute bat density estimates are shown in Table 10.6a.

Table 10.6a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	42.5 (25-90)
Sex ratio	1
Maternity roost density	0.1 (0.08-0.12)

Population estimation and range

Table 10.6b Area within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km^2)*	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	2,550	8.5	4.0	21.6	21,600	10,200	55,000
Scotland	0	8.5	4.0	21.6	0	0	0
Wales	29†	8.5	4.0	21.6	247	116	626
Britain	**2,580**	**8.5**	**4.0**	**21.6**	**21,800**	**10,300**	**55,600**

*Broadleaved woodland only.
† No breeding colonies are currently known in Wales, and this value is derived from the use of smoothed kernels to estimate range. However, expert opinion suggests that there are suitable areas of habitat in the south of Wales. Given the presence of breeding colonies in Herefordshire, Worcestershire and Gloucestershire, it is likely that the species also breeds in Wales.

The Article 17 Report on Bechstein's bat population size 2007-2012 is shown in Table 10.6c; (Joint Nature Conservation Committee, 2013b).

Table 10.6c Article 17 Report on Bechstein's bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	1,500	1,500
Scotland	0	0
Wales	Not estimated	Not estimated
Britain	**1,500**	**1,500**

Note: maximum and minimum estimates were the same values for this species.

The current geographical range for the species, based on known records of Bechstein's bats since 1995, is shown in Table 10.6d.

Table 10.6d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	23,300	n/a
Scotland	0	0
Wales	155	n/a
Britain	**23,500**	**37,900**

Critique

Considerable effort has gone into monitoring Bechstein's bats over the past 10 years, although many of the findings have not yet been formally published. Experts were able to provide information on a large number of roosts, and reported having reasonable confidence in the evidence they submitted. Therefore, despite the challenges of identifying tree roosts, and the need to identify the species by trapping rather than acoustic monitoring (because of its quiet calls and overlap in its call parameters with other *Myotis* species, (Russ, 2012), it is possible to derive population estimates for this species.

Bechstein's bats have a fission-fusion social structure — not only do colonies switch roosts very frequently, but the group can also divide across multiple sites before re-joining. It is possible that there is some overestimation caused by smaller subunits of the colony not being counted, biasing the data towards roosts containing larger numbers of individuals. Given that roosts usually have to be identified by radio-tracking, there is a higher probability of catching and trapping a bat from a larger than a smaller roost. However, this bias may be counteracted by the difficulty of performing complete exit counts (bats emerge after dark and tree roosts are particularly challenging to study owing to multiple access points). The median roost count estimated in this project was very similar to that obtained for 7 well-studied colonies monitored by Durrant et al. (2009); in that study, the genetic estimate of effective population size suggested that the roost counts were in the correct order of magnitude.

The range may be more extensive than estimated here. Considerable improvements in identifying the species have been made in recent years, encouraged by a systematic trapping programme run by the Bat Conservation Trust (Miller, 2011) which identified 37 new sites and extended the known range. However, the selection criteria used to target survey effort excluded some areas of south west England and Wales that are now thought likely to be suitable for the species.

It has not been possible to adjust the estimates for occupancy rates owing to a lack of data. Although the Bat Conservation Trust's Bechstein's Bat Project identified the species in 19% of broadleaved woodlands surveyed (Miller, 2011), it is unclear how to extrapolate this information to broadleaved woodlands in general: the survey sites were selected according to certain habitat criteria (which would tend to overestimate occupancy if extrapolated to all broadleaved woodland), but trapping was of short duration (which may have underestimated true occupancy).

The estimates presented here are based on the assumption that all bats in pre-breeding maternity colonies are female, and that males will be dispersed singly or in small groups throughout the woodland or among trees in adjacent habitats (e.g., hedgerows, parkland and gardens). The strategy for computing population sizes has therefore been to estimate total adult density as being twice that of the adult females counted at maternity roosts. However, if some broadleaved woodlands are occupied exclusively by females, and others exclusively by males, then this approach may substantially overestimate the total population size (by up to a factor of 2).

The estimates used in this review were derived almost entirely from expert opinion. An alternative approach for calculating bat density is simply to divide the total number of adult bats recorded pre-breeding in a given site by the site area. Based on data from 6 sites (Grafton, Bernwood, Brackett's Coppice, Ebernoe, Stonehill, and Trowbridge), the median density estimate is 109 bats/km^2. All of these sites are known to have substantial Bechstein's bat populations, and if the density estimate is adjusted for the 19% occupancy rates found in the Bat Conservation Trust Project (Miller, 2011), then the density estimate falls to 21 bats/km^2 in good quality broadleaved woodland, and would fall further if all types of woodland — such as those without understory — were included. The results are therefore within the plausible ranges previously identified.

Three main sources of error are identified. Firstly, there is uncertainty about occupancy rates for broadleaved woodland. Secondly, the range may be underestimated, as it is difficult to identify Bechstein's bats with certainty using acoustic surveys, and tree roosts are difficult to find. Surveys therefore depend heavily on the availability of personnel suitably qualified to trap bats. Finally, the extent to which Bechstein's bats use hedgerows and parkland for roosting and foraging is unknown. The current focus on broadleaved woodlands may therefore underestimate the true population size.

Table 10.6e Reliability assessment for Bechstein's bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	
	1	A few (4 to 6)	1
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**2**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 1,500 individuals, all in England, was given in Harris et al. (1995), this estimate was graded as having very poor reliability. At the time, no breeding colonies were known, and all summer records were just of single individuals. Arnold (1993) reported that there were only 19 hectads (10km x 10 km squares) with accepted records since 1960. However, there has been a substantial change in survey intensity and techniques over the past decade, and so comparisons with earlier estimates are not appropriate.

The range is slightly smaller than that given in the JNCC Article 17 Report (Joint Nature Conservation Committee, 2013b).

Other evidence of changes through time

Population genetic data suggest that, in addition to suffering a historical bottleneck, the species has undergone recent declines in Great Britain (Durrant et al., 2009). However, recent evidence suggests that levels of inbreeding are less than previously feared, with most populations being comparable with those in continental Europe (Wright et al., 2018).

Table 10.6f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in monitoring techniques and observer effort.

Drivers of Change

Table 10.6g Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Unknown.*			

* There are insufficient data on population change to permit drivers of change to be identified.

Data deficiencies

Table 10.6h Areas where further research is required to improve the reliability of population size estimates and/or inform conservation management.

Data deficiencies	Habitat	Details
Density of roosts outside woodland.	Hedgerows and single trees	Density is poorly estimated in non-woodland habitats.
Roost and/or colony size.	Trees	Thermal imaging/infra-red video-photography and/or genetic approaches are needed to improve estimates, given that the species is crevice-dwelling and emerges late in the evening. Investigation of roost switching and colony structure would help identify the extent to which the colony is dependent on individual trees.
Occupancy of woodland.	Broadleaved woodland	Data on the proportion of occupied woodlands are required throughout the species' range.
Effects of the cumulative pressures of land use change and urban/lighting encroachment on roosting and foraging areas.	All	No data available.
Impacts of road casualties, and fragmentation of landscapes by roads, on British populations.	Roads	Evidence from Germany suggests that home ranges are smaller close to roads, and the species crosses roads using under-passes rather than by flying over roads (Kerth and Melber, 2009). Road casualties are found in continental Europe (Fensome and Mathews, 2016).
Impacts of changing woodland management (including new planting, coppicing and wood-pasture), affecting the total woodland area, amount of standing deadwood, and structure of understory on roost and foraging availability.	Broadleaved woodland	No data are available for this species. Work on other woodland bats suggests these may be important issues (e.g. Boughey et al., 2011; Murphy et al., 2012).

Data deficiencies	Habitat	Details
Identification and protection of swarming sites and routes used by bats to access them.	Quarries, tunnels, potentially other habitats including woodland glades	The species is dependent on gene flow away from maternity sites; and swarming sites are likely to play an important role (see, e.g., Parsons et al., 2003). The degree to which access is being lost, either through obstruction of the site, or loss of connecting habitat, is unknown.

Future prospects

Table 10.6i An assessment of the future prospects for the Bechstein's bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Decline

10.7 Daubenton's bat *Myotis daubentonii*

Habitat preferences

The Daubenton's bat preys mainly on species with aquatic larval stages, particularly nematoceran Diptera (mainly midges) and Trichoptera (caddisflies). Lepidoptera (moths), Coleoptera (beetles), and Ephemeroptera (mayflies) are also taken, but in smaller quantities (Swift and Racey, 1983; Sullivan et al., 1993). It primarily forages by gaffing insects from the surface of the water with its feet or mouth, but it can also use aerial hawking (Jones and Rayner, 1988). Areas of water with ripples or surface vegetation such as duckweed are avoided: not only is prey detection by echolocation more challenging in these areas, but the abundance of flying insects just above the water surface is higher in areas where the surface is smooth (Boonman et al., 1998; Rydell et al., 1999; Warren et al., 2000). Nutrient enrichment of waterways by effluent may influence activity, but the literature is conflicting on the direction of the effect (Vaughan et al., 1996; Racey et al., 1998; Abbott et al., 2009).

The species is strongly associated with riparian habitats. It prefers large waterways with abundant woodland in the local environment (surrounding 1km square; Langton et al., 2010) and, at least in upland riverine environments, it appears to select locations with trees on both banks (Warren et al., 2000). Maternity roosts are usually located in trees, most commonly in broadleaved woodland, but solitary trees, bat boxes, buildings, bridges and other artificial structures are also used. Roosts are commonly, but not always, located close to riparian habitats. In North Yorkshire, the overall mean distance between the roost and foraging site was approximately 6km (range 1km-17km), with a shorter foraging range (c. 2km) for lactating females (Altringham and Senior, 2005); in Scotland, distances of up to 2km were recorded (Swift and Racey, 1983). Roosts tend to be sexually segregated during the maternity season (Swift and Racey, 1983; Senior et al., 2005; August et al., 2014). There may also be segregation along altitudinal gradients in upland regions, with the poorest quality regions being used exclusively by males (Russo, 2002; Senior et al., 2005). In North Yorkshire, radio-tracking data suggest that whilst females exploit optimal habitat exclusively, and males use poorer habitat, intermediate areas include mixed-sex roosts and are used by both sexes for foraging. (Senior et al., 2005; Angell et al., 2013). This contrasts with the social structure observed in southern England, where, although roosts were sexually segregated, there was no evidence of spatial separation of male and female roosts (August et al., 2014).

Daubenton's bats, particularly males, are one of the species most commonly captured at swarming sites (Parsons and Jones, 2003; Glover and Altringham, 2008), and individuals can travel long distances (up to 27km) to reach them (Parsons and Jones, 2003). Offspring from all-female maternity colonies have a high probability of being fathered by bats caught at swarming sites (Angell et al., 2013), and data indicating high levels of gene flow in local populations in Scotland also point towards an important role for swarming (Ngamprasertwong et al., 2008). However, there is also good evidence that mating occurs at maternity sites when roosts are mixed sex (Encarnação, 2012; Angell et al., 2013). The overall importance of swarming sites for the conservation of this species therefore remains unclear.

In Great Britain, there are only a few records of long-distance movements, although these are known in continental Europe (e.g., 260km to reach a hibernation site (Urbanczyk, 1990)). Recent population genetic evidence suggests that there is some structuring of the population between Scotland and northern England. However, the same study indicated no substantial difference between bats sampled in the UK and continental Europe, which implies that there must be some movement of individuals across the English Channel or North Sea (Atterby et al., 2010). Hibernation sites include tunnels, caves, and probably also tree holes (Dietz and Keifer, 2016).

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A species' distribution map is provided in Figure 10.7a. Gaps in the species' distribution in Scotland are likely to represent areas lacking survey effort, rather than true absences, with the exception of the areas in the far north of the country.

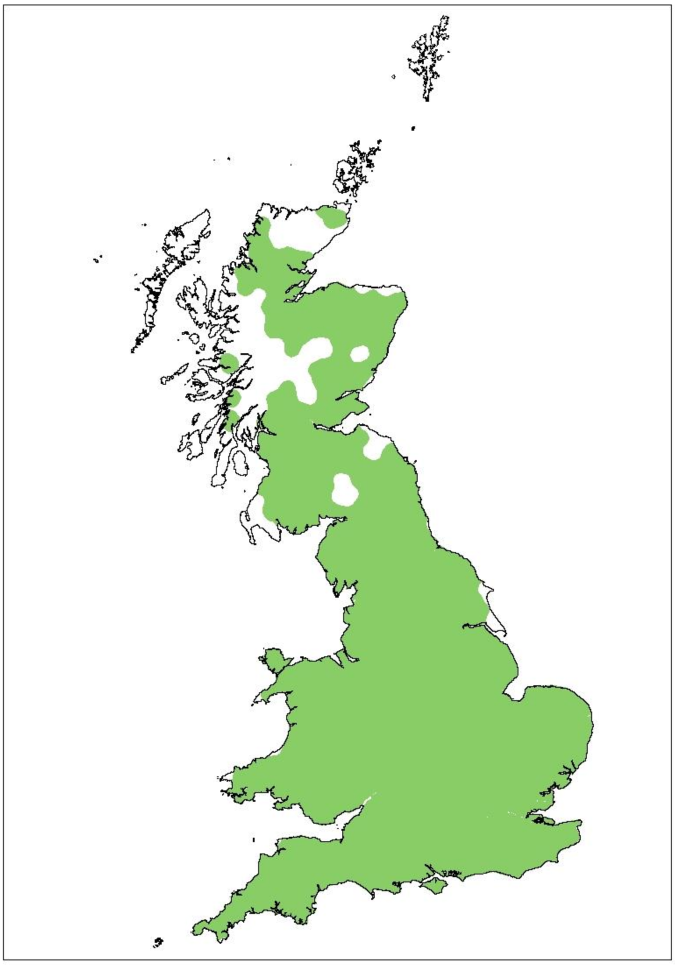


Figure 10.7a Current range of the Daubenton's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Information was available from 15 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses. Small roosts were not excluded from the assessment because the fission-fusion social structure of the species means that colonies are divided across several roosts: even those locations with <10 bats can include breeding individuals. The median pre-breeding roost size derived from the available datasets was 44 (95%CI = 20-143, range = 2-257, n=15 sites).

Expert opinion was obtained from 6 individuals. A further 3 experts responded to requests for input on this species but were unable to provide the information necessary for the calculation of population size. Only two experts provided information on the sex ratio of maternity roosts pre-breeding: one suggested that colonies were 100% female and the other 50%-100%. The remaining experts all reported being unsure. This uncertainty corresponds with the literature. Therefore, the main estimates of the number of female bats per roost, and the upper plausible limit, were computed on the assumption that the roosts were entirely female; but the lower plausible limit was based on the assumption that only 50% of the roost was female. No expert was able to provide information on the sex ratio of the population.

Only one roost density estimate was provided by an expert (reliability score 8/10); data were therefore also extracted from the published report on roosts in North Yorkshire (Jones et al., 1996). The estimates in Jones et al. (1996) were based on an assumption that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). Given that no data were available to verify this assumption, a second density estimate was derived for the purpose of the current calculations by using the entire 2,500km^2 study area (which gives a density estimate of 0.007 bats/km^2). The highest and lowest values of the available estimates (expert opinion and literature) were adopted to define plausible roost densities in good and poor quality habitats.

Data from the Cotswold Water Park where very high roost densities were found (0.21 roosts/km^2; Harris 2014) were not used for this species. This is because the exceptionally high availability of riparian habitat in this study area means it would be unrepresentative of even good quality habitat, and so would lead to an overestimation of national population sizes.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

The population estimate was calculated as follows:

Adult bat density (bats/km^2)

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*

*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*

*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is the typical maternity roost in the pre-parturition period. n. is the number of adults.

‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The estimate of population size was based on adult population density across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density (bats/km^2) * total area within range (km^2)*

*Lower limit = Lower limit adult density (bats/km^2) * total area within range (km^2)*

*Upper limit = Upper limit adult density (bats/km^2) * total area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.7a.

Table 10.7a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	44 (20-143)
Sex ratio	1 (0.5-1)
Maternity roost density	0.06* (0.007**-0.08†)

* Expert opinion; Jones et al. (1996) provided the same value.
** Jones et al. (1996).
† Expert opinion.

Population estimation and range

Table 10.7b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain. .

		Bat density (adults/km^2)			Adult population size		
Country	**Area within range (km^2)**	**Estimate**	**Plausible intervals**		**Estimate**	**Plausible intervals**	
			Lower	**Upper**		**Lower**	**Upper**
England	129,000	5.3	0.1	22.9	682,000	18,100	2,950,000
Scotland	44,400	5.3	0.1	22.9	235,000	6,220	1,020,000
Wales	20,400	5.3	0.1	22.9	108,000	2,860	466,000
Britain	**194,000**	**5.3**	**0.1**	**22.9**	**1,030,000**	**27,000**	**4,440,000**

The Article 17 Report on Daubenton's bat population size 2007-2012 is shown in Table 10.7c (Joint Nature Conservation Committee, 2013b).

Table 10.7c Article 17 Report on Daubenton's bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	95,000	95,000
Scotland	40,000	40,000
Wales	Not estimated	Not estimated
Britain	**135,000**	**135,000**

Note: maximum and minimum estimates were the same values for this species.

The current geographical range of the species, based on known records of Daubenton's bats since 1995, is shown in Table 10.7d.

Table 10.7d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	129,000	n/a
Scotland	44,400	n/a
Wales	20,400	n/a
Britain	**194,000**	**224,000**

Critique

The plausible range of the estimated population size for Daubenton's bats is extremely wide. This is partly because of uncertainty about roost size, as reflected in the very wide confidence intervals (95% CI = 20-143 individuals). It appears likely, based on data from elsewhere in Europe, that Daubenton's bats have a fission-fusion social structure, were there is frequent movement between roosts, and groups can divide across multiple sites before re-joining (Lučan and Radil, 2010). It is possible that there is some overestimation caused by smaller subunits of the colony not being counted, and a tendency for observers to be biased towards the reporting of large roosts. However, this bias may be counteracted by the difficulty of performing complete exit counts (the species emerges about 40 minutes after sunsets, and tree and bridge roosts are particularly challenging to study owing to multiple access points). The plausible limits to the roost counts used in the current review did not overlap with the mean value of 16 bats reported by Jones et al. (1996), but those authors highlighted that their value was probably an underestimate, citing a nearby roost containing 60 females. Speakman et al. (1991) also reported a wide range of roost sizes: of four studied colonies, two had <10 bats, 1 had 40 and the other >100 individuals. The bat density estimates reported by these authors of 2 bats/km^2 (Jones et al., 1996) and 2.4 bats/km^2 (Speakman et al., 1991) is about half the central estimate given here, but falls within its plausible limits. The roost density estimates are likely to be underestimated in both the published literature and expert opinion, because a relatively low proportion of all roosts are in houses, and it is difficult to find roosts in trees, bridges and tunnels. Therefore, the true population size is likely to be somewhat higher than the lower limit presented here.

There is uncertainty about the sex ratio of the pre-parturition maternity colonies. Based on the literature, it appears likely that most pre-breeding roosts are very largely comprised of adult females (Lučan and Hanák, 2011). This provides additional justification for considering that the population size is at least as large as the central estimate.

The range reported here is likely to reflect the true distribution. The species has characteristic low flight over water that is readily recognised (notwithstanding some potential for confusion with *Pipistrellus spp*., which also frequently flies over water but usually at a greater height), and its echolocation calls are more distinctive than most other *Myotis* species.

It has not been possible to adjust the estimates for occupancy rates owing to a lack of data. Although some occupancy information relating to activity is available from the Bat Conservation Trust's Daubenton's bat field survey, this is limited to waterways where the species is relatively easy to identify. However, Daubenton's bats are also capable of using other habitat types, and travel in the wider landscape to reach roost locations. Population estimates therefore cannot be based on activity in riparian habitats alone.

Roost sizes estimated by experts were similar to those derived from our dataset. Based on experience of 35 roosts, their median estimate was 40 bats, with lower and upper plausible intervals of 20-100 (derived from the median of their estimates of lower and upper typical counts in good and poor habitat). Therefore, the very wide ranges may simply reflect high variability in true roost size for this species. If the values from experts had been substituted for those used in our calculations, there would be little change in the main population estimate or the lower plausible limit, but the upper limit would be reduced by about a quarter: England 2,066,343; Scotland 710,675; Wales 326,038; Britain 3,103,055.

The estimates of roost density were based on expert opinion alone, and may therefore introduce an unquantifiable error into the calculations.

Several main sources of error are identified. Firstly, there is uncertainty about roost size and the sex ratio in maternity colonies pre-parturition is poorly understood. Secondly, roost density is likely to be underestimated because of the difficulty of locating roosts in trees, bridges and tunnels. It is also unclear whether bat densities differ across habitat and geographical gradients for most of Great Britain. Finally, the species is likely to be under-recorded in non-riparian habitats, particularly in woodland, since in this environment its call parameters can be confused with other *Myotis spp.*

Table 10.7e Reliability assessment for Daubenton's bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**1**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 150,000 individuals was given in Harris et al. (1995) (England 95,000; Scotland 40,000; Wales 15,000), this estimate was graded as having very poor reliability. It lies within the plausible intervals given around the current estimate. The distribution is fairly similar to that shown in Arnold (1993).

The range is slightly smaller than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b).

Other evidence of changes through time

Table 10.7f Trends in Daubenton's bat activity from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)	Mean annual trend (%)
England	Hibernation	277	1993	42.8*†	0.3
	Waterway	**654**	**1996**	**-6.0**	**-0.4**
Scotland	Hibernation	n/a	n/a	n/a	n/a
	Waterway	112	1996	37.8*‡	2.0
Wales	Hibernation	99	1990	16	0.9
	Waterway	**46**	**1998**	**35.8***	**2.1**
Britain	Hibernation	401	1998	40.2*†	2.1
	Waterway	**822**	**1998**	**4.6**	**0.3**

* Indicates that the trend is significant (p<0.05).
† This result is heavily influenced by a strong increase in the index in 2015. Caution is advised until further data are available.
‡ There has been no change in the Scottish waterways population index since 2003, and the significant trend is strongly influenced by the selection of the baseline year (Magurran et al., 2010).

Table 10.7g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient		All countries*		

*Although the figures are comparable with those presented by Harris et al. (1995), both the original estimates and those presented here are scored as having extremely low reliability. The assessment is therefore based on the trends in activity recorded in the National Bat Monitoring Programme field survey.

Drivers of Change

Table 10.7h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Loss of roosts during works to bridges, tunnels and other structures.	Loss of roost location.		Negative
Alterations to water quality and riparian vegetation management.	Alteration in prey abundance.	Abbott et al. (2009) Racey et al. (1998) Vaughan et al. (1996)	Positive/Negative
Lighting of waterways and bridges.	Loss of foraging habitat and roosts, and increased fragmentation of suitable areas in landscape.	Fiona Mathews (*pers. obs.*)	Negative
Noise.	Reported negative impact of loud music on one studied maternity colony: national impacts of noise are possible but need investigation.	Shirley et al. (2001)	Negative
Effects of road casualties on local populations.	Collisions with vehicles.	Fensome and Mathews (2016)	Negative

Data deficiencies

Table 10.7i Areas where further research is required to improve the reliability of population size estimates and/or inform conservation management.

Data deficiencies	Habitat	Details
Density of roosts.	All	Very poor estimates available.
Roost and/or colony size.	Trees	Thermal imaging/infra-red video-photography and/or genetic approaches are needed to improve estimates, given that the species is crevice-dwelling and emerges late in the evening. Investigation of roost switching and colony structure would help identify the extent to which the colony is dependent on individual roosts.
Occupancy of riparian and non-riparian habitats.	All	Data on the proportion of occupied habitat are required throughout the species' range.
Effects of lighting of bridges and waterways on population viability.	All	No data available.
Impacts of road casualties, and fragmentation of landscapes by roads on British populations.	Roads	Road casualties are found in continental Europe (Fensome and Mathews, 2016).
Impacts of change in agricultural practice, particularly management of field margins and hedgerows, on prey abundance and local bat population sizes.	Agricultural land	No data available.
Greater understanding of the importance of swarming sites to gene flow.	Quarries, tunnels, potentially other habitats including woodland glades	Species shows considerable genetic mixing — possibly dependent on gene flow at swarming sites.
Impact of aquatic pollution on the population.	Riparian	Data on nitrogen enrichment are conflicting. There is no information on other pollutants affecting aquatic systems such as polycyclic aromatic hydrocarbons (PAHs) from roads.

Future prospects

Table 10.7j An assessment of the future prospects for the Daubenton's bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Unknown

10.8 Greater mouse-eared bat *Myotis myotis*

Habitat and roosting preferences

The diet of the greater mouse-eared bat in continental Europe is largely comprised of large Carabidae (ground beetles; 35%-65%) together with Lepidoptera (caterpillars), Melanotha *spp*. (cockchafers) and ground-dwelling Orthoptera (grasshoppers and crickets) (Arlettaz, 1996; Zahn et al., 2006). Because all these prey, except for cockchafers, are caught on the ground, the species tends to forage in deciduous woodland with little ground vegetation. Similarly, it will also take advantage of recently mown or grazed meadows and pasture, where the ground can be readily accessed (Zahn et al., 2006; Rudolph et al., 2009; Dietz and Keifer, 2016).

In central Europe, the species forms large maternity colonies mainly in large roof spaces, but occasionally in cellars and large bridges. In contrast, they are mainly found in caves in the Mediterranean region. Colonies make use of extensive areas (>1,000ha) for foraging, but the core areas are 1ha-15ha. These are usually found in a 5km-15km zone around the roost, and individuals may use several distinct areas within a night (Rudolph et al., 2009; Dietz and Keifer, 2016). Males tend to roost away from the maternity colony in a variety of structures. The species undergoes long-distance seasonal migration, moving between maternity, swarming and hibernation sites, frequently covering distances of 50km-100km. Only hibernation sites are known in Great Britain, and these are all in underground locations.

Status

Native.

Conservation Status

- IUCN Red List (GB: CR; England: [CR]; Scotland: n/a; Wales: n/a; Global: LC).
- This species has not been assessed for Article 17 of the EU Habitats Directive.

Species' distribution

Only a single, ringed, male is currently known. This individual has been recorded since 2002 in hibernation sites within close proximity of each other in West Sussex. The same locations were previously used by a hibernating population of up to 30 bats (Phillips and Blackmore, 1970), but this reduced to 1 male from 1985 to 1990. A small hibernating population, which probably always had fewer than 10 individuals, was discovered in Dorset in 1956, but was no longer present by 1980 (Blackmore, 1956). There are also isolated records of two other individuals: one male recorded in Kent in the winter of 1985 (thought likely to be a vagrant); and one old female found in Bognor, West Sussex, in January 2001.

Because of the limited distribution of records, no map is presented. It was also not possible to compute an alpha-hull encompassing the species' range.

Results

No estimate was made of population size or geographical range because only a single individual is known in Great Britain. Given the long-distance seasonal migrations made by the species, it is plausible that this animal is derived from a continental population. However, it is also possible that there are undiscovered summer roosts — of either maternity colonies or solitary males — in southern England. According to IUCN (2001), a species may only be declared extinct in the wild when exhaustive searches fail to find even a single individual.

Critique

Although extensive monitoring has been conducted at the hibernation sites where greater mouse-eared bats have been recorded in England, there have not been exhaustive searches of potential summer roosting locations or swarming sites.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993)

The population size is the same as that assessed by Harris et al. (1995), and the distribution is the same as shown by Arnold (1993).

Table 10.8a Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable		England		
	Decrease				
	Data deficient				

Drivers of Change

Table 10.8b Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
None known.			

Data deficiencies

Table 10.8c Areas where further research is required to improve the reliability of population size estimates and/or inform conservation management.

Data deficiencies	Habitat	Details
Roost and swarming site identification.	Trees and buildings	Exhaustive searches are required to demonstrate whether only a single individual is truly present in Great Britain. Focus should be around current and historical locations.
Identification of areas suitable for the species.	All	Given the potential for northward movement of this species, coupled with loss of range in other parts of the distribution because of climate change, habitat suitability for this species should be assessed to inform future conservation management plans.

Future prospects

Table 10.8d An assessment of the future prospects for the greater mouse-eared bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Decline*
Habitat	Stable

* The recent patterns have been of a decline in the species' range, and this will continue if the current population is lost. However, the European distribution of the species may move northwards because of the influence of climate change, as long as suitable habitat is available (Rebelo et al., 2010).

10.9 Natterer's bat *Myotis nattereri*

Habitat and roosting preferences

The Natterer's bat, *Myotis nattereri*, has a diet high in Diptera (flies) — particularly dungflies and midges — and these form 42%-60% of prey items (Shiel et al., 1991; Swift, 1997). It primarily forages in and around trees and hedgerows, and relies on very wide bandwidth calls to distinguish prey from vegetation clutter when hawking (Siemers and Schnitzler, 2000). Gleaning is also used extensively as a foraging technique, which may aid its niche separation from other *Myotis* species (Swift and Racey, 2002; Siemers and Swift, 2006). Most of its dipteran prey are diurnal and roost at night (Vaughan, 1997); and in one study in Ireland (Shiel et al., 1991), 68% of the diet was presumed to have been gleaned, including a high proportion of non-flying prey (e.g., 12% Aranea (spiders) and 5% Opiliones (harvestmen). Unlike the brown long-eared bat (*Plecotus auritus*), the Natterer's bat includes in its diet only a low proportion of Lepidoptera (moths; Shiel et al., 1991; Swift, 1997). This difference may reflect the bats' contrasting foraging strategies. Whereas the brown long-eared bat detects the fluttering of wings using passive listening, and relies on sight rather than echolocation at close range in order to avoid detection by tympanate (hearing) moths, the Natterer's bat relies on echolocation throughout its foraging activity (Swift and Racey, 2002).

The species is commonly associated with trees, particularly broadleaved woodland, but also makes use of tree-lined river corridors, trees in parkland, and hedgerows adjacent to pasture (Parsons and Jones, 2003; Smith and Racey, 2008; Zeale et al., 2016). It also forages over grass and thistles on roadsides (Swift, 1997), and uses mature Corsican pine plantations in Scotland (Mortimer, 2006). Maternity roosts are located in trees, bat boxes and buildings — predominantly in barns, churches and old dwelling houses (Smith and Racey, 2005). Although they tend to be situated within 500m of woodland, the size of the woodland does not appear important (Boughey et al., 2011).

There are three main sources of radio-tracking data from Great Britain for this species. One project was located in the Welsh Borders and studied bats using building roosts and natural tree roosts (Smith, 2001); another was conducted in a commercial forestry plantation in Fife and studied bats that used bat boxes and natural tree holes (Mortimer, 2006); and a recent project, investigating potential use of deterrents in situations where large colonies are damaging English churches, radio-tracked 48 Natterer's bats from 8 colonies (Zeale et al., 2016). While the behaviour of this latter group may not be entirely representative of the population, not least because colony sizes were very large (ranging from 30 to more than 150 individuals), it nevertheless provides some useful information. The Welsh study found the maximum distance travelled in a night when foraging was 5.5km for adult females and 6.7km for adult males, and the colony home ranges were 11 km^2-13km^2, but the core foraging areas for adults lay within 3km-5km of the roost (Smith, 2001). This compares with colony home ranges of 4.4 km^2-6.5km^2 in Fife (Mortimer, 2006), and 1 km^2-25km^2 in the English church study (Zeale et al., 2016). The core foraging areas in these two studies were 100m-4.2km, and 1.4km-7.7km, from the roost respectively. Evidence to support the exclusive use of core foraging areas by a colony, and of discrete core foraging areas for individual animals, was provided by all projects. Roost switching occurred very frequently in all roost types (every 2-7 days): in the case of churches, movements were usually to locations within the same building, although there were also some records from trees close to foraging grounds.

Natterer's bats are the most commonly recorded species at swarming sites in Great Britain, and the catchment areas for these sites are large (20-60km radius; Parsons and Jones, 2003; Rivers et al., 2005; Glover and Altringham, 2008). There is evidence for high natal philopatry, and therefore genetic interchange associated with swarming sites is extremely important for Natterer's bat conservation (Rivers et al., 2005). The species is generally considered to be non-migratory across Europe (Dietz & Kiefer 2016). Underground sites including tunnels, caves and ice-houses are used for hibernation, but the extent of use of trees is unclear (Smith, 2001; Dietz and Keifer, 2016). Natterer's bats emerge regularly from hibernation, with torpor lasting from 1-20 days, with individuals in poorer body condition arousing more frequently (Hope and Jones, 2012). Habitat quality around hibernacula is therefore likely to be very important to the conservation of this species.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A species' distribution map is presented in Figure 10.9a. Gaps in the species' distribution are likely to reflect areas with low survey effort, rather than true gaps in the species' range.

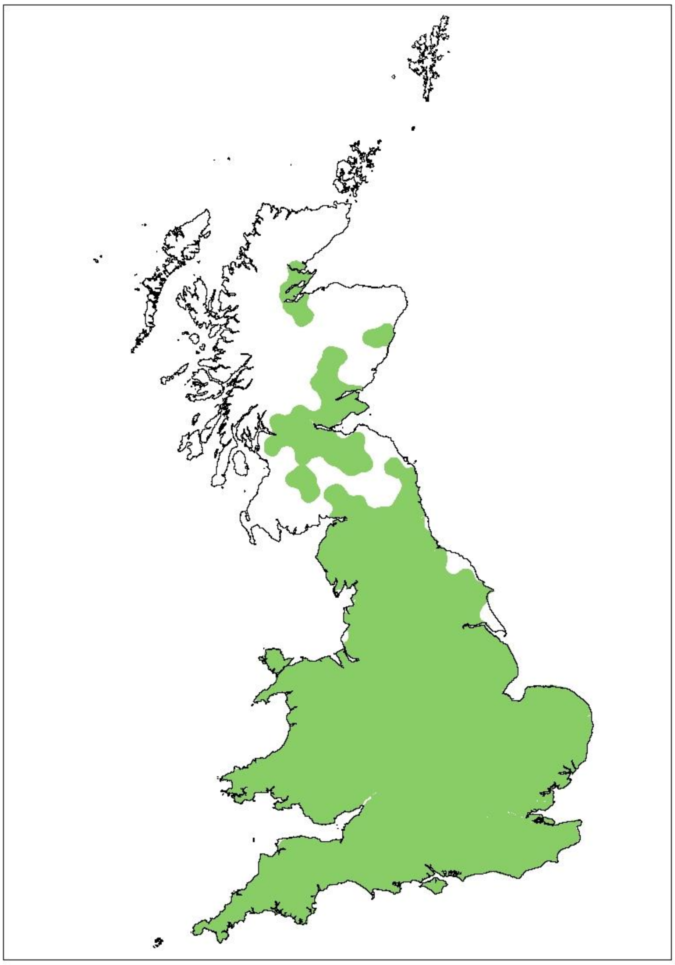


Figure 10.9a Current range of the Natterer's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records (for example, two roosts known in Fort Augustus, Scotland) may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Information was available from 124 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The most recently available peak count before July was used for the analyses. Small roosts were not excluded from the assessment because the fission-fusion social structure of the species means that colonies are divided across several roosts: even those locations with <10 bats can include breeding individuals. The median pre-breeding roost size for Natterer's bats derived from the available datasets was 23.5 individuals (95%CI = 16-35, range = 1-194, n=124 roosts).

Little information on the sex ratio of pre-parturition maternity colonies was available. Maternity roosts in the Welsh and Scottish studies were described as being largely comprised of adult females (Smith, 2001; Mortimer, 2006). An extensive ringing study of 11 social groups using bat boxes in a broadleaved woodland in southern England found that all colonies were mixed sex and 72% of the bats were female (August et al., 2014). Expert opinion was obtained from 6 individuals. A further 2 experts responded to requests for input on this species but were unable to provide the information necessary for the calculation of population size. No expert was able to provide information on the sex ratios of the population as a whole, and this information was not available from Harris et al. (1995).

Only one estimate for roost density was available from experts (typical density 0.01 roosts/km^2, plausible range 0.01-0.02 roosts/km^2), and it had a very low reliability score (3/10). Therefore, values were based on the only published data on maternity roosts. Harris (2014) reported a study of building roosts and bat boxes studied over a 10-year period in the Cotswold Water Park (100km^2). Here a density of 0.23 roosts/km^2 was found. In a study of a 2500km^2 area of North Yorkshire, a roost density of 0.06 roosts/km^2 (all in buildings) was reported (Jones et al., 1996). The estimates in Jones et al. (1996) were based on an assumption that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). Given that no data were available to verify this assumption, a second density estimate was derived for the purpose of the current calculations by using the entire 2,500km^2 study area (which gives a density estimate of 0.004 roosts/km^2). The highest and lowest values of the available estimates in the literature were used to define plausible roost densities in good and poor quality habitats. The roost density obtained from expert opinion fell within the ranges given in the published literature.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

The population estimate was calculated on the basis of adult bat density and the geographical range. Density was calculated as follows:

Adult bat density (bats/km^2)

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*
*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*
*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is a typical maternity roost in the pre-parturition period. n. is the number of adults.
‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

For comparative purposes, bat densities estimated directly from radio-tracking studies were also considered. There was one available study for mixed habitat. Here a density of 5.8 adult bats/km^2 was reported in a Welsh population tracked from buildings and natural roosts, based on observations of 2.9 adult females/km^2 (Smith, 2001).

The population estimate was based on adult population density and extent of occupancy across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density (bats/km^2) * total area within range (km^2)*
*Lower limit = Lower limit adult density (bats/km^2) * total area within range (km^2)*
*Upper limit = Upper limit adult density (bats/km^2) * total area within range (km^2)*

A separate set of population estimates was also made, based on resident bat densities within woodland. Unlike most other tree-dwelling species, there is some evidence from radio-tracking that natural roosts are primarily located within woodland blocks rather than in individual trees in hedgerows or parkland (Smith, 2001; Mortimer, 2006). Because limited data were available, no attempt was made to derive separate estimates for broadleaved and coniferous woodland. Extrapolation to total population size was based on the observation that 65%-69% of roost locations identified in radio-tracking studies were in natural tree crevices rather than buildings or bat boxes (Smith, 2001; Mortimer, 2006). Some caution is required with this extrapolation, as the sample sizes are relatively small, and it may not necessarily follow that the proportion of bats roosting in trees is the same as the proportion of roost locations found in trees.

In two different regions within a Scottish population, densities of 20 adult bats/km^2, and 50 adult bats/km^2, were reported (Mortimer, 2006): these radio-tracked animals used bat boxes and natural roosts. In a well-studied population using boxes in a lowland woodland in southern England (largely broadleaved), 37 adult bats/km^2 have been reported (Danielle Linton, *pers. comm.*). These values were therefore used as the central estimate and upper and lower plausible limits in good and poor habitat.

The total population size, based on estimates in woodland alone, was calculated as follows:

*Total Adult Population = Median adult density in woodland habitat (bats/km^2) * total woodland area within range (km^2) *(1/median proportion of roosts in trees)*[a]

*Lower limit = Lower limit adult density in mixed habitat (bats/km^2) * total habitable area within range (km^2) * (1/lower limit proportion of roosts in trees)*[a]

*Upper limit = Upper limit adult density in mixed habitat (bats/km^2) * total habitable area within range (km^2) * (1/upper limit proportion of roosts in trees)*[a]

[a] Multiplication by the inverse of the proportion of roosts found in trees generates an estimate for all roosts (not just those in trees).

Results

The values used to derive the density estimates are shown in Table 10.9a.

Table 10.9a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	23.5 (16-35)
Sex ratio	0.9 (0.72*-1)
Maternity roost density (roosts/km^2)	0.06** (0.004**-0.23†)
Proportion of roosts in trees	0.67 (0.65-0.69)
Direct estimate of adult bat density in mixed habitat (bats/km^2)	5.8††
Direct estimate of adult bat density in woodland (bats/km^2)	37‡ (20-50‡‡)

* August et al. (2014).
**Jones et al. (1996).
† Harris (2014).
†† Smith (2001).
‡ Danielle Linton (*pers. comm.*).
‡‡ Mortimer (2006).

Population estimation and range

Table 10.9b Area within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales and the whole of Britain. The table below presents two alternative estimates, one based on mixed habitat, and one based on an extrapolation from woodland: the values are therefore alternatives and should not be summed.

Basis*	**Country**	**Area within range (km^2)**	**Bat density (adults/km^2)**			**Adult population size**		
			Estimate	**Plausible intervals**		**Estimate***	**Plausible intervals***	
				Lower	**Upper**		**Lower**	**Upper**
Mixed habitat	England	126,500	2.5	0.1	6.1	321,000	11,700	2,040,000
	Scotland	16,200	2.5	0.1	6.1	41,000	1,490	260,000
	Wales	20,600	2.5	0.1	6.1	52,000	1,900	332,000
	Britain	**163,300**	**2.5**	**0.1**	**6.1**	**414,000**	**15,100**	**2,630,000**
Wood-land	England	11,800	37	20	50	654,000	343,000	911,000
	Scotland	3,100	37	20	50	171,000	89,700	238,000
	Wales	2,680	37	20	50	148,000	77,700	206,000
	Britain	**17,600**	**37**	**20**	**50**	**973,000**	**510,000**	**1,360,000**

* For estimates based on woodland area, population sizes account for the likely proportion of total roosts in natural tree crevices. Differences in column/row totals are because of rounding.

The adult bat densities derived by Smith (2001) from radio-tracking fell within the plausible intervals derived from the combination of roost size and roost density. Therefore, no additional calculations were performed.

The estimates in the Article 17 Report on Natterer's bat status 2007-2012 (Joint Nature Conservation Committee, 2013a) are considerably lower than those estimated here (beyond the lower plausible limit) (see Table 10.9c).

Table 10.9c Article 17 Report on Natterer's bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	70,000	70,000
Scotland	17,500	17,500
Wales	12,500	12,500
Britain	**100,000**	**100,000**

Note: maximum and minimum estimates were the same values for this species.

The geographical range for the species, based on known records of Natterer's bats since 1995, is shown in Table 10.9d.

Table 10.9d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	127,000	n/a
Scotland	16,200	n/a
Wales	20,600	n/a
Britain	**163,000***	**216,000**

*Total does not sum because of rounding errors.

Critique

The very large range of plausible values, and the extreme alterations that could be generated by basing estimates on woodland rather than building roosts, emphasise the uncertainty around all estimates for this species. There was little information on which to base calculations of adult bat density, with uncertainty about roost density being the major source of uncertainty. The only alternative source of information available in mixed habitats suggested an adult bat density that equalled the upper estimate from our calculations. It is therefore likely that the population size is towards the upper rather than the lower end of the ranges presented.

The roost size estimated from the available dataset was slightly larger than the mean value of 16.5 (SE = 2.5) reported by Mortimer (2006) for bat boxes and natural tree roosts, but this may reflect the tendency for smaller group sizes in bat boxes. (In a well-studied broadleaved woodland in southern England where >775 occupied bat box records are available, roost sizes of 10 are typical (Danielle Linton, *pers. comm.*)). No data at all were available for tree-roosts, so it is possible that these differ substantially from building or bat-box roosts.

For comparative purposes, population estimates were also derived on the basis of minimum bat densities in woodland. Extrapolations to all habitats were made using data suggesting that 65%-69% of roosts used by Natterer's bats are in trees within woodland. The main source of error with this approach is that the two estimates of bat density in woodland were derived from locations with very well-established and extensive bat box schemes, and it is unclear whether the presence of bat boxes artificially increases bat density compared with other woodlands. The data were also derived from just two woodlands, presumably selected for detailed research on the basis of having substantial bat populations. Therefore, although the two woodlands gave reasonably similar density estimates, it is unclear whether these can be generalised to other areas. Whether bats roosting within woodland make extensive use of other habitats is unknown. If they do, and they exclude other individuals from these regions, then the effective density may be much lower than that estimated on the basis of woodland area alone. The plausible intervals from these approaches overlap. The upper limit from the first approach — which used evidence from two separate sources (Smith, 2001; Harris, 2014) to estimate plausible densities — is higher than that derived from woodland. Therefore, the overall conclusion must be that the population is likely to be greater than 400,000 individuals, and possibly much higher.

The range in Scotland, particularly in the west and Borders, may be more extensive than estimated here. This is partly owing to lower recording effort in Scotland, but also because tree-roosts are critically under-recorded. Further, acoustic surveys are not reliable because the call parameters of Natterer's bat overlap with those of other *Myotis* species (Russ, 2012). No expert could provide estimates of roost density that they considered robust (one provided an estimate, but with a confidence score of 3/10), and no expert had any information on tree-roosts. This emphasises the potential for distributions and densities to be underestimated in this report.

Six experts provided information on roost size, whilst the other two had no information to contribute. Their estimates of usual roost counts (usual size 29; typical range 20-60, n=71) were larger than those derived here, possibly because they combined data from bats in boxes that were known to be part of the same colony. Nevertheless, they lie within the plausible values (10-50) used in this report.

There is some discrepancy between the sex ratios reported in the literature in pre-breeding maternity roosts and the experience of experts. Two reported that >80% of individuals captured from roosts were female, whilst the other experts were uncertain.

Several important sources of error are identified. Firstly, no roost counts or density estimates are available for natural tree roosts. The ratio of building:tree roosts is founded on very limited data. As a result, the scale of bias introduced by basing estimates primarily on data from buildings is unquantifiable. There is also uncertainty about the sex ratio of bats in maternity roosts pre-parturition. The range may be underestimated in some parts of Scotland, particularly where there is little potential for roosts in buildings, as it is difficult to identify Natterer's bats with certainty using acoustic surveys, and tree roosts are difficult to find. Finally, the extent to which Natterer's bats from woodland use adjacent habitat for foraging, and whether this use excludes other colonies roosting outside the woodland, is unknown. As a consequence, it is difficult to extrapolate density estimates from focal woodlands to the wider landscape.

Table 10.9e Reliability assessment for Natterer's bats. Scores are based on the availability of roost location data, roost count data and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	1
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates†	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**2**

* Either from the literature or from expert opinion with high reliability scores.
† No evidence on roost size is available for tree roosts.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2012)

Although a population estimate of approximately 100,000 individuals was given in Harris et al. (1995) (England 70,000; Scotland 17,500; Wales 12,500), this estimate was graded as having very poor reliability and was largely derived from expert opinion on the ratio of Natterer's to pipistrelle bats (roosts and individuals). Direct comparison is therefore not possible.

The distribution is similar to that reported by Arnold (1993). The range is slightly smaller than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b); this difference is likely to reflect the differing methodologies.

Other evidence of changes through time

The National Bat Monitoring Programme hibernation and roost count data do not indicate any change over time. No data are available from field surveys.

Table 10.9f Trends in Natterer's bat activity from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available for Scotland to estimate trends. Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Hibernation**	**347**	**1999**	**116.6***	**5.0**
	Roost	68	2002	2.8	0.2
Scotland	Hibernation	n/a	n/a	n/a	n/a
	Roost	n/a	n/a	n/a	n/a
Wales	Hibernation	143	1999	94.4*	4.2
	Roost	n/a	n/a	n/a	n/a
Britain	Hibernation	**512**	**1999**	**84.6***	**3.9**
	Roost	81	2002	-11.2	-0.9

* Indicates that the trend is significant (p<0.05).
† The baseline year was set as 2001 because few roosts were monitored before this date.

Table 10.9g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of Change

Table 10.9h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Increased availability of broadleaved woodland and bat boxes.	Increased roosting opportunities (4.7% increase in broadleaved woodland, and 6.4% increase in coniferous woodland, between 1990 and 2007).	Countryside Survey 2007 (Carey et al., 2008).	Positive
Loss of viable roosts during barn and other building conversions.	Reduction in roost suitability, particularly reduction in the loft area.	Briggs (2000)	Negative
Urban development encroaching on traditional roosts.	Loss of foraging habitat and increased isolation of woodland fragments in the landscape.	Boughey et al. (2011)	Negative
Impact of road casualties on local populations.	Collisions with vehicles.	Fensome and Mathews (2016)	Negative
Artificial night lighting.	Species is extremely light-shy; artificial light at roosts is highly damaging. Lighting potentially severs commuting routes and reduces moth availability.	Zeale et al. (2016) Plummer et al. (2016)	
Change of prey abundance in agricultural landscape, caused by habitat change and effects of avermectins on dung flora.	Dung flies are a key prey item.	Swift (1997)	Negative

Data deficiencies

Table 10.9i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts.	All	No data are available in woodlands. Density is poorly estimated in other habitats.
Proportions of roosts found in trees compared with buildings.	n/a	No data available. Information is required to assess any bias introduced by deriving estimates from roosts in buildings, and to assess the conservation importance of woodlands.
Roost size in trees and buildings.	Buildings and trees	Thermal imaging/infra-red video-photography and/or genetic approaches would improve estimates, given that the species is crevice-dwelling and emerges late in the evening. Intensive radio-tracking of bats in building roosts would identify whether the colony is divided across multiple roosts.
Effects of cumulative pressures of land use change and urban encroachment on roosts.	All	No data available.
Impacts of road casualties on British populations.	Roads	No data available.
Impacts of change in agricultural practice, particularly management of field margins and hedgerows, on prey abundance and local bat population sizes.	Agricultural land	No data available.
Impacts of changing woodland management, affecting the total woodland area and the amount of standing deadwood, on roost availability.	Broadleaved woodland	No data available.
Effectiveness of mitigation for development in maintaining the functionality of roosts in buildings.	Buildings	Very limited data available.

Future prospects

Table 10.9j An assessment of the future prospects for the Natterer's bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Decline

10.10 Serotine bat *Eptesicus serotinus*

Habitat preferences

The serotine bat, *Eptesicus serotinus*, is often associated with pasture and parkland. With slow, highly manoeuvrable flight it can fly very close to the ground as well as among the canopies of trees. It preys mainly on large Coleoptera (beetles), including *Aphodius spp*. (dung beetles) and *Melonotha spp*. (cockchafers), and on larger Lepidoptera (moths) (Robinson and Stebbings, 1993; Vaughan, 1997). Many Diptera (flies), including dung flies, and small prey items are also eaten, particularly early in the season (Catto et al., 1994). Prey is taken in flight and eaten on the wing, but capture from the ground has also been reported anecdotally.

The foraging range of the species is relatively large, with average commutes of 6.5km recorded in a pastoral region (Catto et al., 1996), and 8km in a more arable region of southern England (Robinson and Stebbings, 1997). The maximum distance recorded was over 41km, and the bats commonly commuted along hedgerows and treelines and over pasture. Individual home ranges appear highly variable e.g., 0.16km^2-47km^2, and there was considerable overlap, even of core areas, between individuals (Robinson and Stebbings, 1997).

Maternity colonies are thought to be almost exclusively formed by adult females, with males roosting separately or in small groups (Catto, 1993; Moussy et al., 2015). Radio-tracking data indicate that females are faithful to a roost during the breeding season, whereas males use several alternative roosts (Catto et al., 1996). Maternity roosts are almost exclusively located in buildings, particularly residential houses constructed in the late 19th and early 20th century which have high gables and a substantial roof-space. They are found only very occasionally in bat boxes. Roosts are closer to woodland, water and pasture than would be expected by chance — although studies differ in the spatial scale at which these effects are seen (Battersby, 1999; Boughey et al., 2011; Tink et al., 2014).

Across Europe, the species is generally considered sedentary, despite its capacity for strong flight and relatively large nightly movements. In the south east of England, a large ringing study did not generate any recaptures at distances >10km (Hutson et al., 2008). In continental Europe, most hibernation sites are within 50km of the summer roost (Dietz and Keifer, 2016). Little information exists on the hibernation sites used by the species, and only very few individuals are found in underground hibernacula. It is presumed that most remain in roof spaces and cavity walls (Dietz and Keifer, 2016).

Status

Native.

Conservation Status

- IUCN Red List (GB: VU; England: [VU]; Scotland: n/a; Wales: [VU]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Unknown; Scotland: n/a; Wales: Unknown).

Species' distribution

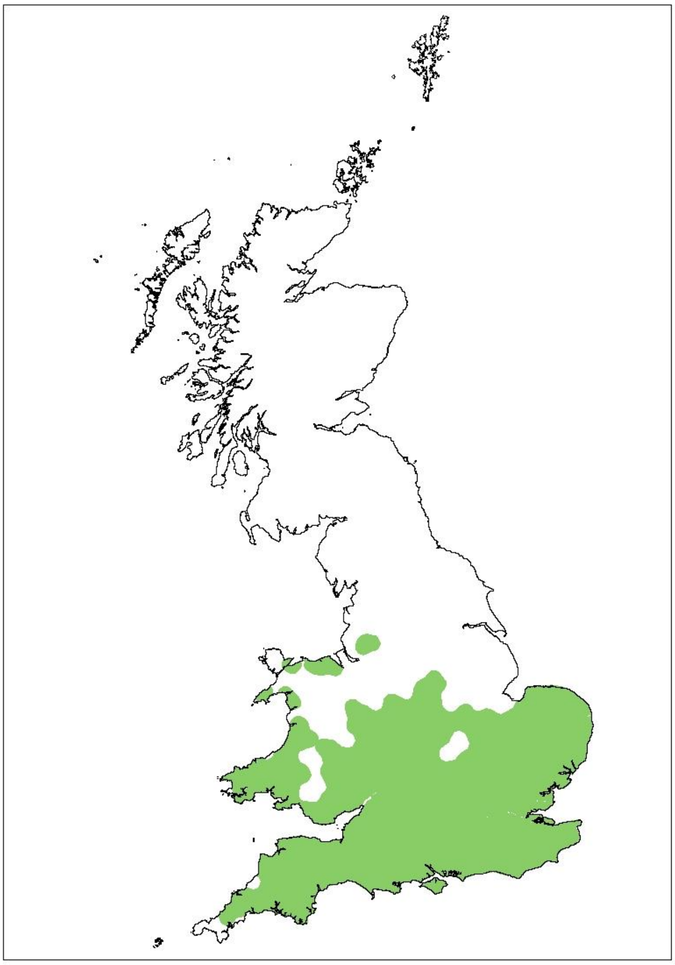


Figure 10.10a Current range of the serotine bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

There appears to be a distinct structuring of the population in Great Britain, in contrast with continental Europe: based on population genetics, three populations in the south of England — east, west, and the Isle of Wight — have been identified, and these have only low levels of gene flow (Smith et al., 2011; Moussy et al., 2015). There is some evidence from bat workers, as well as from population genetics, of a westward expansion of the population, possibly corresponding with a decline in the east (Moussy et al., 2015). Genetic evidence also suggests that there must be some gene flow across the English Channel (Moussy et al., 2015). Females are highly philopatric, according to both ecological observations and molecular analyses (Harbusch and Racey, 2006; Moussy et al., 2015). Gene flow is therefore likely to be mediated by male dispersal, and possibly by the use of mating or swarming sites. It has been proposed that the much greater structuring of populations seen in serotine compared with Daubenton's bat populations could reflect a lack of mixing at swarming sites (Smith et al., 2011). However, the species is regularly captured at a range of different swarming sites in southern England (Parsons et al., 2003; Fiona Mathews, *pers. obs.*), and it is currently unclear where mating takes place. It is therefore possible that swarming sites are important for gene flow in this species, but that the catchment area is smaller than for the Daubenton's bat.

Species-specific methods

Information was available from our datasets for 122 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses. Small roosts were not excluded from the assessment because the fission-fusion social structure of the species means that colonies are divided across several roosts: even those locations with <10 bats can include breeding individuals. The median pre-breeding roost size calculated from the available data was 15 individuals (95%CI = 10-19, range = 1-287, n=122 roosts).

Expert opinion was obtained from 5 individuals. A further 4 experts responded to requests for input on this species but were unable to provide the information necessary for the calculation of population size. No expert was able to provide information on the sex ratios of the population as a whole, and this information was not available from Harris et al. (1995). One expert indicated that maternity colonies pre-parturition were 100% female. The literature also provided support for colonies being exclusively female, so it was assumed that there was no uncertainty in this variable.

Only one roost density estimate was provided by an expert and this was scored as unreliable (score 4/10). Data were therefore derived from 4 published studies, each of which covered a wide geographical region (Robinson and Stebbings, 1997; Battersby, 1999; Harris, 2014; Tink et al., 2014). Two separate values were derived from Battersby (1999); one was based on roosts recorded in a Natural England database, and the proportion of active maternity roosts was estimated by revisiting a proportion of these sites; and the other was derived from an extrapolation of surveys of randomly selected buildings. The value from Tink et al. (2014) was the total number of maternity colonies recorded in the study area rather than the somewhat higher estimates presented in that study for kernel densities in areas of high prevalence; and the value from Robinson and Stebbings (1993) was derived from back-extrapolation of bat density data based on the colony sizes observed in that study. The median of the published values (0.5 roosts/km^2) (Robinson and Stebbings, 1997; Battersby, 1999; Harris, 2014; Tink et al., 2014) was used as the central estimate. The highest and lowest values of the available estimates in the literature were used to define plausible roost densities in good and poor quality habitats.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

Adult bat densities (bats/km^2) were calculated as follows:

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*
*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*
*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is a typical maternity roost in the pre-parturition period. n. is number of adults.
‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The population estimate was based on adult population density and extent of occupancy across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density in mixed habitat (bats/km^2) * total area within range (km^2)*
*Lower limit = Lower limit adult density in mixed habitat (bats/km^2) * total area within range (km^2)*
*Upper limit = Upper limit adult density in mixed habitat (bats/km^2) * total area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.10a.

Table 10.10a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	15 (10-19)
Sex ratio	1
Maternity roost density (roosts/km^2)	0.5 (004*-0.12**)

*Tink et al. (2014).
** Battersby (1999) maximum estimate of maternity colony density, derived from re-visiting roosts identified by the English Nature dataset.

Population estimation and range

Table 10.10b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	78,100	1.5	0.1	4.6	117,000	6,250	356,000
Scotland	0	1.5	0.1	4.6	0	0	0
Wales	12,500	1.5	0.1	4.6	18,700	1,000	57,000
Britain	**90,600**	**1.5**	**0.1**	**4.6**	**136,000**	**7,250**	**413,000**

The Article 17 Report on serotine bat population size 2007-2012 is shown in Table 10.10c (Joint Nature Conservation Committee, 2013b).

Table 10.10c Article 17 Report on serotine bat population size 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	14,800	14,800
Scotland	0	0
Wales	250	250
Britain	**15,000**	**15,000**

Note: maximum and minimum estimates were the same values for this species.

The geographical range estimate for the species is based on known records of serotine bats since 1995 and is shown in Table 10.10d.

Table 10.10d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	78,100	n/a
Scotland	0	0
Wales	12,500	n/a
Britain	**90,600**	**100,200**

Critique

The plausible range of estimated population size for serotine bats is wide. Mainly, this reflects uncertainty about maternity roost density. The lowest plausible value (0.004 roosts/km^2) was derived from Tink et al. (2014). It is likely to be an underestimate since it was based on data collated by a local biological records centre, and only 15 of 97 roosts were specified as ‘maternity’ but some of the remaining 82 roosts could also have been breeding sites. The next lowest value of 0.01 (Harris, 2014) would increase the population estimate to 15,616 in England, 2,500 in Wales and a total of 18,116 in Great Britain. All of the other estimates of serotine bat roost density available from the literature ranged between 0.04 and 0.12 roosts/km^2. These studies were all conducted within known strongholds for the species, and are therefore likely to be somewhat higher than those expected elsewhere: the median value of 0.05 roosts/km^2 used as the typical roost density appears reasonable as an estimate of density across the range.

The calculated density total of bats/km^2 (1.5, plausible range 0.1-4.6) corresponds with the estimate of 1.7 given by Robinson and Stebbings (1997). These estimates overlap with those based on random building surveys in Battersby (1999), but are lower than those derived from adjusting the number of known roosts in an English Nature Database for the proportion likely to be active maternity colonies.

The range reported in the current review is likely to reflect the true distribution. The species is almost entirely dependent on building roosts and its droppings are distinctive. Therefore, despite being inconspicuous at its roost sites — colonies are small and individuals tend to be hidden in crevices — it is nevertheless well-recorded compared with many bat species that rely primarily on tree roosts rather than buildings. It also has a loud echolocation call with fairly distinctive call parameters (although there is some potential for confusion with other Nyctaloid bats, particularly on heterodyne detectors).

The values for roost counts by experts differed from those used in the calculations above. Based on their experience at 55 roosts, the median roost count reported at typical roosts was 27 individuals (plausible intervals (PIs) = 11-68, derived from the median of their estimates of lower and upper typical counts in good and poor habitat). This compares with the value of 15 individuals (PIs = 10-19) derived from the literature. Therefore, the typical value, and the upper plausible value in good habitat, are higher than the value used here. This difference may reflect the tendency of experts to be aware of larger roosts; the values used in our calculations were derived from a range of sources, including European Protected Species Licence Applications, and may therefore more closely represent the true roost sizes typically encountered in buildings. If the values from experts had been substituted for those used in our calculations, the estimates for Britain would increase to 244,567 individuals (PIs = 7,971-1,478,274).

The main sources of error in the current review relate to defining plausible upper and lower limits; there is reasonable confidence about the values used to derive the typical estimate. These errors are: uncertainty about roost density; and lack on information on variability in roost densities or colony sizes across habitat and geographical gradients. The populations in the east and west of England seem distinct, and yet few data are available for the west of England or Wales.

Table 10.10e Reliability assessment for serotine bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	
	1	A few (4 to 6)	1
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	
	1	<150 roosts	1
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**3**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 15,000 individuals was given in Harris et al. (1995) (England 14,750; Scotland 0; Wales 250), this estimate was graded as having very poor reliability. The distribution estimated in the current review is considerably larger than that shown in Arnold (1993), with the range spreading west and north to include the south west of England, the Midlands, the Welsh Borders and Merseyside. It is unclear how much this reflects a true range change rather than increased observer effort; and occupancy is thought to be low in some of these new areas. There are also expert opinion reports of declining populations in the east of England.

The range is larger than that given in the JNCC Article 17 Report (Joint Nature Conservation Committee, 2013b).

Other evidence of changes through time

The National Bat Monitoring Programme field survey and roost count data are suggestive of recent declines. However, sample sizes are relatively small, and the trends are not statistically significant. In addition, serotine bats can be easily confused with other Nyctaloid bats when detection is based on heterodyne detectors: the primary technique used in the

NBMP field survey. Nevertheless, changes in agricultural practice and reductions in prey abundance, particularly in the east of England, may be expected to lead to a decline.

Table 10.10f Trends in serotine activity from baseline to 2015 as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available for Wales to estimate trends. Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Field**	**416**	**1998**	**-9.7**	**-0.6**
	Roost	101	1996	-26.5	-1.9
Scotland	Field	n/a	n/a	n/a	n/a
	Roost	n/a	n/a	n/a	n/a
Wales	Field	n/a	n/a	n/a	n/a
	Roost	n/a	n/a	n/a	n/a
Britain	**Field**	**450**	**1998**	**-9.5**	**-0.6**
	Roost	102	1998	-22.1	-1.6

* Indicates that the trend is significant (p<0.05).
† Percentage change since the 1999 baseline.

Table 10.10g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Considered data deficient for range because of expert opinion as well as uncertainties relating to the identification of acoustic data.

Drivers of Change

Table 10.10h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Legal protection of roosts.	The species is strongly dependent on building roosts, so is likely to benefit from increased legislative protection.	None available	Positive
Agricultural intensification, decline of pastoral farming, and use of anthelmintics and pesticides.	A reduction in prey availability, particularly that associated with dung.	Catto et al. (1994)	Negative
Climate change and weather fluctuations.	High juvenile fatality rates in first few months of life, so the species is likely to be vulnerable to poor summer weather.	Harbusch and Racey (2006) Chauvenet et al. (2014)	Negative
Alterations to roost conditions in buildings, including the use of breathable roofing membranes.	High dependency on building roosts and crevice-dwelling nature makes the species vulnerable, and there are many case reports of entanglement.	Waring et al. (2013)	Negative

Data deficiencies

Table 10.10i Areas where further research is required to improve the reliability of population size estimates and/or inform conservation management.

Data deficiencies	Habitat	Details
Density of roosts.	All	Very poor estimates available.
Occupancy data at the edge of ranges.	All	Systematic monitoring at the edges of the species' range would help determine whether the range is truly expanding.
Impacts of change in agricultural practice, particularly management of field margins and use of avermectins, on prey abundance and local bat population sizes.	Agricultural land	No data available.
Effects of cumulative pressures of land use change and urban encroachment on roosting and foraging areas.	All	No data available.

Future prospects

Table 10.10j An assessment of the future prospects for the serotine bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Increase
Habitat	Decline

10.11 Leisler's bat *Nyctalus leisleri*

Habitat preferences

The Leisler's bat, *Nyctalus leisleri*, is a fast-flying species capable of long-distance flight. It is sympatric throughout most of its global range with the larger noctule bat: similarities in echolocation patterns and behaviour mean that the two species are frequently confused, although the Leisler's bat is generally considered rarer (Dietz and Pir, 2009). Ireland, where the noctule bat is absent, is a stronghold for the species, whereas validated records derived from bats identified in the hand or by DNA analysis of droppings are relatively infrequent in Great Britain. The Leisler's bat feeds on the wing, and tends to fly lower than noctule bats whilst foraging. Its diet is mainly comprised of small and medium-sized insects including Diptera (flies) — particularly Chironomids (midges) and Scathophagidae (dung flies) — Lepidoptera (moths), and Coleoptera (beetles). However, there appears to be a regional variation in diet both within Great Britain and internationally, depending on whether the animals are foraging near to water, in cattle-grazed areas or adjacent to woodland (Shiel et al., 1998; Waters et al., 1999; Kaňuch et al., 2005). The species emerges early, particularly during lactation (Shiel et al., 1998; Waters et al., 1999), and is one of few species of bats for which there is clear evidence of higher activity at mercury-vapour and high-pressure sodium streetlights than in dark control areas (Mathews et al., 2015).

There are few studies on the foraging behaviour of the species in Great Britain. In Ireland, foraging flights of up to 13km have been recorded (Shiel and Fairley, 1999), whereas at sites studied in southern England, most foraging occurred within 4km of the roost (Waters et al., 1999). Pasture appears to be a preferred foraging habitat in both Great Britain and Ireland (Shiel and Fairley, 1999; Waters et al., 1999), although there is some evidence from Northern Ireland of avoidance of improved grassland (Russ and Montgomery, 2002). Use is also made of woodland edges and tree-lined roads (Waters et al., 1999; Russ and Montgomery, 2002).

Summer roosts are usually located in buildings in Great Britain and Ireland, in contrast with parts of Europe where the species is predominantly tree-dwelling (Dietz and Keifer, 2016). Bat boxes area also used, particularly outside the maternity period (Collin, 1995; Jim Mullholland, *pers. comm.*). The hibernation preferences of the species are not well known in Great Britain. In Northern Ireland, radio-telemetry has indicated that hibernacula are found exclusively in trees (Hopkirk and Russ, 2004). The species is considered migratory in continental Europe, with long-distance movements taking place between maternity and hibernation sites (Wohlgemuth et al., 2004; Dondini et al., 2013; Moussy et al., 2013). Recent molecular evidence indicates that the British-Irish population belongs to a separate lineage from that found in the rest of Europe, and no contemporary gene flow occurs (Boston et al., 2015). However, it is unclear whether there is any long-distance movement of individuals either within Great Britain or between Britain and Ireland.

Status

Native.

Conservation Status

- IUCN Red List (GB: NT; England: [NT]; Scotland: [NT]; Wales: [NT]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Unknown; Scotland: Unknown; Wales: Unknown).

Species' distribution

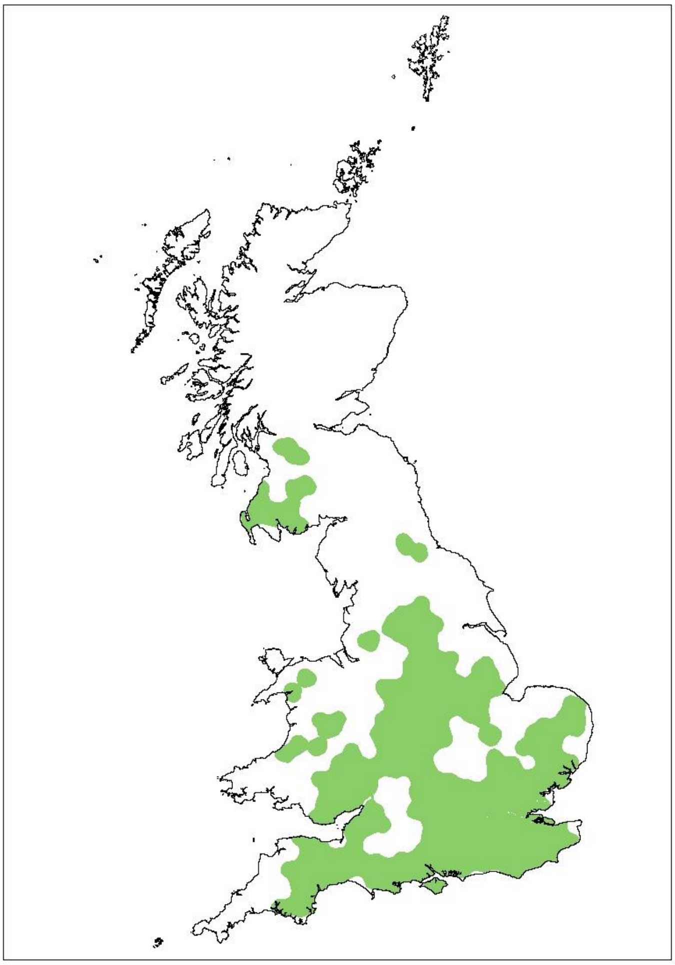


Figure 10.11a Current range of the Leisler's bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Data were only available for 2 maternity roosts from our datasets. The median of the most recently available peak counts before July was used for the calculation of population size (median = 64 individuals, 95%CI = 14-114, range 14-114, n=2 roosts). Small roosts were not excluded from the assessment because the minimum group size for breeding is unclear for this species.

No data were available from the literature or from Harris et al. (1995) on the sex ratio of maternity roosts pre-breeding. Expert opinion was obtained from 1 individual for this species, but this respondent was unable to provide an estimate of the sex ratio: a further 5 experts responded to requests for input on this species but were unable to provide the information necessary for the calculation of population size. No information was available from the literature or expert opinion on the sex ratio of the population. No data were available from experts or the literature on the density of maternity roosts. It was therefore not possible to compute population size.

Habitable area was defined as the entire extent of occupancy. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

The upper and lower limits for the plausible intervals for roost size were defined as the upper and lower 95% confidence limits for the median.

Results

The values available for the calculation of density estimates are shown in Table 10.11a.

Table 10.11a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	64 (14-114)
Sex ratio	n/a
Maternity roost density	n/a

Population estimation and range

Given the absence of data on roost density, it was not possible to calculate a population estimate. As it is considered unlikely that most maternity roosts in Britain are known, it was also not possible to make a total count. No population genetics study has been conducted, and so no alternative metrics of population size are available.

Table 10.11b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area of within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	68,400	n/a	n/a	n/a	n/a	n/a	n/a
Scotland	5,000	n/a	n/a	n/a	n/a	n/a	n/a
Wales	6,800	n/a	n/a	n/a	n/a	n/a	n/a
Britain	**80,100**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**	**n/a**

The Article 17 Report on Leisler's bat population size 2007-2012 is shown in Table 10.11c (Joint Nature Conservation Committee, 2013b).

Table 10.11c Article 17 Report on Leisler's bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	9,750	9,750
Scotland	250	250
Wales	Not estimated	Not estimated
Britain	**24,000**	**40,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

The geographical range estimate for the species, based on known records since 1995, is shown in Table 10.11d.

Table 10.11d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	68,400	n/a
Scotland	4,980	n/a
Wales	6,740	n/a
Britain	**80,100**	**128,000**

Critique

There is no basis for making a population estimate for this species.

Very few roosts are known and the species is highly likely to be under-recorded. A very small number of maternity roosts in buildings have been identified, and in Scotland, the species has been observed roosting in trees in Ayrshire and Dumfries and Galloway (Robert Raynor, *pers. comm)*. It is impossible to estimate the relative probability of finding a grounded bat of this species compared with species commonly found in dwelling houses, so the ratio of grounded Leisler's bats to other species cannot be used as the basis for making a population estimate. Roost density estimates were not available from the literature, from other data sources, or from expert opinion. The estimate of roost size was based on a very low sample size, and was almost double that derived from expert opinion (usual value given as 35 individuals, usual range 8-40).

While the species makes loud echolocation calls that are readily recorded on modern broadband bat detectors, there is considerable overlap in the call parameters of the other Nyctaloid bats (noctule and serotine bats). Many acoustic records, and all of those in Wales, are not supported by regional records of bats identified in the hand (or by molecular analysis of droppings); this raises doubts about their validity. Given that Leisler's bat appears to use a wide range of habitats, and exhibits flexibility in its primary prey items, habitat suitability modelling is likely to be extremely difficult.

Experts were unable to provide estimates of roost density, and only one could provide information on roost size. The median roost count of 64 individuals (95%CI = 14-114) derived from the available datasets is almost double the estimate derived from expert opinion based on experience at 3 roosts (35 individuals, usual range 8-40).

Three main sources of error are identified. Firstly, the density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. No expert was able to provide estimates, and no further information was available from the literature. There is currently no understanding of Leisler's bat roost (or colony) density. Given the generalist nature of the species, and the likelihood that very large numbers of roosts are unreported, models of roost distribution are likely to be highly speculative. From the data currently available, precise estimates of expected roost counts across Britain, or even regionally, are not possible. Finally, the range of the species is uncertain. Modern broadband bat detectors have increased the number of records based on acoustic data, but the scale of misidentification when Nyctaloid bats are classified to species is unclear. In Wales, all of the records for this species are based on acoustic data, and have not been verified by either the capture of animals or the genetic profiling of droppings.

Table 10.11e Reliability assessment for Leisler's bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 10,000 individuals was given in Harris et al. (1995) (England 9,750; Scotland 250; Wales 0), this estimate was graded as having very poor reliability. Given that there is no basis for deriving a current population estimate, comparison with Harris et al. (1995) was not attempted.

The distribution reported in the current review is larger than that given by Arnold (1993), who showed the species as being virtually absent from the south west of England, Wales and Scotland. It is unclear whether this represents true range expansion or a focused increase in observer effort, especially in relation to new wind farm developments in the borders and the south west of Scotland. The change from heterodyne to broadband acoustic detectors also increases the probability of recording Leisler's bat. However, it is also possible that some of the new acoustic records are owing to misidentification. The range is smaller than that given in the Article 17 Reports (Joint Nature Conservation Committee, 2013b); some of this difference may be caused by the methodological differences. The Southern Scotland Bat Survey has suggested a wider range in south west Scotland than presented in Figure 10.11a, so the range size may be underestimated. These findings need to be confirmed by genetic analysis of droppings or visual identification of bats in the hand, owing to the difficulty of conclusively identifying the species acoustically.

Other evidence of changes through time

No other data are available with which to assess trends over time.

Table 10.11f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of Change

Table 10.11g Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts and variability in occupancy across geographical or habitat gradients.	n/a	No data available: formal study is urgently required.
Size of roosts.	n/a	Very limited data available: formal study is urgently required. Alternatively, a widescale population genetics study is required to estimate the effective population size.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change on local population.	All	No data available.
Impacts of anthropogenically-induced mortality (wind turbines, vehicles, etc.) on populations.	All	No data available.
Impacts of changes in agricultural practice, particularly the use of anthelmintic agents and insecticides, on prey abundance and local population sizes.	Grazing land	No data available.
Impacts of changing woodland management, affecting total woodland area and the amount of standing deadwood, on roost availability.	Broadleaved woodland	No data available.

Future prospects

Table 10.11h An assessment of the future prospects for the Leisler's bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Stable

10.12 Noctule bat *Nyctalus noctula*

Habitat preferences

The noctule bat, *Nyctalus noctula*, is a fast-flying species capable of commuting long distances. It feeds on the wing, and takes a combination of large Coleoptera (beetles), Lepidoptera (moths) and small Diptera (flies) ((Jones, 1995; Mackenzie and Oxford, 1995; Vaughan, 1997). The species emerges early, particularly during lactation (Jones, 1995; Mackie and Racey, 2007), and is therefore sometimes thought to benefit from artificial night lighting. However, there is no evidence of higher noctule bat activity in areas that are lit compared with dark control sites (Mathews et al., 2015). There are relatively few studies on the foraging behaviour of the species, although it is thought that flights of 10km are easily within the species' range. In south west England, a preference for broadleaved woodland and pasture has been reported, with animals travelling an average maximum distance of 4.5km to foraging grounds (Mackie and Racey, 2007). However, the very rapid movement of the species, its high altitude flight in open space, and the relatively long distance over which its calls can be heard (≥30m) mean that it is often difficult to identify habitat preferences. Recent work using global positioning system (GPS) collars in Germany indicates that bats in an agricultural landscape, including a wind farm, showed a preference for wetlands and an avoidance of arable fields relative to their abundance (Roeleke et al., 2016).

Summer roosts are usually located in broadleaved trees or Scots pine — including solitary trees in parkland and suburban areas as well as woodlands. Rot holes, splits in trees, and woodpecker holes are all used, and the noctule bat will also roost in bat boxes mounted on trees (Mackie, 2002). Elsewhere in Europe it often roosts in buildings; this is less common but not unknown in Great Britain. Colonies often have alternative roost locations (with the potential for the colony to be split) across several sites. In some locations, it switches between roosts frequently, while remaining within the same general area (Mackie and Racey, 2007). The hibernation preferences of the species are not well known in the UK, but it is assumed it largely uses holes deep within trees. Elsewhere in Europe, it also uses large bat boxes designed for hibernation and cracks in rock faces (Jasja Dekker, *pers. comm.*). Noctule bats migrate long distances between hibernation and summer roosts in both eastern and western Europe (Sluiter and van Heerdt, 1966; Petit and Mayer, 2000; Lehnert et al., 2014). There is currently no evidence that British noctule bats migrate, but no detailed studies have been undertaken.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Unknown; Scotland: Unknown; Wales: Unknown).

Species' distribution

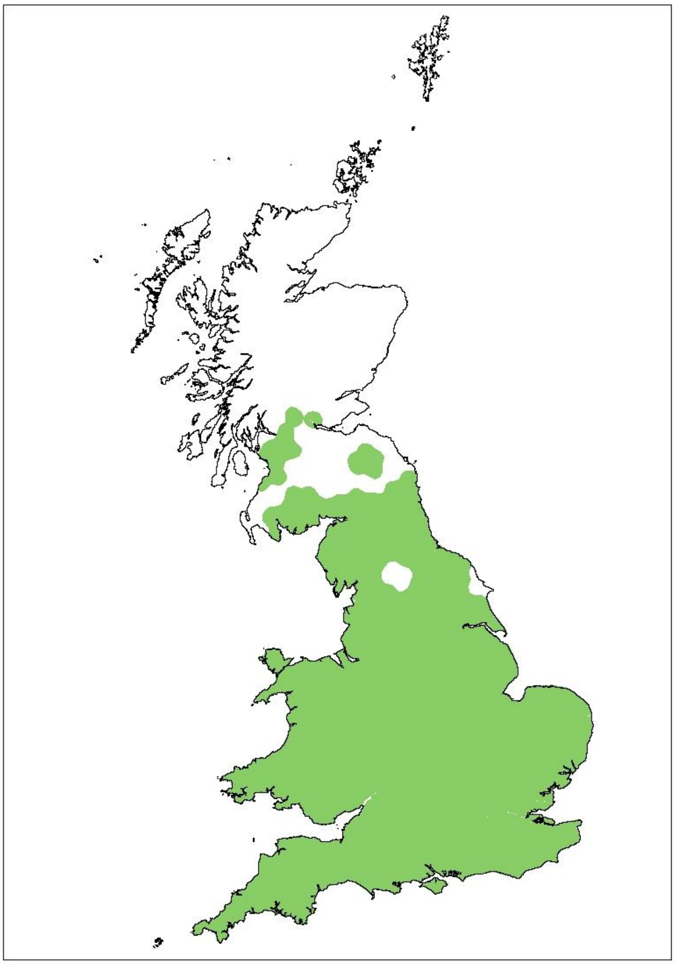


Figure 10.12a Current range of the noctule bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Information was available from 12 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses. The upper and lower plausible limits for the roost count were defined by the 95% confidence intervals for the median. Small roosts were not excluded because the minimum group size required for breeding is not clear for this species. The median pre-breeding roost size calculated from the available data was 40.5 individuals (95%CI = 16-59, range = 11-124, n=12 roosts).

Expert opinion was obtained from 7 individuals. No experts had information on the sex ratio of the population, and this information was not available from Harris et al. (1995). One publication suggested a female bias in juvenile and adult samples, but the degree was not stated, so this could not be used as a basis for any adjustments to the estimates (Harris and Yalden, 2008).

No expert was able to provide an estimate of the pre-breeding sex ratio in maternity colonies, and no data were available from the literature or from Harris et al. (1995). In a sample of 93 adults caught at roosts in Cambridgeshire, 18 were males (19.3%). However, some of these were caught after the young were born, so the composition of the roost pre-breeding is likely to be much less than 19% male (Tony Mitchell-Jones, *pers. comm.*). For the purposes of the current calculation, it was therefore assumed that pre-breeding roosts contain only female bats.

Only two experts provided estimates of roost densities (for two regions of England). These were 0.06 roosts/km^2 and 0.05 roosts/km^2) (reliability scores 7/10 and 4/10 respectively). In a study of a 2500km^2 area of North Yorkshire, a roost density of 0.055 roosts/km^2 (all in buildings) was reported (Jones et al., 1996). The estimates in Jones et al. (1996) were based on an assumption that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). Given that no data were available to verify this assumption, a second density estimate was derived for the purpose of the current calculations by using the entire 2,500km^2 study area (which gives a density estimate of 0.004 roosts/km^2). The highest and lowest values of the available estimates in the literature were used to define plausible roost densities in good and poor quality habitats. The median of the two expert opinion values (0.055 roosts/km^2) is identical to the density reported by Jones et al. (1996).

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

The population estimate was calculated on the basis of adult bat density and the geographical range. Adult bat densities (bats/km^2) were calculated as follows:

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*
*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*
*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is a typical maternity roost in the pre-parturition period. n. is the number of adults.
‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The population estimate was based on adult population density and the extent of occupancy across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density (bats/km^2) * total area within range (km^2)*
*Lower limit = Lower limit adult density (bats/km^2) * total area within range (km^2)*
*Upper limit = Upper limit adult density (bats/km^2) * total area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.12a.

Table 10.12a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	40.5 (16-59)
Sex ratio	1
Maternity roost density (roosts/km^2)	0.055 (0.004*-0.125**)

* Jones et al. (1996).
** Expert opinion.

Population estimation and range

Table 10.12b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km²)	Bat density (adults/km²)			Adult population size*		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	127,000	4.5	0.1	14.8	565,000	17,700	1,872,000
Scotland	9,500	Not assessed**					
Wales	20,600	4.5	0.1	14.8	91,900	2,880	304,000
Britain	**157,000**	**Not assessed****					

* Row and column totals may not sum because of rounding.
** In view of the uncertainty around the density estimates, the population size for this species is not shown.

The Article 17 Report on noctule bat population size 2007-2012 (Joint Nature Conservation Committee, 2013b), shown in Table 10.12c, is below the plausible range estimated in the current review for each country and for Great Britain.

Table 10.12c Article 17 Report on noctule bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	45,000	45,000
Scotland	250	250
Wales	4,750	4,750
Britain	**50,000**	**50,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

The current distribution of the species, based on known records since 1995, is shown in Table 10.12d.

Table 10.12d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	127,000	n/a
Scotland	9,490	n/a
Wales	20,600	n/a
Britain	**157,000**	**171,600**

Critique

There is considerable uncertainty surrounding the population estimates for this species. Relatively few roosts are known because they are primarily in woodland; and comparisons of ratios of grounded noctule bats to other species will be equally unreliable as encounter rates are likely to be very low for noctule bats.

There is an extreme lack of data on the density of roosts. Although the estimate provided by Jones et al. (1996) of 0.055 roosts/km^2 exactly corresponds with expert opinion on typical roost density, it cannot be viewed as a gold standard comparison; it was not based on systematic survey effort, and tree roosts would have been under-reported. The roost density calculated here from Jones (based on the entire study area) provides a very low plausible limit for bat density estimates. The use of expert opinion alone would increase the estimate from 0.1 to 0.6 bats/km^2, and produce corresponding increases in the lower limits of the population estimates, to give the following values: England 81,200; Scotland 6,100; Wales 13,200; and Great Britain 100,500.

The estimate of roost sizes is based on a low sample size. In addition, a colony may make use of multiple roosts and switch between them, meaning that there is likely to be high variability in counts at individual sites. The confidence intervals around the median of 40.5 bats are therefore quite wide, ranging from 16 to 59. As a result, the overall estimate may have under- or overestimated the population by about a half.

Habitat suitability modelling is unlikely to yield major insights for this species because of their wide-ranging flight and use of a variety of roosting locations. Noctule bats can be found in a trees ranging from young field maple to ancient oaks (Fiona Mathews *pers. obs*.); and they inhabit trees in parks and hedgerows, as well as those in woodland. Buildings are also sometimes used as roosting sites.

The lack of data on the pre-breeding sex ratio in maternity sites introduces an additional source of error. The calculations presented here are based on an assumption that all individuals in these sites are female. If half of the individuals are male, this would halve the estimates presented here.

Hibernation data could not be incorporated into this report owing to a lack of information. The median roost count of 40.5 individuals (95%CI = 16-59) is comparable with the estimates provided by experts (36.5 individuals; typical range = 9.5-74). It is also compatible with the only published value available in the recent literature (mean 26.1 individuals based on 8 sites in Yorkshire; Jones et al., 1996). Only 2 experts provided estimates of roost density (confidence scores 7/10 and 4/10), and therefore confidence in the values used for this parameter is low.

Several sources of error are identified. Firstly, the density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. Only two experts provided opinions, and no further information was available from the literature. Estimates were also only available for England. Given the generalist nature of the species, and the likelihood that very large numbers of roosts are unreported, models of roost distribution are likely to be highly speculative. Secondly, no information is available on the sex ratio within maternity colonies pre-breeding. Given the large effect on the total population size, further research is therefore urgently required. Roost count data were derived from a relatively small sample size. Whilst these are comparable with the published literature, it is unclear whether or how colony size varies across Great Britain. Finally, no occupancy data or information on trends in density across geographical gradients is available. It has therefore been assumed that the overall roost density estimate applies throughout the entire range.

Table 10.12e Reliability assessment for noctule bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The main population size estimates provided here are of an order of magnitude greater than those in Harris et al. (1995) and the Article 17 Reports (Joint Nature Conservation Committee, 2013b). Nevertheless, the values previously estimated do fall within the plausible limits, with the exception of Scotland. The previous estimates were given a moderate reliability score.

The distribution is larger than reported by Arnold (1993), which showed the species as being virtually absent from Scotland. It is unclear whether this represents true range expansion or focused increase in observer effort, especially in relation to new wind farm developments in the borders and south west of Scotland. The range is comparable with that given in the Article 17 Reports (Joint Nature Conservation Committee, 2013b).

Other evidence of changes through time

The National Bat Monitoring Programme (NBMP) includes 600 sites in its field survey for noctule bats. These have been monitored since 1998. The field survey suggests that there has been no change in the activity index during the survey period (see Table 10.12f).

Table 10.12f Trends in noctule bat activity from baseline to 2015 as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available for Wales or Scotland to estimate trends.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	Field	491	1998	9.2	0.5
Scotland	n/a	n/a	n/a	n/a	n/a
Wales	n/a	n/a	n/a	n/a	n/a
Britain	Field	600	1998	16.3	1.0

* Indicates that the trend is significant ($p<0.05$).

† Percentage change since the 1999 baseline.

Table 10.12g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.12h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Collisions with wind turbines.	One of the primary species killed at wind turbines. It is unclear whether the scale of the casualties is sufficient to impact on local populations.	Mathews et al., 2016	Negative

Data deficiencies

Table 10.12i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts, and variability in occupancy across geographical or habitat gradients.	n/a	No data available: formal study is urgently required.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change on the local population.	All	No data available.
Impacts of anthropogenically-induced mortality (wind turbines, vehicles, etc.) on populations.	All	No data available.
Impacts of change in agricultural practice, particularly the use of anthelmintic agents, on prey abundance and local population sizes.	Grazing land	No data available.
Impacts of changing woodland management, affecting the total woodland area and the amount of standing deadwood, on roost availability.	Broadleaved woodland	No data available.

Table 10.12j An assessment of the future prospects for the noctule bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Unknown

10.13 Common pipistrelle bat *Pipistrellus pipistrellus*

Introductory note

The common pipistrelle (*Pipistrellus pipistrellus*) and the soprano pipistrelle (*P. pygmaeus*) are the most abundant and widespread bats in Great Britain. The separation of these cryptic species, which typically differ in their phonic patterns, occurred relatively recently (Barratt et al., 1997). The last population (Harris et al., 1995) and distribution (Arnold, 1993) reviews, as well as many scientific papers, do not distinguish *P. pygmaeus* from *P. pipistrellus.* Direct comparison with these previous population reviews is therefore difficult. Where data are known to relate exclusively to one of the species, the term *sensu stricto* (*s.s.*) is used after the species' name. Where data may relate to a combination of common and soprano pipistrelle bats, usually because they were recorded prior to the separation of the two species, the suffix *sensu lato* (*s.l.*) is applied.

Habitat preferences of common and soprano pipistrelle bats

Both the common and the soprano pipistrelle bat are extremely widespread species, and are found in almost any habitat type, ranging from grasslands to urban and surburban environments. Although both species, and notably the common pipistrelle bat, are considered to be well adapted to built environments, recent evidence shows that there is a strong negative response of common pipistrelle bats to the degree of urbanisation at a relatively local scale (1 km; Lintott et al., 2016). The soprano pipistrelle bat is frequently reported to make particular use of riparian habitat (Davidson-Watts and Jones, 2006; Nicholls and A. Racey, 2006; Lintott et al., 2016). However, the reverse association has also been reported (Warren et al., 2000; Glendell and Vaughan, 2002; Lintott et al., 2015). Whilst both species feed predominantly on Diptera (suborder Nematocera), there is some dietary differentiation, with soprano pipistrelle bats making greater use of the families Chironomidae and Ceratopogonidae (Barlow, 1997), as might be expected if there is greater use of riparian habitats. The common pipistrelle bat frequently forages over pasture, and there is concern that activity is lower where cattle have been treated with anthelmintic drugs (avermectins; Downs and Sanderson, 2010). In woodlands, the activity of the soprano pipistrelle bat is positively linked with the amount of habitat fragmentation, possibly because it makes use of edge environments; whereas the activity of the common pipistrelle bat is higher at sites with grazing livestock (Fuentes-Montemayor et al., 2013).

There is some evidence that the foraging behaviour of the two species differs, with the common pipistrelle bat making more foraging flights of shorter duration; the soprano pipistrelle bat spends less time foraging, makes fewer sorties, but flies further (Davidson-Watts and Jones, 2006). Limited data are available on foraging ranges, but most activity appears to occur within 2.5km of summer roosts (Davidson-Watts and Jones, 2006; Stone et al., 2015). However, much larger home ranges are reported for the soprano pipistrelle bat when it uses conifer plantations as its primary habitat — here, some lactating individuals regularly make nightly flights of >10km (Kirkpatrick, 2017). There is also evidence that, at least in soprano pipistrelle bats, females require higher quality habitats than males (Lintott et al., 2014).

Both species usually roost in buildings. They are the species most regularly reported in houses and churches (European Protected Species Licence data), but they can use a wide variety of constructions, including barns, warehouses and amenity buildings. Roosts of the soprano pipistrelle bat are differentially located in areas close to waterways (Jenkins et al., 1998; Oakley and Jones, 1998), particularly in the case of large roosts (Fiona Mathews, *pers. obs.*). Roost habitat selection has not been assessed for the common pipistrelle bat. Both species are also known to use bat boxes (although these are usually non-breeding individuals) and are only rarely found roosting in trees. Pipistrelle bats are rarely visible within buildings, as they are concealed in crevices, soffit boxes, beneath tiles and under woodwork.

Colonies of common and soprano pipistrelle bats will use several alternative roosts within a given area. Not only will individuals switch between them, but different roosting locations will be favoured at different times. One study has investigated the impact of exclusion of soprano pipistrelle bats from dwelling houses under licence (Stone et al., 2015). This confirmed that the species frequently switches roosts and, when excluded, the bats continued to make use of alternative roosts without any apparent impacts on home range, foraging behaviour, or the frequency of roost switching. This roost switching behaviour makes deriving an overall estimate of abundance particularly challenging. The National Bat Monitoring Programme cautions that, because of roost switching, long-term trends inferred from roost counts may be unreliable (Bat Conservation Trust, 2016).

The soprano and the common pipistrelle bat are generally considered to be sedentary across Europe, although there are recent suggestions of long-distance movements for the soprano pipistrelle. In winter, pipistrelle bats are occasionally found during building renovation works (e.g., under tiles or in cavity walls), but it is unclear where most of the British population hibernates: individuals are found only very rarely in underground sites.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A species' distribution map is presented in Figure 10.13a. Gaps in the species' distribution in Scotland are likely to reflect areas with low survey effort, rather than true gaps in the species' range. Although all records used in creating the map are for *P. pipistrellus s.s.*, and do not include records submitted using a generic term (e.g. pipistrelle bat), in the earlier part of the date range, some records are likely to be for *P. pygmaeus*, as the two species were not distinguished until 1997.

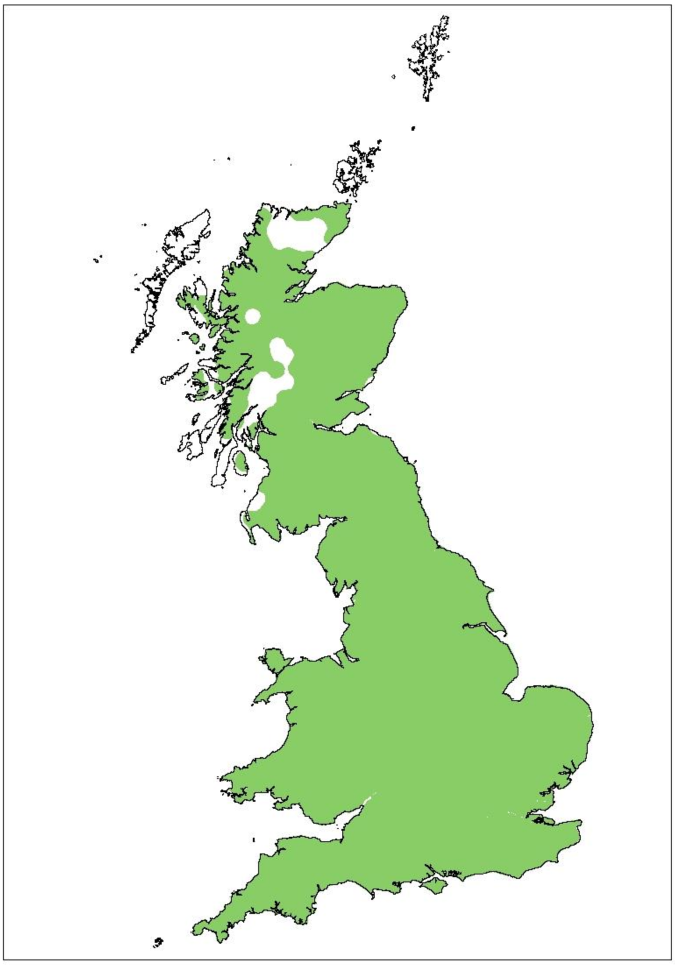


Figure 10.13a Current range of the common pipistrelle bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Estimating the population size for common pipistrelle bats is challenging, despite their being one of Britain's most commonly recorded bats. Recording of roosts is insufficiently comprehensive to permit a direct estimate of total roost numbers. It is also not possible to estimate the density of bats based on acoustic surveys (since bat numbers cannot be inferred from the number of calls recorded) or from capture records (since capture success is not proportional to abundance in the environment; and efforts to trap bats tend to be focused on particular sites with a high probability of capture success, such as swarming sites for *Myotis spp.*). No comprehensive population genetics surveys have been conducted.

Information was available from 465 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses (unless the count had fewer than 30 individuals, in which case the next available year where the count was ≥30 was used). If no counts with ≥30 bats were available, then the site was excluded on the grounds that it was unlikely to be used as a maternity roost for this species, and including it would risk double counting the same individuals when they moved on to maternity roosts. The upper and lower plausible limits for the roost count were defined by the 95% confidence intervals for the median. The median pre-breeding roost size for the common pipistrelle bat was 72 individuals (95%CI = 67-79, range 30-512, n=465 roosts).

No data were available from the literature on the sex ratio of adult common pipistrelle bats in maternity roosts pre-parturition. Studies conducted before the two phonic species were separated suggest that there are few, if any, adult males in maternity colonies (Racey, 1969). Expert opinion on the species was obtained from 7 individuals. A further 2 experts responded to requests for input on this species, but were unable to provide the information necessary for the calculation of population size. One expert reported that roosts were 100% female, whilst the others were unsure. For the purposes of these calculations, it was assumed that the proportion of females in pre-breeding maternity colonies is 1.0. No data were available from the literature, or elsewhere, on the sex ratio of the whole population.

Two roost density estimates were available from experts, and these were judged reasonably reliable (scores 6/10 and 8/10). The median of the values they gave for typical habitat (0.105 roosts/km^2) was used as the central estimate, and the median of their values for poor and good quality habitat were used as the lower and upper plausible limits (0.035 roosts/km^2 and 0.225 roosts/km^2 respectively). There was also one estimate from Harris (2014) of common pipistrelle maternity roost densities in a 100km^2 survey area, derived from a local 10-year survey initiative (0.07 roosts/km^2). This estimate was within the plausible ranges given above, and its inclusion or exclusion made no material difference to the median value. Earlier reports of common pipistrelle bat density (e.g., in Jones et al., 1996) were not used because they were conducted before distinguishing between common and soprano bats was routine practice.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

Adult bat densities (bats/km^2) were calculated as follows:

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*
*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*
*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is a typical maternity roost in the pre-parturition period. n. is the number of adults.
‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The population estimate was based on adult population density and extent of occupancy across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density (bats/km^2) * total area within range (km^2)*
*Lower limit = Lower limit adult density (bats/km^2) * total area within range (km^2)*
*Upper limit = Upper limit adult density (bats/km^2) * total area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.13a.

Table 10.13a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	72 (67-79)
Sex ratio	1
Maternity roost density (roosts/km^2)	0.105 (0.035-0.225)*

* Expert opinion.

Population estimation and range

Table 10.13b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	130,000	14.4	4.7	35.6	1,870,000	609,000	4,620,000
Scotland	60,800	14.4	4.7	35.6	875,000	285,000	2,160,000
Wales	20,600	14.4	4.7	35.6	297,000	96,600	732,000
Britain	**211,000**	**14.4**	**4.7**	**35.6**	**3,040,000**	**991,000**	**7,510,000**

The estimates were sensitive to the exclusion of roosts with <30 bats. The use of the latest available peak count obtained prior to July, regardless of size, increased the number of sites to 554 and changed the density estimate to 10.2 bats/km^2 (PIs = 3.0-25.4). Repeating the calculations using these data reduced the population estimate by approximately a third. The values are as follows: England 1,325,126 (PIs = 386,495-3,303,071); Scotland 620,078 (PIs = 180,856-1,545,635); Wales 210,132 (PIs = 61,289-523,785); Britain 2,155,336 (PIs = 628,640-5,372,491).

The estimates in the Article 17 Report on common pipistrelle bat status 2007-2012 shown in Table 10.13c (Joint Nature Conservation Committee (2013b) come to less than half of the estimates given by the current review, but the plausible ranges include the values given in those reports for each country and for Great Britain.

Table 10.13c Article 17 Report on common pipistrelle bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	800,000	800,000
Scotland	352,000	352,000
Wales	128,000	128,000
Britain	**1,390,000**	**1,390,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

The geographical range of the species, based on known records since 1995, is shown in Table 10.13d.

Table 10.13d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	130,000	n/a
Scotland	60,800	n/a
Wales	20,600	n/a
Britain	**211,000**	**226,400**

Critique

There is considerable evidence from bat detector records, roost records and recoveries of grounded bats that common and soprano pipistrelle bats are the most abundant bat species in the UK. Owing to their strong association with buildings (and therefore humans), their geographical ranges are confidently defined. However, there is an extreme lack of data on the density of roosts. Given that common pipistrelle bats can be found virtually anywhere, habitat suitability modelling is unlikely to provide useful insights. A further important source of error is the very limited data on the sex ratio pre-breeding in maternity sites. If half of the individuals are male, this would substantially reduce the overall estimate. Because of the lack of information on the location of hibernacula, it has not been possible to use hibernation data in this report.

There are two published reports in the literature which attempt to estimate pipistrelle bat density, but neither distinguishes between the two phonic types. The values given were approximately 36 bats/km^2 for a 3,200km^2 area in northern Scotland (Speakman et al., 1991), and 25.2 bats/km^2 for a 775km^2 area in the Vale of York (Jones et al., 1996), assuming in each case that roosts were almost entirely comprised of adult females. By comparison, in the current review, the density is estimated at approximately 14 bats/km^2. If approximately half of the colonies studied by Speakman et al. (1991) and Jones et al. (1996) comprise soprano pipistrelle bats, then the current estimate of common pipistrelle bat density is similar to these earlier reports. Both of the published papers used rigorous methods to achieve density estimates, performing most roost counts pre-breeding. However, they did include an important assumption about area occupied by the roost. It was assumed that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). No data were available to verify this assumption. These earlier studies also noted that the number of identified roosts in the study areas did not reach an asymptote, suggesting that the overall densities were underestimates. In addition, the studies were conducted in northern England and northern Scotland where densities are likely to be lower than in warmer regions of Britain.

The median roost count value used in this report (72 individuals) is comparable with the value of 90 (typical range 30-197) given by experts based on their experience at more than 200 sites. It is also similar to the only published value available in the recent literature (median 76 individuals (range 20-223) based on 33 roosts studied by Barlow and Jones (1999)).

Our estimates excluded colonies surveyed as part of European Protected Species Applications that contained fewer than 30 bats. This ensured that counts did not include individuals in formation roosts that were then re-counted at maternity sites. As a consequence there may have been some overestimation of population size: when all roosts were included, the bat population density estimate fell by approximately a third. However, most data were derived from the National Bat Monitoring Programme. The objective of that project is longitudinal monitoring, so it is likely that non-breeding roosts were included. Given that the estimated roost size is close to expert opinion and published data, it is likely to be a reasonable basis for the calculations performed in this review. Only three estimates of roost density were available, so there is some uncertainty about whether they are nationally representative.

Three main sources of error are identified. Firstly, the density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. Only two experts provided opinions, and no further information was available from the literature, indicating that there is little or no understanding of common pipistrelle roost density. Given the generalist nature of the species, and the likelihood that very large numbers of roosts are unreported, models of roost distribution are likely to be highly speculative. Secondly, few data are available on the sex ratio within maternity colonies pre-breeding. The large potential effect on the total population size means that further research on sex ratios is urgently required. Finally, no occupancy data or information on trends in density across geographical gradients is available. It has therefore been assumed that the overall roost density estimate applies throughout the range.

Table 10.13e Reliability assessment for common pipistrelle bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates†	0	<100 roosts	
	1	<150 roosts	1
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**2**

* Either from the literature or from expert opinion with high reliability scores.
† No evidence on roost size is available for tree roosts, so this is scored as 1.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

It is difficult to make a direct comparison with Harris et al. (1995) because the two phonic types were not separated in that report. Harris drew largely on densities estimated in northern Scotland (Speakman et al., 1991), which is towards the edge of the range for *P. pipistrellus s.l.*, and ranked the overall reliability of the population assessment as moderate.

The estimated density of bats in the current review is higher than that assumed by Harris (14 bats/km^2 compared with 10/km^2 for *P. pipistrellus s.l*, and therefore presumably approximately 5/km^2 for *P. pipistrellus s.s.*).

The distribution of *P. pipistrellus s.s.* is similar to that given for *P. pipistrellus s.l* in Arnold (1993) and Harris et al. (1995). It is also similar to that given in the Article 17 Reports (Joint Nature Conservation Committee, 2013b). It is therefore concluded that there has been no change in range.

Other evidence of changes through time

The National Bat Monitoring Programme (NBMP) includes 599 sites with common pipistrelle bats in its field survey and 488 roosts. These have been monitored since 1998 and 1997 respectively. The field survey has recorded a consistent and significant increase in acoustic records of common pipistrelle bats, whereas the roost counts have shown a consistent and significant decrease across the survey period (see Table 10.13f). The Bat Conservation Trust notes that roost counts may be unreliable for trend analysis owing to the propensity of the species to switch roosts. The acoustic detectors used to record bat activity in the field have also changed considerably over the recording period, becoming much more sensitive. In addition, volunteer observers find it difficult to distinguish between common and soprano pipistrelle bats using heterodyne acoustic detectors: there is considerable misidentification of the two phonic types, and also confusion with *Myotis spp.* (Kate Barlow, *pers. comm.*). The true trend may be intermediate between the two trends reported for common and soprano pipistrelle bats.

Table 10.13f Population trends in common pipistrelle bats from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Field**	**490**	**1998**	**89.4***	**4.0**
	Roost	389	1990	-50.0*	-4.2
Scotland	**Field**	**75**	**1998**	**46.8**	**2.4**
	Roost	62	1997	-51.9*	-4.5
Wales	n/a	n/a	n/a	n/a	n/a
Britain	**Field**	**599**	**1998**	**81.1***	**3.8**
	Roost	488	1990	-51.6*	-4.4

* Indicates that the trend is significant ($p<0.05$).
† Percentage trend since the 1999 baseline.

Table 10.13g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated. The previous publications did not distinguish P*. pipistrellus* (*sensu stricto)* and *P. pygmaeus;* comparisons are therefore made with the data presented for *P. pipistrellus (sensu lato).*

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.13h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Collisions with wind turbines.	One of the primary species killed at wind turbines. It is unclear whether the scale of casualties is sufficient to affect local populations.	Mathews et al. (2016)	Negative
Vehicle collisions.	One of the primary species recorded in vehicle collisions. It is unclear whether the scale of casualties is sufficient to affect local populations.	Fensome and Mathews (2016)	Negative
Protection of roosts.	Legislative protection of maternity roosts, in particular, has been introduced to prevent destruction and disturbance.	n/a	Positive
Predation by cats.	One of the species most frequently injured and killed by cats. Where cats are able to access roost entrance, there can be significant effects on individual colonies.	Andrew Kelly, RSPCA (*pers. comm.*).	Negative
Changes to the structure of buildings and insulation methods.	Changes to building regulations, and efforts to make buildings more energy-efficient, have tended to reduce their accessibility and thermal suitability for bats. Breathable roofing membranes also pose a threat of entanglement.	Waring et al. (2013)	Negative

Data deficiencies

Table 10.13i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts, and occupancy in different habitat types/geographical areas.	n/a	No data available: formal study is urgently required.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change, lighting, etc., on local population.	All	No data available.
Impacts of anthropogenically-induced mortality (wind turbines, vehicles, cats, entanglement in breathable roofing membranes, etc.) on populations.	All	Very limited data available.
Effectiveness of current planning and licensing systems protecting roosts.		No data available.

Future prospects

Table 10.13j An assessment of the future prospects for the common pipistrelle bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Stable

10.14 Soprano pipistrelle bat *Pipistrellus pygmaeus*

Introductory note

Common pipistrelle (*Pipistrellus pipistrellus*) and soprano pipistrelle bats (*P. pygmaeus*) are the most abundant and widespread bats in Great Britain. The separation of these cryptic species, which typically differ in their phonic patterns, occurred relatively recently (Barratt et al., 1997). The last population (Harris et al., 1995) and distribution (Arnold, 1993) reviews, as well as many scientific papers, do not distinguish *P. pygmaeus* from *P. pipistrellus.* Direct comparison with previous population reviews is therefore difficult. Where data are known to relate exclusively to one of the species, the term *sensu stricto* (*s.s.*) is used after the species' name. Where data may relate to a combination of common and soprano pipistrelle bats, usually because they were recorded prior to the separation of the two species, the suffix *sensu lato* (*s.l.*) is applied.

Habitat preferences

Both the common and the soprano pipistrelle bat are extremely widespread species, and are found in almost any habitat type, ranging from grasslands to urban and surburban environments. Although both species, and notably the common pipistrelle bat, are considered to be well adapted to built environments, recent evidence shows that there is a strong negative response of common pipistrelle bats to the degree of urbanisation at a relatively local scale (1 km; Lintott et al., 2016). The soprano pipistrelle bat is frequently reported to make particular use of riparian habitat (Davidson-Watts and Jones, 2006; Nicholls and A. Racey, 2006; Lintott et al., 2016). However, the reverse association has also been reported (Warren et al., 2000; Glendell and Vaughan, 2002; Lintott et al., 2015). Whilst both species feed predominantly on Diptera (suborder Nematocera), there is some dietary differentiation, with soprano pipistrelle bats making greater use of the families Chironomidae and Ceratopogonidae (Barlow, 1997), as might be expected if there is greater use of riparian habitats. The common pipistrelle bat frequently forages over pasture, and there is concern that activity is lower where cattle have been treated with anthelmintic drugs (avermectins; Downs and Sanderson, 2010). In woodlands, the activity of the soprano pipistrelle bat is positively linked with the amount of habitat fragmentation, possibly because it makes use of edge environments; whereas the activity of the common pipistrelle bat is higher at sites with grazing livestock (Fuentes-Montemayor et al., 2013).

There is some evidence that the foraging behaviour of the two species differs, with the common pipistrelle bat making more foraging flights of shorter duration; the soprano pipistrelle bat spends less time foraging, making fewer sorties but flying further (Davidson-Watts and Jones, 2006). Limited data are available on foraging ranges, but most activity appears to occur within 2.5km of summer roosts (Davidson-Watts and Jones, 2006; Stone et al., 2015). However, much larger home ranges are reported for the soprano pipistrelle bat when it uses conifer plantations as its primary habitat — here, some lactating individuals regularly make nightly flights of >10km (Kirkpatrick, 2017). There is also evidence that, at least in soprano pipistrelle bats, females require higher quality habitats than males (Lintott et al., 2014).

Both species usually roost in buildings. They are the species most regularly reported in houses and churches (European Protected Species Licence data), but they can use a wide variety of constructions, including barns, warehouses and amenity buildings. Roosts of the soprano pipistrelle bat are differentially located in areas close to waterways (Jenkins et al., 1998; Oakley and Jones, 1998), particularly in the case of large roosts (Fiona Mathews, *pers. obs.*). Roost habitat selection has not been assessed for the common pipistrelle bat. Both species are also known to use bat boxes (although these are usually non-breeding individuals), and are hardly ever found roosting in trees. Pipistrelle bats are rarely visible within buildings, as they are concealed in crevices, soffit boxes, beneath tiles and under woodwork.

Colonies of common and soprano pipistrelle bats will use several alternative roosts within a given area. Not only will individuals switch between them, but different roosting locations will be favoured at different times. One study has investigated the impact of exclusion of soprano pipistrelle bats from dwelling houses under licence (Stone et al., 2015). This confirmed that the species frequently switches roosts and, when excluded, the bats continued to make use of alternative roosts without any apparent impacts on home range, foraging behaviour, or the frequency of roost switching. This roost switching behaviour makes deriving an overall estimate of abundance particularly challenging. The National Bat Monitoring Programme cautions that because of roost switching, long-term trends inferred from roost counts may be unreliable (Bat Conservation Trust, 2016).

The soprano and the common pipistrelle bat are generally considered to be sedentary across Europe, although there are recent suggestions of long-distance movements for the soprano pipistrelle. In winter, pipistrelle bats are occasionally found during building renovation works (e.g., under tiles or in cavity walls), but it is unclear where most of the British population hibernates: individuals are found only very rarely in underground sites.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 Overall Assessment 2013. Annex IV; UK: Favourable; England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A species' distribution map is presented in Figure 10.14a. Gaps in the species' distribution in Scotland are likely to reflect areas with low survey effort, rather than true gaps in the species' range. The species was only distinguished from *P. pipistrellus* in 1997, and although some records have been retrospectively amended (e.g., where colonies are known to have used the same roost), most data derive from 2000 onwards.

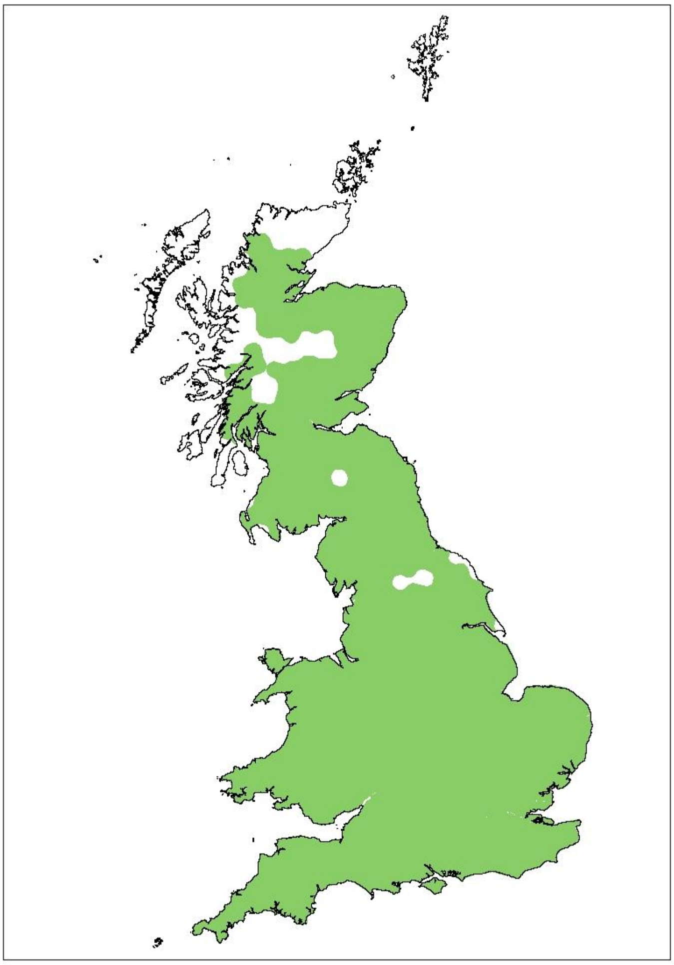


Figure 10.14a Current range of the soprano pipistrelle bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Estimating the population size for soprano pipistrelle bats is challenging, despite their being one of Britain's most commonly recorded bats. Recording of roosts is insufficiently comprehensive to permit a direct estimate of total roost numbers. It is also not possible to estimate the density of bats based on acoustic surveys (since bat numbers cannot be inferred from the number of calls recorded) or from capture records (since capture success is not proportional to abundance in the environment; and efforts to trap bats tend to be focused on particular sites with a high probability of capture success, such as swarming sites for *Myotis spp.*). No comprehensive population genetics surveys have been conducted. Information was available from 389 maternity roosts (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median of the most recently available peak counts before July was used for the analyses (unless the count had fewer than 30 individuals, in which case the next available year where the count was ≥30 was used). If no counts with ≥30 bats were available, then the site was excluded on the grounds that it was unlikely to be used as a maternity roost for this species, and including it would risk double counting the same individuals when they moved on to maternity roosts. The upper and lower plausible limits for the roost count were defined by the 95% confidence intervals for the median. The median pre-breeding roost size for the soprano pipistrelle bat was 198 individuals (95%CI = 175-213, range 30-1,429, n=389 roosts).

No data were available from the literature on the sex ratio of adult soprano pipistrelle bats in maternity roosts pre-parturition. Studies conducted before the two phonic species were separated suggest that there are few, if any, adult males in maternity colonies (Racey, 1969). Expert opinion on the species was obtained from 7 individuals. A further 3 experts responded to requests for input on this species but were unable to provide the information necessary for the calculation of population size. Three experts offered values for the proportion of female bats in maternity colonies pre-parturition. These values were 0.8, 0.90, and 0.98, and were used as the lower plausible value, typical value, and upper plausible value respectively. No data were available from the literature, or elsewhere, on the sex ratio of the whole population.

Two roost density estimates were available from experts, and these were judged reasonably reliable (scores 6/10 and 8/10). The median of the values they gave for typical habitat (0.065 roosts/km^2) was used as the central estimate, and the median of their values for poor and good quality habitat were used as the lower and upper plausible limits (0.035 roosts/km^2 and

0.1 roosts/km^2 respectively). There was also one estimate from Harris (2014) of common pipistrelle maternity roost densities in a 100km^2 survey area derived from a local 10-year survey initiative (0.56 roosts/km^2), and separate consideration was given to this estimate because of the anticipated high abundance of soprano pipistrelle bats in the geographical location of the study (see Results). Earlier reports of common pipistrelle bat density (e.g. in Jones et al., 1996) were not used because they were conducted before distinguishing between common and soprano bats became routine practice.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat.

Adult bat densities (bats/km^2) were calculated as follows:

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*
*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*
*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† 'Roost' is a typical maternity roost in the pre-parturition period. n. is the number of adults.
‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The population estimate was based on adult population density and extent of occupancy across mixed habitat types. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

*Total Adult Population = Median adult density (bats/km^2) * total area within range (km^2)*
*Lower limit = Lower limit adult density (bats/km^2) * total area within range (km^2)*
*Upper limit = Upper limit adult density (bats/km^2) * total area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.14a.

Table 10.14a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	198 (175-213)
Sex ratio	0.9 (0.8-0.98)*
Maternity roost density (roosts/km^2)	0.065 (0.035 -0.1)*

* Expert opinion.

Population estimation and range

Table 10.14b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower limits for England, Scotland, Wales, and the whole of Britain.

Country	**Area within range (km^2)**	**Bat density (adults/km^2)**			**Adult population size**		
		Estimate	**Plausible intervals**		**Estimate***	**Plausible intervals**	
			Lower	**Upper**		**Lower**	**Upper**
England	128,000	23.2	9.8	41.7	2,980,000	1,260,000	5,360,000
Scotland	52,200	23.2	9.8	41.7	1,210,000	512,000	2,180,000
Wales	20,600	23.2	9.8	41.7	478,000	202,000	862,000
Britain	**201,000**	**23.2**	**9.8**	**41.7**	**4,670,000**	**1,970,000**	**8,400,000**

The estimates were sensitive to the exclusion of roosts with <30 bats. The use of the latest available peak count obtained prior to July, regardless of size, increased the number of sites to 441 and changed the mid-density estimate to 16.1 bats/km^2 (PIs = 6.9-32.7). Repeating the calculations using these data reduced the population estimates approximately a third. The values are as follows: England 2,074,084 (PIs = 884,819-4,204,689); Scotland 843,201 (PIs = 359,715-1,709,380); Wales 333,298 (PIs = 142,187-675,679); Britain 3,250,583 (PIs =1,386,722-6,589,748).

The upper plausible population limit was also re-calculated using the density of maternity roosts reported by Harris (2014) for the Cotswold Water Park (0.56/km^2). However, this gave a value of 47,067,375, which was considered implausible, even as an upper limit, for the national population. The habitat in this particular geographical region may be particularly favourable for the species.

The Article 17 Report estimates of soprano pipistrelle bat population sizes 2007-2012 , shown in Table 10.14c (Joint Nature Conservation Committee, 2013b), are less than half of the lower plausible limit estimated in the current review. Even if there were only one roost per 100km^2, our calculations would still give an estimate of 1,395,178, which is much higher than the maximum population size in the Article 17 Report (Joint Nature Conservation Committee, 2013b). Similarly, even were the estimates based on all roosts, not just those with >30 bats (see above), the lower plausible estimate is still double that previously reported in the Article 17 Report.

Table 10.14c Article 17 Report on soprano pipistrelle bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	450,000	450,000
Scotland	198,000	198,000
Wales	72,000	72,000
Britain	**720,000**	**720,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

The geographical range of the species, based on known records since 1995, is shown in Table 10.14d (most data are derived from 1997 onwards; see Species-specific Methods).

Table 10.14d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	128,000	n/a
Scotland	52,200	n/a
Wales	20,600	n/a
Britain	**201,000**	**219,500**

Critique

There is considerable evidence from bat detector records, roost records and recoveries of grounded bats that common and soprano pipistrelle bats are the most abundant bat species found in the UK. Owing to their strong association with buildings (and therefore humans), their geographical ranges are confidently defined. While roost sizes vary considerably, the availability of data from a large sample of roosts means that the median can be estimated with reasonable precision, so is likely to be an adequate basis for the subsequent calculations. The exclusion of sites with peak counts of <30 bats did mean that the estimates are higher than they would have been had all sites been included. However, except in years with very unfavourable weather, it would be expected for this species that most females would join maternity colonies by the end of June.

There is an extreme lack of data on the density of roosts. Given that soprano pipistrelle bats can be found virtually anywhere, habitat suitability modelling is unlikely to provide useful insights. Some insight into the plausibility of the values used can be obtained from assessing the ratio of common to soprano pipistrelle bat roosts. Examination of all the data available to this project, via Local Records Centres and other sources, identified 222 soprano pipistrelle and 337 common pipistrelle roosts specifically flagged as maternity sites, a ratio of approximately 2:3. The expert opinions for the two species were 6.5 soprano pipistrelle bat roosts vs. 10.5 common pipistrelle bat roosts per 10km^2, which is also gives a 2:3 ratio. Hence, if roost densities are correct for common pipistrelle bats, then the estimates for soprano pipistrelle bats also appear reasonable.

The limited evidence on the sex ratio pre-breeding in maternity sites introduces an additional source of error: if half of the individuals are male, this would mean that the estimates presented here would be substantially reduced. The lack of information on the location of hibernacula means that it has not been possible to use hibernation data in this report. There are two published reports in the literature which attempt to estimate pipistrelle bat density, but neither distinguishes between the two phonic types. The values given were approximately 36 bats/km^2 for a 3,200km^2 area in northern Scotland (Speakman et al., 1991), and 25.2 bats/km^2 for a 775km^2 area in the Vale of York (Jones et al., 1996), assuming in each case that roosts were almost entirely comprised of adult females. By comparison, in the current review, the density is estimated at approximately 16 soprano pipistrelle bats/km^2. If approximately half of the colonies studied by Speakman et al. (1991) and Jones et al. (1996) are soprano pipistrelle bats, then the current density estimate is similar to these earlier reports. Both of the published papers used rigorous methods to achieve density estimates, performing most roost counts pre-breeding. However, they did include an important assumption about area occupied by the roost. It was assumed that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square, then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). No data were available to verify this assumption. These earlier studies also noted that the number of identified roosts in the study areas did not reach an asymptote, suggesting that the overall densities were underestimates. In addition, the studies were conducted in northern England and northern Scotland, where densities are likely to be lower than in warmer regions of Britain.

The median roost count of 198 is comparable with the estimates provided by experts (235; typical range 20-1500), based on experience at more than 200 sites. It is also very similar to the only published value available in the recent literature (median of 203 individuals (range 30-650) based on 40 roosts (Barlow and Jones, 1999)).

Our estimates excluded colonies surveyed as part of European Protected Species Applications that contained fewer than 30 bats. This ensured that counts did not include individuals in formation roosts that were then counted again at maternity sites. As a consequence, there may have been some underestimation of population size. However, most data were derived from the NBMP. The objective of that project is longitudinal monitoring, so it is likely that non-breeding roosts were included. Given that the estimated roost size is close to expert opinion and published data, it provides a reasonable basis for the calculations performed in this review. Only 2 experts provided estimates of roost density, so there is some uncertainty about whether they are nationally representative.

Three main sources of error are identified. Firstly, the density of maternity roosts in Great Britain, and within each individual country, is highly uncertain. Only two experts provided opinions, and no further information was available from the literature, indicating that there is little or no understanding of soprano pipistrelle roost density. Given the generalist nature of the species, and the likelihood that very large numbers of roosts are unreported, models of roost distribution are likely to be highly speculative. Secondly, the value for the sex ratio of maternity colonies was based on limited data. This may have a substantial effect on the estimate. Finally, no occupancy data or information on trends in density across geographical gradients is available. It has therefore been assumed that the overall roost density estimate applies throughout the range.

Table 10.14e Reliability assessment for soprano pipistrelle bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates†	0	<100 roosts	
	1	<150 roosts	1
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**2**

* Either from the literature or expert opinion with high reliability scores.
† There is no evidence from tree roosts, so this is scored as 1.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

It is difficult to make a direct comparison with Harris et al. (1995) or Arnold (1993), because in those reports the two phonic types were not separated. Harris drew largely on densities estimated in northern Scotland (Speakman et al., 1991), towards the edge of the range for *P. pipistrellus s.l.*, to derive population sizes. The estimated density of bats in the current review 23 bats/km^2) is higher than that used for *P. pipistrellus s.l.* in Harris et al. (1995) (10/km^2).

The distribution of *P. pygmaeus* is similar to that given for *P. pipistrellus s.l* in Arnold (1993) and Harris et al. (1995). It is also similar to that given in the Article 17 Reports (Joint Nature Conservation Committee, 2013b). It is therefore concluded that there has been is no change in range.

Other evidence of changes through time

The National Bat Monitoring Programme (NBMP) includes 601 sites with soprano pipistrelle bats in its field survey and 385 roosts. These have been monitored since 1998 and 1997 respectively. The field survey has recorded a consistent and significant increase in acoustic records of soprano pipistrelle bats, whereas the roost counts have shown a consistent and significant decrease across the survey period (see Table 10.14f). The Bat Conservation Trust notes that roost counts may be unreliable for trend analysis owing to the propensity of the species to switch roosts. Acoustic detectors used to record bat activity in the field have also changed considerably over the recording period, and have become much more sensitive. In addition, volunteer observers find it difficult to distinguish between *P. pipistrellus* and *P. pygmaeus* in the field using heterodyne acoustic detectors: there is considerable misidentification of the two phonic types, and also confusion with *Myotis spp.* (Kate Barlow, *pers. comm.*). The true trend may be intermediate between the two trends reported for common and soprano pipistrelle bats.

Table 10.14f Population trends in soprano pipistrelle bats from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	Number of sites included in trend analysis	Start year for monitoring	Long-term trend) (%)†	Mean annual trend (%)
England	**Field**	**492**	**1998**	**39.7***	**2.1**
	Roost	251	1998	-44.5*	-3.6
Scotland	**Field**	**75**	**1998**	**46.9**	**2.4**
	Roost	86	1997	-50.7*	-4.3
Wales	n/a	n/a	n/a	n/a	n/a
Britain	**Field**	**601**	**1998**	**52.4***	**2.7**
	Roost	385	1990	-47.4*	-3.9

* Indicates that the trend is significant ($p<0.05$).
† Percentage trend since the 1999 baseline year.

Table 10.14g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated. The previous publications did not distinguish *P. pipistrellus sensu stricto* and *P. pygmaeus;* comparisons are therefore made with the data presented for *P. pipistrellus sensu lato.*

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.14h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Collisions with wind turbines.	One of the primary species killed at wind turbines. It is unclear whether these fatalities have local population-level effects.	Mathews et al. (2016)	Negative
Vehicle collisions.	One of the primary species recorded in vehicle collisions. It is unclear whether the scale of casualties is sufficient to affect local populations.	Fensome and Mathews (2016)	Negative
Protection of roosts and	Legislative protection of maternity roosts, in particular, has been introduced to prevent destruction and disturbance.	n/a	Positive
Predation by cats.	One of the species most frequently injured and killed by cats. Where cats are able to access roost entrances, there can be significant impacts.	Andrew Kelly, RSPCA (*pers. comm.*)	Negative
Changes to the structure of buildings and insulation methods.	Changes to building regulations, and efforts to make buildings more energy-efficient, have tended to reduce their accessibility and thermal suitability for bats. Breathable roofing membranes also pose a threat of entanglement.	Waring et al. (2014)	Negative

Data deficiencies

Table 10.14i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts.	n/a	No data available: formal study is urgently required.
Sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available.
Effects of cumulative pressures of land use change, lighting, etc., on local populations.	All	No data available.
Impacts of anthropogenically-induced mortality (wind turbines, vehicles, cats, entanglement in breathable roofing membranes, etc.) on populations.	All	Very limited data available.
Effectiveness of current planning and licensing systems protecting roosts.		No data available.

Table 10.14j An assessment of the future prospects for the soprano pipistrelle bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Stable

10.15 Nathusius' pipistrelle bat *Pipistrellus nathusii*

Habitat preferences

There is a general lack of information on the Nathusius' pipistrelle bat in Great Britain, and until very recently it was considered a vagrant. Most detector records come from within a few kilometres of large freshwater lakes, and this is where recent capture efforts have focused. However, the Nathusius's pipistrelle bat is also associated with other water sources (around coastal areas, estuaries, and canals), as well as being recorded in other areas (e.g., agricultural wind turbine sites, woodland edges and rides).

Only five maternity roosts have been identified in England, and none in Scotland or Wales (Jon Russ, *pers. comm.*). One of these colonies used a bat box, and the remainder were in buildings. Approximately 50 mating roosts have been identified, mainly in buildings (Jon Russ and Daniel Hargreaves, *pers. comm.*). These are occupied by a territorial male who advertises to mates using a song-call.

The Nathusius' pipistrelle bat is widespread across Europe, although its abundance is unclear. It is known to undertake large-scale migrations, with most breeding taking place in north eastern regions and hibernation in the south and west (Hutson et al., 2008; Moussy et al., 2013). Whilst the migration patterns of Nathusius' pipistrelle bat are relatively well known in continental Europe from long-term, large-scale ringing studies (e.g. Hutterer, 2005; Ijäs et al., 2017), the geographical origins of individuals found in the UK, and their migration routes, are much less well defined.

In the UK, this species was considered a migrant winter visitor until the late 1990s (Speakman et al., 1991) when a small number of maternity colonies were found in Northern Ireland and two juveniles were caught in the south east of England. Records from grounded bats and acoustic detectors show peaks of activity, particularly in autumn, but also in spring, suggesting migration into Great Britain. It is thought that the range of this species has been expanding in recent years, possibly linked with climate change (Lundy et al., 2010), in addition to the evident increase in observer effort. Nevertheless, information on the distribution of this species in the UK is still poor: in the most recent Article 17 Report, the population status is assessed as 'Unknown' in the UK and across its range (Hutson et al., 2008; Joint Nature Conservation Committee, 2013b).

Recent increases in capture and ringing effort have revealed the movement of the Nathusius' pipistrelle bat between south west England and the Netherlands, and between Latvia and Estonia and south east England (Daniel Hargreaves, *pers. comm.*). Some of these journeys, of more than 1,000km, have been made in less than 3 weeks. In addition, recordings of Nathusius' pipistrelle bat have been made in the English Channel using acoustic detectors installed on passenger ferries (Fiona Mathews, *pers. obs*.; BSG Ecology, *pers. comm.*). The National Nathusius' Bat Project, run by The Bat Conservation Trust and the University of Exeter with the help of local voluntary bat workers, was established in 2013 to gain a better understanding of the species in Great Britain, particularly of its migratory status. Trapping with the aid of acoustic lures was conducted at 63 sites, all of which were close to large water bodies. Nathusius' pipistrelle bats were captured at 19 of these sites (n=61 individuals). Stable isotope analyses of the fur samples collected as part of the project have provided further evidence that at least part of the British population is derived from the far east of Europe (Barlow et al., 2016). Whilst there is a peak in Nathusius' pipistrelle bat acoustic records and grounded animals in the autumn, corresponding to the hypothesis that some animals come to the UK to hibernate, there are no records of hibernating individuals. The sex ratio of captured individuals was heavily biased: 87% were male. This may be because males are more responsive to acoustic lures, because the trap sites were not close to maternity sites, or because few females are present until later in the season. Of the 8 females captured in the project, only one was caught in early summer, whereas the remainder were captured in autumn at sites across England. The number of male advertising roosts far outnumbers the number of maternity sites (Jon Russ, *pers. comm.*). Nevertheless, it is likely that at least some of the population is also permanently resident (or arrives in spring), since volant juveniles and lactating females have been found at various locations in England in early August.

Status

Native.

Conservation Status

- IUCN Red List (GB: NT; England: [NT]; Scotland: [VU]; Wales: [VU]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Unknown; England: Unknown; Scotland: Unknown; Wales: Unknown).

Species' distribution

Records for the Nathusius' pipistrelle bat are highly dispersed. This reflects the relatively short time for which appropriate acoustic recording equipment has been widely available, and also the localised nature of concentrated survey effort. The extent to which records reflect individual migrants and vagrants, rather than larger populations, is unclear. In Scotland, no colonies of this species have been recorded.

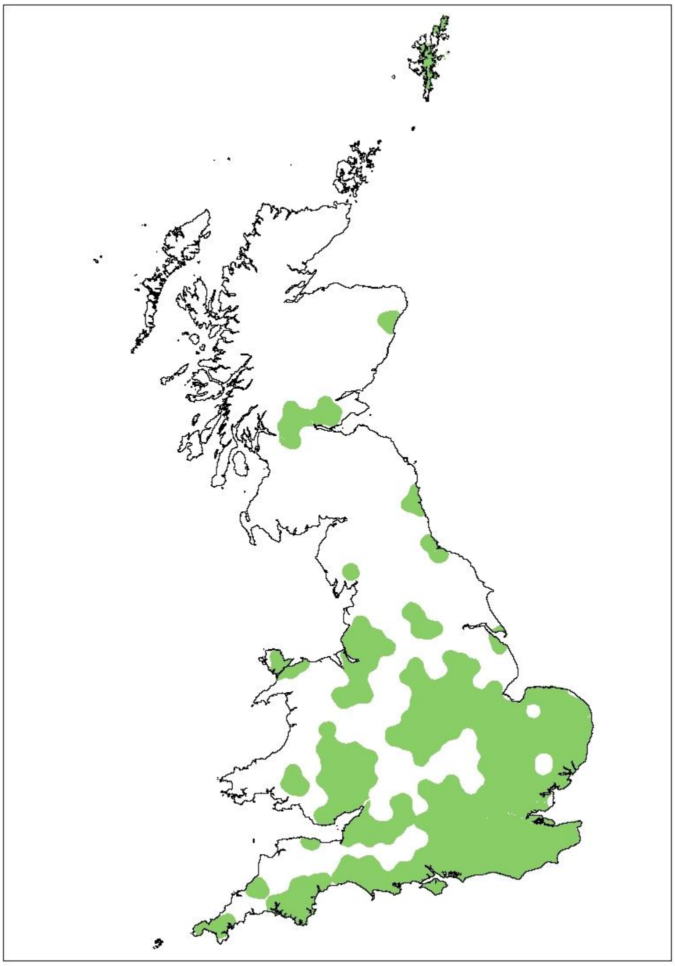


Figure 10.15a Current range of the Nathusius' pipistrelle bat in Britain. Range is based on presence data collected between 1995 and 2016. There are no known roosts of any bat species in Shetland, so records are therefore likely to be from vagrant or migrant individuals. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

Because of the limited amount of study on this species, expert input was constrained to individuals experienced with working on Nathusius' pipistrelle bats. Data were available from 5 maternity sites, but counts included adult females and juveniles (Russ, 2014). The mean colony count was 32 individuals (range 5-80), but the data were not generally gathered pre-parturition. On the assumption that approximately one third of the individuals are juveniles, this equates to an adult count of 21 individuals (range 3-53). This is somewhat smaller than roosts visited in Northern Ireland and the Republic of Ireland, which hold up to 200 individuals each. No information was available on the sex ratio of colonies pre-parturition, or on the sex ratio of the population as a whole. Information on the density of maternity roosts was also lacking. However, several lines of evidence suggest that maternity sites of the species are under-recorded:

1. The species is frequently recorded using acoustic detectors throughout the maternity season.
2. Individuals are captured relatively easily when appropriate techniques are used at suitable sites.
3. Approximately 100 advertising sites used by males in autumn have been identified.

Results

Population estimation and range

The lack of information on roost (or colony) density makes population estimation extremely difficult. No alternative sources of information (e.g., from population genetics) are available for the UK. No estimate was made in the most recent Article 17 Report for any country (Joint Nature Conservation Committee, 2013b). Given the number of individuals that have been captured and ringed, and assuming that these are only a fraction of the total population, it is likely that there is a population of at least several hundred in Great Britain.

The current distribution of the species, based on known records since 1995, is shown in Table 10.15a. This range is derived from all record types, most of which are acoustic. Given the species' great mobility, the range therefore may not correspond with the roost range.

Table 10.15a Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	70,300	n/a
Scotland	4,210	n/a
Wales	6,920	n/a
Britain	**81,400**	**149,400**

Table 10.15b Reliability assessment for Nathusius' pipistrelle bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

The number of Nathusius' pipistrelle bat acoustic records has increased rapidly over the past decade. This is partly owing to increased observer effort, coupled with the greater ease with which the species can be identified using modern equipment compared with heterodyne detectors. Records of grounded bats have also increased, and this is likely to reflect increased awareness of the presence of the species in the UK and improved identification. Nevertheless, the scale of the change is such that it seems reasonable to infer that there is also a genuine increase in the number of Nathusius' pipistrelle bats in Great Britain.

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Harris et al. (1995) considered the species to be a migrant winter visitor with an unknown population size. Arnold did not report a range for this species, and recorded it instead as a rare vagrant. Given the recent increases in observer effort, the current range is therefore considered more appropriate than the previous report. The alpha hull estimate of range size is smaller than the surface range given in the Article 17 Report: this is likely to be because of methodological differences.

Other evidence of changes through time

No further evidence is available.

Table 10.15c Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995).

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort.

Drivers of change

Table 10.15d Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Climate change.	Alteration to migration routes and summering/wintering grounds.	Lundy et al. (2010)	Positive

Data deficiencies

Table 10.15e Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts.	n/a	No data available: formal study is urgently required.
Roost size and structure.	n/a	No data available.
Occupancy of different regions.	All	No occupancy data or information on trends in density across geographical gradients are available.
Impacts of wind turbines.	Offshore and onshore	As this is the only species with clear evidence of considerable movement between Great Britain and continental Europe, it is vital to clarify the risk posed by offshore turbines which may affect migratory routes. The species is also known to be at high risk of collision (based on evidence elsewhere in Europe). To date, only a single onshore wind farm casualty has been identified, but few sites in coastal or other high-risk areas have been monitored.
Locations of migratory routes.	Coastal and offshore areas	The migratory routes are currently not known. This means that it is currently impossible to ensure that routes are protected as required under the Bonn Convention. It is also not known whether any movement occurs between Great Britain and Ireland (where the species is relatively common).

Future prospects

Table 10.15f An assessment of the future prospects for the Nathusius' pipistrelle bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Unknown
Habitat	Stable

10.16 Barbastelle bat *Barbastella barbastellus*

Habitat preferences

The barbastelle bat is highly dependent on broadleaved woodland. It is a specialist moth feeder (>99% of diet; Sierro and Arlettaz, 1997; Zeale, 2011), although it occasionally takes other items in the winter (Rydell and Bogdanowicz, 1997). Radio-tracking shows that riparian margins and broadleaved woodland are strongly selected for foraging, but that unimproved grassland, field margins and hedgerows are also important (Zeale et al., 2012). It is likely that the relative importance of these areas varies seasonally, reflecting changes in moth abundance. The species makes fast and direct flights to core foraging areas, where it usually forages at heights of 4m-5m (or within woodland canopies). Radio-tracking in southern England has shown that the mean core range of females from maternity colonies is 8km, but they can fly long distances rapidly, frequently crossing very open habitat including downland and moorland, to reach other woodlands or core foraging areas up to 20km away (Greenaway, 2001; Zeale, 2011). These flights are often at low level (<2m from the ground). The core foraging areas often form only a small fraction of the total home range (Zeale et al., 2012). With the advent of widespread use of static acoustic detectors, it has become apparent that the species is widely distributed — although never common — across the rural landscape of southern Britain and parts of Wales. Work in Italy has shown that the barbastelle bat can continue to use formerly forested landscapes long after they have changed to apparently unsuitable habitat, indicating that habitat suitability models based on woodland availability must be used with great caution (Ancillotto et al., 2015).

In Great Britain, the first maternity colonies were only identified in 1997. More than 30 maternity roosting locations have now been found, all of which are in tree holes. Whilst in Great Britain there appears to be a preference for old or dead oak, almost any tree with suitable cavities can be used (Zeale, 2011), and elsewhere in Europe the species preferentially roosts in beech trees (Russo et al., 2004). Caution must therefore again be used before inferring habitat suitability from woodland composition. Maternity colony sites and foraging areas are often close to riparian habitat (Greenaway, 2001; Zeale, 2011). Maternity colonies are sometimes found in buildings elsewhere in Europe, particularly in areas with little woodland. Although individual bats are found in barns and other buildings in Britain (Gareth Harris, *pers. comm.*), there is only one known roost in a building (Paston Great Barn NNR, Norfolk). Maternity colonies are small (<30 females), and the animals move their nursery roosts very frequently whilst remaining loyal to a general area (Russo et al., 2005). In addition, the colonies frequently fragment into smaller subunits. Males appear solitary for most of the year and often roost in cracks in trees or under peeling bark. Occasionally, individual males are found roosting in maternity groups.

The species is found regularly, although in low numbers, at underground sites including disused railway tunnels and ice-houses: the large clusters (which can include >1,000 individuals) observed in eastern Europe (Rydell and Bogdanowicz, 1997; Schober, 2004) are not recorded here. Individuals are also sometimes found at sub-zero temperatures under the loose bark of trees; and groups of animals can use hollow trees and large bat boxes for hibernation. Barbastelles are also caught in small numbers at swarming sites in late summer and autumn (Fiona Mathews, *pers. obs*.; Daniel Hargreaves, *pers. comm*.; Keith Cohen, *pers. comm*.). Individuals are also regularly captured entering large barns in the middle of the night in late summer, perhaps making use of them as night-roosts or mating locations (Fiona Mathews, *pers. obs.*).

Elsewhere in Europe, the species is known to undertake large-scale movements of up to 290km (Rydell and Bogdanowicz, 1997). It is generally assumed to be sedentary in the Great Britain, but no direct evidence exists to support or refute this assumption.

Status

Native.

Conservation Status

- IUCN Red List (GB: VU; England: [VU]; Scotland: n/a; Wales: [VU]; Global: NT.).
- National Conservation Status (Article 17 overall assessment 2013. Annex II and IV; UK: Unknown England: Unknown; Scotland: n/a; Wales: Unknown).

Species' distribution

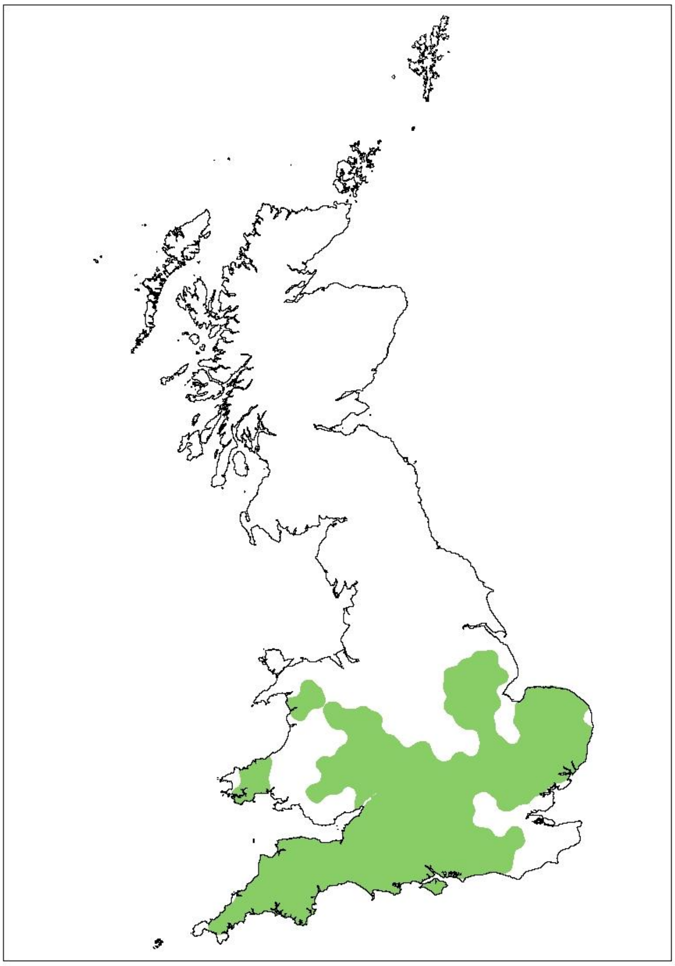


Figure 10.16a Current range of the barbastelle bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

The barbastelle bat is generally considered to be a broadleaved woodland specialist. Experts were therefore asked to provide estimates of roost density within this habitat. Expert opinion on the species was obtained from 4 individuals. A further 5 experts responded to requests for input on this species, but were unable to provide information on the variables needed. One expert provided an opinion on a likely total national population estimate. The values given by experts for typical pre-breeding maternity roost sizes (with PIs), based on experience with 20 colonies, were 25 (7-40); 30 (20-40); 50 (20-80). The median typical roost size is therefore 30 individuals, and the median values for the lower and upper plausible limits are 20 and 40.

No data were available from the literature or from Harris et al. (1995) on the sex ratio of maternity roosts pre-breeding. Two experts commented that they had only ever caught females at maternity roosts, but most of this trapping avoided the period immediately pre-breeding in order to minimise disturbance. Information on the overall sex ratio of the population is not available in the literature or elsewhere. No expert was able to provide an estimate of roost density. The propensity of colonies of this species to fragment very frequently was noted by 2 of the experts; it is therefore particularly challenging to derive a density estimate for this species.

Habitable area was defined as all broadleaved woodland within the range. Although the species forages beyond woodlands, roosts and most records are usually associated with this habitat.

Results

The values used to derive the density estimates are shown in Table 10.16a.

Table 10.16a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	30 (20-40)
Sex ratio	n/a
Maternity roost density	n/a

Population estimation and range

It was not possible to derive a population estimate for this species because of a lack of evidence.

The estimates in the most recent Article 17 Report are provided in Table 10.16b.

Table 10.16b Article 17 Report on barbastelle bat population sizes 2007-2013 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	4,500	4,500
Scotland	0	0
Wales	500	500
Britain	**5,000**	**5,000**

Note: maximum and minimum estimates were the same values in the country-level reports.

The current geographical range of the species, based on known records since 1995, is shown in Table 10.16c. The total habitable area for Great Britain (broadleaved woodland within the range) is 6,100km^2.

Table 10.16c Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	67,600	n/a
Scotland	0	0
Wales	6,390	n/a
Britain	**74,000**	**90,500**

Critique

Very little information is available for this species, so it was therefore impossible to make a population size estimate. Further information on occupancy is also urgently required in order to estimate the range more precisely.

Four main sources of error are identified. Firstly, the density of maternity roosts in Great Britain, and within each individual country, is entirely unknown. The extent to which maternity colonies can use isolated trees is also unknown. (Therefore, basing population estimates solely on broadleaved woodland may be unsafe.) Secondly, no occupancy data are available for woodlands of different structure or in different regions. The ability of barbastelle bats to use almost any type of tree with suitable cavities further compounds the difficulty of creating habitat suitability models for this species. Information on roost size is based on very limited information, and the relationship with overall colony size is unclear. Finally, this species is recorded infrequently but regularly by acoustic detectors and found across a wide geographical area in the south of England and Wales. The lack of a central repository for acoustic data hinders the precise definition of the species' range.

Table 10.16d Reliability assessment for barbastelle bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	0
	1	Yes	
		Overall reliability score	**0**

* Either from the literature or from expert opinion with high reliability scores.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The population size is currently unknown. It is therefore impossible to determine whether there has been any change over time. There was no evidence base for the population estimate of 5,000 (4,500 in England and 500 in Wales) given for Great Britain in Harris et al. (1995).

The distributions reported in these previous reports were based on very sparse data compared with the data currently available. Whilst Arnold suggested that there had been a serious decline in the population, based on the difference in the range of the species inferred from records up to 1959 compared with those from 1960 onwards, the current data indicate that range is similar to all available historical data with the exception that there are no longer any records north of the Humber (whereas Arnold (1993) shows positive hectads in South Yorkshire).

The slight difference in range compared with the Article 17 Report (Joint Nature Conservation Committee, 2013b) is partly because of methodological differences. However, the recent changes in detector technology also ensure that the current estimate is likely to be more accurate than previous reports.

Other evidence of changes through time

None available.

Table 10.16e Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				England Wales

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort. Records from Arnold (1993) are scattered throughout the species' current range, with no increase in overall range size (Figure 10.16a).

Drivers of change

Table 10.16f Drivers of population change between 1995 and the present. Drivers are limited to those affecting the population at a national level.

Driver	Mechanism	Source	Direction of effect
Unknown.*			

* There are insufficient data on population change to permit drivers of change to be identified.

Data deficiencies

Table 10.16g Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts & sex ratio of adults in maternity colonies pre-breeding.	n/a	No data available: formal study is urgently required to examine both occupancy and abundance across geographical and habitat gradients. Alternatively, a population-genetics approach may be used to estimate abundance.
Effects of cumulative pressures of land use change, lighting, etc., on local population, particularly through the fragmentation of habitat which may restrict access to core foraging areas.	Woodland edge, riparian corridors	No data available (species is light-sensitive).
Impacts of anthropogenically-induced mortality (wind turbines, vehicles collisions, cats, etc.) on populations.	All	No data available for most of these threats. Collisions with road vehicles are recorded elsewhere in Europe.
Impacts of woodland management, particularly alteration to understory and management of deadwood.	Woodland	No data available.
Migratory status.	All	No data available for Great Britain; the species is known to make long-distance movements elsewhere in Europe, so could potentially be migratory.
Impact of agri-environment schemes on moth abundance and foraging activity by the species.	Farmland	There is concern that widely-reported declines in abundance of many moth species will have a negative impact (Conrad et al., 2006b). Although it is commonly thought of as a woodland bat, the barbastelle also forages outside woodland. It is therefore likely that wet meadows, field margins, etc., are important for the species.

Future prospects

Table 10.16h An assessment of the future prospects for the barbastelle bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	**Status**
Population	Unknown
Range	Unknown
Habitat	Decline

10.17 Brown long-eared bat *Plecotus auritus*

Habitat preferences

The brown long-eared bat gleans approximately half its prey from vegetation, and catches the remainder in the air (Swift and Racey, 1983; Anderson and Racey, 1991; Anderson and Racey, 1993). Gleaning is facilitated by its capability to hover in addition to using slow horizontal flight (Norberg, 1976b; Norberg, 1976a). It is adapted to foraging in cluttered habitats, and makes extensive use of sight, passive listening, and short duration echolocation (Anderson and Racey, 1991; Anderson and Racey, 1993; Eklöf and Jones, 2003). A high proportion of its diet is Lepidoptera (particularly noctuid moths) and Coleoptera (beetles), but it takes a range of large insects (≥ 3mm body length) as well as non-flying prey (Vaughan, 1997; Swift, 1998).

The species is commonly associated with trees, particularly broadleaved and mixed woodland, and it can fly at a variety of heights, including within the canopy. It also makes use of native conifers such as Scots pine, but tends to be found only at the edge of commercial conifer plantations (Entwistle et al., 1996). It uses linear features such as treelines and large hedgerows to move between roosts and alternative foraging areas (Howard, 1995; Murphy et al., 2012), and individuals are regularly captured in nets placed in these locations. It also forages around trees in more open habitats, including parks, orchards and gardens (Dietz and Keifer, 2016).

Maternity roosts are located in trees, bat boxes and buildings — predominantly barns, churches and dwelling houses with large internal flight spaces (Boyd and Stebbings, 1989; Dietz and Keifer, 2016). In one region, a preference for old stone buildings was found (Moussy, 2011). There is also evidence for a link between maternity roost location and nearby presence of broadleaved woodland (Boughey et al., 2011; Moussy, 2011). Individuals in the north east of Scotland have been found to travel up to 2.8km to forage, but most activity occurred within 500m of the roost (Entwistle et al., 1996), corresponding with data elsewhere in Europe (Dietz and Keifer, 2016). In England, females in the maternity period have been found to return repeatedly to non-overlapping core foraging areas which averaged 2.1ha (range 0.7ha-5.4ha) (Murphy et al., 2012).

Maternity roosts contain adult males as well as females, although with some female bias (Park et al., 1998; Entwistle et al., 2000). There is a high degree of fidelity to roosts by both sexes (Park et al., 1998; Entwistle et al., 2000), with evidence of natal philopatry, yet colonies do not appear to be inbred (Burland et al., 1999; Burland et al., 2001). Swarming sites therefore appear particularly critical for brown long-eared bat conservation because of their contribution to genetic exchange (Burland et al., 2001; Furmankiewicz and Altringham, 2007; Furmankiewicz, 2008), and bats may travel considerable distances to reach them (e.g., >30km recorded in Poland Furmankiewicz (2008)). Yet the species forms only a very low proportion of total captures at swarming sites (Parsons et al., 2003b). It is generally considered to be non-migratory across Europe (Dietz and Kiefer, 2016), and no long-distance movements between maternity and hibernation sites have been recorded in Great Britain. Underground sites including tunnels, caves and ice-houses are used for hibernation, but the extent of tree use is unclear (Swift, 1998; Glover and Altringham, 2008). Brown long-eared bats fly very frequently, and sometimes daily, during the winter (Daan, 1973; Hays et al., 1992), and so habitat quality around hibernacula is likely to be very important to their conservation.

Status

Native.

Conservation Status

- IUCN Red List (GB: LC; England: [LC]; Scotland: [LC]; Wales: [LC]; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Favourable; England: Favourable; Scotland: Favourable; Wales: Favourable).

Species' distribution

A species' distribution map is provided in Figure 10.17a. Gaps in the species' distribution in Scotland are likely to represent areas lacking survey effort, rather than true absences, with the exception of the areas in the far north.

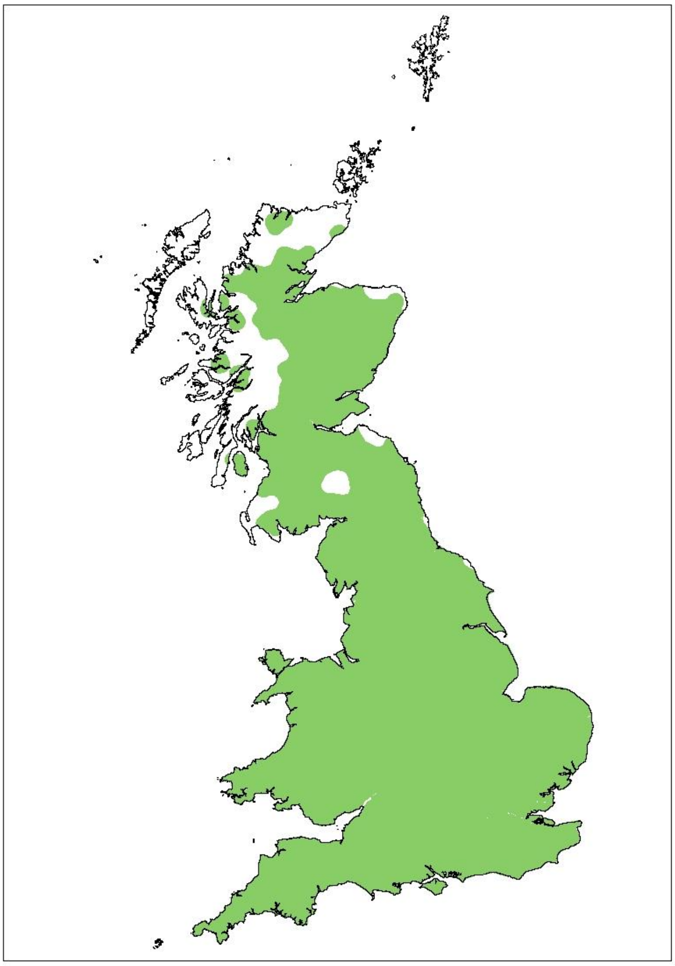


Figure 10.17a Current range of the brown long-eared bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific Methods

Information was available from our datasets for 397 sites (including sites monitored as part of the National Bat Monitoring Programme and European Protected Species Licence Applications). The median number of bats per roost was derived from the peak counts pre-breeding in the most recently available year. The most recently available peak count before July was used for the analyses. Small roosts were not excluded from the assessment because the fission-fusion social structure of the species means that colonies are divided across several roosts: therefore, even those locations with <10 bats can include breeding individuals. The median pre-breeding roost size for brown long-eared bats derived from the available datasets was 10 individuals (95%CI = 9-13, range = 1-144, n=397 roosts).

Consideration was also given to previously published values (Entwistle et al., 2000), because of reports in the literature that roost counts are unreliable for this species. Capture-mark-recapture analysis suggests that the observed numbers of individuals in roosts is likely to underestimate the true population. Entwistle et al. (2000), working on 1,365 bats ringed across 30 summer roosts, found that 10-20 bats were typically observed. However, analysis of intensive recapture data from 12 of these sites (mean roost count 16 bats; the same value as observed by Speakman et al. (1991)) produced estimates of 30-50 bats per roost. This discrepancy was explained by the difficulty of conducting emergence surveys with a late-emerging species, and the possibility that not all individuals within roosts are readily visible on internal inspections (Entwistle et al., 2000). It is also possible that colony estimates deviate from roost counts because individuals are distributed across numerous adjacent sites, switching roosts every few days, as seen in brown long-eared bat populations using bat boxes (Danielle Linton, *pers. comm.*; Dietz & Kiefer 2016). This social structure has also recently been demonstrated for building-dwelling soprano pipistrelle bats (*P. pygmaeus*) using intensive radio-tracking (Stone et al., 2015). Individuals in buildings may be more faithful to an individual roost, possibly because buildings frequently offer several different potentially suitable locations: intensive radio-tracking of 16 individuals by Entwistle et al. (1996) found no evidence of roost switching. Whatever the correct explanation, the inflation factor derived by Entwistle et al. (2000) is useful because it accounts for the high proportion of the total population that is not observed despite intensive survey effort.

Expert opinion was obtained from 7 individuals. A further 2 experts responded to requests for input on this species, but were unable to provide information on the variables needed. Three estimates of the proportion of female bats in pre-parturition roosts were available from the literature: 70% and 63% in northern Scotland (Speakman et al., 1990; Entwistle et al., 2000), and 65% in southern England (Park et al., 1998). The median of these values (65%) was used to derive the number of adult female bats in a roost from the total counts. Overall, in the population, the sex ratio pre-parturition appears to be 1:1 (Park et al., 1998).

Estimates of roost density were not available from experts, so information was derived from the literature. An intensive search for roosts in buildings in northern Scotland identified 30 roosts in a 1000km^2 area, giving a density of 0.03 roosts/km^2 (Entwistle et al., 2000). Jones et al. (1996) reported a density of 0.08 roosts/km^2 based on building roosts in Yorkshire. The estimates in Jones et al. (1996) relied on an assumption that the foraging area of each roost was the 5km x 5km grid square that the roost was located in, and if one or more roosts fell within a particular square, then that square was used as part of the density calculation, whereas squares without records were excluded entirely (following Speakman et al., 1991). Given that no data were available to verify this assumption, a second density estimate was derived for the purpose of the current calculations by using the entire 2,500km^2 study area (which gives a density estimate of 0.02 roosts/km^2).

Battersby reported building roost densities in Sussex ranging from 0.14 roosts/km^2 (focal study area in east Sussex together with scrutiny of Natural England database for the region) to 0.09 roosts/km^2 (based on a random survey of the whole of Sussex). There was also one estimate from Harris (2014) of maternity roost densities in a 100km^2 survey area derived from a local 10-year survey initiative. This project found a density of 0.17 roosts/km^2.The median of all the values reported above was used as the estimate of typical roost density.

The upper and lower limits for the plausible intervals used in computing the population size were defined as follows:

- Roost size: upper and lower 95% confidence limits for the median roost size.
- Sex ratio: upper and lower plausible values from the literature.
- Roost density: number of roosts/typical km^2 for poor quality habitat and for high quality habitat. The highest and lowest values reported in the literature were taken to represent the plausible density in poor and good quality habitat respectively. The distribution of the values gives confidence that the upper and lower values are not extreme outliers, and therefore that their use is reasonable.

The population estimate was calculated on the basis of adult bat density (bats/km^2) and the geographical range. Density was calculated as follows:

*Median density = [(median n. bats/roost†) * (p♀‡) * (n roosts/typical km^2 average habitat)]* 2*

*Lower limit = [(lower plausible n. bats/roost) * (p♀min) * (plausible n. roosts/typical km^2 poor habitat)]* 2*

*Upper limit = [(upper plausible n. bats/roost) * (p♀max) * (plausible n. roosts/typical km^2 good habitat)]* 2*

† Roost' is a typical maternity roost in the pre-parturition period. n. is the number of adults.

‡ p♀: proportion female. p♀min and p♀max are the lowest and highest plausible proportions of adult females in a typical maternity roost.

The population estimate was based on adult population density and extent of occupancy across all habitat types within the range. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

Population size

*Total Adult Population = Median adult density (bats/km^2) * total habitable area within range (km^2)*

*Lower limit = Lower limit adult density (bats/km^2) * total habitable area within range (km^2)*

*Upper limit = Upper limit adult density (bats/km^2) * total habitable area within range (km^2)*

Results

The values used to derive the density estimates are shown in Table 10.17a. No data were available on tree roost density. Estimates were therefore based on roost density in buildings only. The following values were used:

Table 10.17a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	40* (10**-50†)
Sex ratio	1
Maternity roost density (roosts/km^2)	0.09†† (0.02‡-0.17‡‡)

* Based on Entwistle et al., 2000.
** Based on the median value from our datasets.
† Based on Entwistle et al., 2000, reporting that most roosts truly contained 30-50 bats.
†† Based on the median of values provided in: Jones et al., 1996; Battersby, 1999; Entwistle et al. 2000; Harris, 2014.
‡ Jones et al. (1996): see Species-specific Methods.
‡‡ Harris (2014).

Population estimation and range

Given the absence of data on roost density in trees, it is difficult to compute a total population estimate. It is considered unlikely that most maternity roosts in Britain are known, so it has not been possible to make a total count. No population genetics study has been conducted to estimate regional or national population sizes, and therefore no alternative metrics of population size were available.

Table 10.17b Area of suitable habitat within the species' range and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	130,000	4.65	0.26	11.1	607,000	33,700	1,430,000
Scotland	49,100	4.65	0.26	11.1	230,000	12,800	543,000
Wales	20,600	4.65	0.26	11.1	96,600	5,370	228,000
Britain	**200,000**	**4.65**	**0.26**	**11.1**	**934,000**	**51,900**	**2,200,000**

The estimates in the Article 17 Report on brown long-eared bat status 2007-2012 (Table 10.17c; (Joint Nature Conservation Committee, 2013b)) are less than a quarter of the main values estimated in this review, but the plausible ranges include the values given in those reports for each country and for Great Britain.

Table 10.17c Article 17 Report on brown long-eared bat population sizes 2007-2012 (Joint Nature Conservation Committee, 2013b).

Country	Minimum	Maximum
England	155,000	155,000
Scotland	27,500	27,500
Wales	17,500	17,500
Britain	**200,000**	**200,000**

Note: maximum and minimum estimates were the same values for this species.

The current distribution estimate for the species is based on known records of brown long-eared bats since 1995, and is shown in Table 10.17d.

Table 10.17d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	130,000	n/a
Scotland	49,100	n/a
Wales	20,600	n/a
Britain	**200,000**	**226,000**

Critique

The very large range of plausible values for the population estimate, and the very large alterations that could potentially be generated by including higher density estimates for woodland, emphasise the uncertainty around the density and population size of this species. The differences between the upper and lower plausible limits are generated in roughly equal measure by the uncertainty around the roost size (the upper limit being 5 times the lower one) and the roost density (the upper limit being 8.5 times the lower one).

The roost size estimated from the available dataset was similar to the mean value of 12 individuals reported by Entwistle et al. (2000), and slightly smaller than earlier reports (mean = 17 (Jones et al., 1999); mean = 16 (Speakman et al., 1991)). Excluding small roosts (<10 bats) made little difference to the median value. Given the similarity of the observations to those of Entwistle, the revision of colony sizes based on capture-mark-recapture estimates is justified. However, this revision makes more than a three-fold difference to the total population estimate, and it is possible that Entwistle's study, which was based in the north east of Scotland, may not apply to other regions of Great Britain. No data were available on roost sizes in trees; these may differ substantially from building or bat-box roosts.

Roost density estimates were derived from the literature and were entirely based on buildings. Although numerous estimates were available, providing some confidence in the range of values assumed for buildings, data were not available for tree roosts. Given that a high proportion of roosts is likely to be in trees, and roost density may be much higher here, this potentially introduces a major source of error. The only available data for woodland comes from a population using bat boxes in a 400ha woodland (largely broadleaved) in southern England (Danielle Linton, *pers. comm.*), where a minimum annual population of 150 adult bats is observed. This gives a density estimate of 37.5 bats/km^2. As there are 13,333km^2 of broadleaved woodland in Britain, most of it within the species' range, accounting for this habitat could add almost 500,000 additional bats to the estimates if bats use natural tree roosts in the same way as bat boxes. However, it is not clear whether the bats roosting within woodland make extensive use of other habitats outside the wood. If they do, and if bats roost within the woodland in preference to buildings in the surrounding area, then the estimated density would be much too high. In addition, the provision of bat boxes may artificially enhance the density of bats, making this woodland atypical.

One of the largest sources of error is the widescale under-recording of tree roosts. The range in the west of Scotland and the Scottish Borders may be more extensive than estimated here. Acoustic surveys are generally a poor method of assessing the species because of its low amplitude calls (Russ, 2012). There is also potential for the species to be overlooked in open habitats, such as wind farms, as its calls differ substantially from those used in more enclosed areas (Fiona Mathews *pers. obs.*); and because the calls can also be confused with those of *Myotis spp*, particularly when heterodyne detectors are used (Russ, 2012).

No expert could provide estimates of roost density. Three experts commented on the lack of data for tree roosts, and one reported that, in his extensive experience of radio-tracking, most female brown long-eared bats roosted in trees rather than buildings. This emphasises the potential for distributions and densities to be underestimated in this report.

Six experts provided information on roost size, whilst the other two were unable to contribute the information necessary for the calculation of population sizes. Their estimates of usual roost counts (usual size 25 individuals; typical range = 13-75, n = 218 roosts) were larger than those derived here, but lie within the plausible values (10-50) used in this report.

There is some discrepancy between the sex ratios reported in the literature in pre-breeding maternity roosts and the experience of two experts who reported that >80% of individuals captured from roosts were female.

Four main sources of error are identified. Firstly, no roost counts or density estimates are available for tree roosts. Secondly, the ratio of building:tree roosts is unknown, meaning that the scale of bias introduced by basing estimates primarily on data from buildings is unquantifiable. It is also unclear whether the ratio of observed:true colony size estimated by

Entwistle et al. (2000) in northern Scotland applies to the rest of Britain. Finally, the range may be underestimated in some parts of Scotland, particularly where there is little potential for roosts in buildings, as long-eared bats are strongly under-recorded using acoustic surveys, and tree roosts are difficult to find.

Table 10.17e Reliability assessment for brown long-eared bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	
	1	<150 roosts	1†
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1
		Overall reliability score	**2**

* Either from the literature or from expert opinion with high confidence scores.
† Scored as 1 because although extensive data were available, the reliability is very uncertain for this species (Entwistle et al., 2001). No data were available for tree roosts.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

Although a population estimate of approximately 200,000 individuals was given in Harris et al. (1995) (England 155,000; Scotland 27,500; Wales 17,500), this estimate was graded as having very poor reliability and was largely derived from expert opinion on the ratio of brown long-eared to pipistrelle bats (roosts and individuals). Direct comparison is therefore not possible.

The distribution is similar to that reported by Arnold (1993). The range is slightly smaller than that given in the JNCC Article 17 Report (Joint Nature Conservation Committee, 2013b); this is likely to reflect the methodological differences.

Other evidence of changes through time

The National Bat Monitoring Programme hibernation and roost count data do not indicate any change over time. No data are available from field surveys.

Table 10.17f Trends in brown long-eared bat activity from baseline to 2015, as estimated by the National Bat Monitoring Programme (Bat Conservation Trust, 2016). Insufficient data were available for Scotland to estimate trends. Results shown in bold are considered the more reliable index by the NBMP where more than one type of survey is available.

Country	Type of site	No. sites included in trend analysis	Start year for monitoring	Long-term trend (%)†	Mean annual trend (%)
England	**Hibernation**	**316**	**1998**	**39.7**	**0.7**
	Roost	112	1990	5.0	0.4
Wales	Hibernation	106	1998	43.5	2.3
	Roost	n/a	n/a	n/a	n/a
Britain	**Hibernation**	**444**	**1990**	**-7.3**	**-0.5**
	Roost	157	1990	28.2†	1.8

* Indicates that the trend is significant ($p<0.05$)

† Percentage trend since the 2001 baseline (few roosts having been monitored before this date).

Table 10.17g Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

		Range			
		Increase	**Stable**	**Decrease**	**Data deficient**
Population size	**Increase**				
	Stable				
	Decrease				
	Data deficient				All countries*

* Definitive comparisons with earlier distribution maps cannot be made because of changes in acoustic monitoring techniques and observer effort. Records from Arnold (1993) are scattered throughout the species' current range.

Table 10.17h Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	**Mechanism**	**Source**	**Direction of effect**
Increased availability of broadleaved woodland and bat boxes.	Increased roosting opportunities.	No reference	Positive
Loss of viable roosts during barn and other building conversions.	Reduction in roost suitability, particularly a reduction in loft area.	MacKintosh (2016) Waring et al. (2013)	Negative
Urban development encroaching on traditional roosts.	Loss of foraging habitat and increased isolation of roosts in the landscape. The species is thought to be poor dispersers owing to wing morphology.	Ekman and Jong (1996) Entwistle et al. (2000)	Negative
Impact of road casualties on local populations.	Collisions with vehicles.	Fensome and Mathews (2016)	
Artificial night lighting.	Species is extremely light-shy. Lighting potentially severs commuting routes and reduces moth availability.	Plummer et al. (2016) and inferences from other light-shy species	
Change of habitat and prey abundance in agricultural landscape.	Decline in moth populations.	Conrad et al. (2006)	Negative
Coppicing of understory and introduction of woodland grazing.	Removal of diverse and dense understory important to foraging bats.	Murphy et al. (2012)	Negative

Data deficiencies

Table 10.17i Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts.	All	No data available in woodlands. Density is poorly estimated in other habitats.
Proportions of roosts found in trees compared with buildings.	n/a	No data available. Information is required in order to assess bias introduced by deriving estimates from roosts in buildings, and to assess the conservation importance of woodlands.
Roost size in trees and buildings.	Buildings and trees	Thermal imaging/infra-red video-photography and/or genetic approaches are needed to improve estimates. Intensive radio-tracking of bats in building roosts would identify whether a colony is divided across multiple roosts.
Effects of cumulative pressures of land use change and urban encroachment on roosts.	All	No data available.
Impacts of road casualties on British populations.	Roads	No data available.
Impacts of change in agricultural practice, particularly management of field margins and hedgerows, and use of insecticides, on prey abundance and local bat population sizes.	Agricultural land	No data available.
Impacts of changing woodland management, affecting total woodland area and amount of standing deadwood, on roost availability.	Broadleaved woodland	No data available
Effectiveness of mitigation for development in maintaining functionality of roosts in buildings.	Buildings	Very limited data available.

Future prospects

Table 10.17j An assessment of the future prospects for the brown long-eared bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Unknown
Range	Stable
Habitat	Stable

10.18 Grey long-eared bat *Plecotus austriacus*

Habitat preferences

The grey long-eared bat is a cryptic species, very similar in morphology and flight pattern to the brown long-eared bat, *Plecotus auritus*. Very few colonies are known in Great Britain, and these are almost exclusively found in lowland regions of southern England, close to the coast. The grey long-eared is a specialist moth feeder, with Lepidoptera — notably noctuid moths — forming approximately two-thirds of the diet (Bauerova, 1982; Czech Republic; Razgour et al., 2011a; England). Most of the remainder of the diet identified in England was Diptera (flies), particularly craneflies *Tipula oleracea,* but in contrast to research from the Czech Republic, the English research did not identify large chafers and bugs in faecal specimens. However, the sample size was relatively small (n=30 bats caught at 2 locations in Devon and Isle of Wight).

Like the brown long-eared bat, the grey long-eared appears to feed primarily on common Lepidopteran species. The findings of dietary studies correspond with limited radio-tracking evidence from England (fine-scale tracking from two roosts (Razgour, 2012)) and continental Europe, showing that the grey long-eared bat forages in grassland habitats, including meadows and woodland edges, whereas the brown long-eared bat forages primarily within woodland. This spatial separation of foraging habitat, rather than differential prey selection, is thought to be the mechanism by which the two species can co-exist within the same areas (Razgour et al., 2011a).

Radio-tracking evidence from 28 bats studied across 3 maternity roosts (Devon, Isle of Wight and West Sussex (Razgour et al., 2013)) indicates that the mean home range is 4.6km^2. The colony home range was found to vary between locations (17.4 km^2-37.2km^2), and the estimate may be affected by the number of radio-tracked bats and foraging habitat quality. Several different foraging areas were used each night. These areas were located up to 5km away from the maternity colony roost, with around half of all core foraging areas being found more than 2km away.

All maternity roosts in Great Britain are in the loft spaces of residential buildings. The roof spaces used by grey long-eared maternity colonies tend to be large (they typically use Victorian buildings) and include a roof lining of wood or bitumastic underfelt. There is a single report (not in Britain) of use of a bat box (Kowalski and Lesiński, 1994). The hibernation sites for the species in Great Britain are unknown. Elsewhere in Europe, they hibernate in cellars, attics, underground galleries, mines, quarries, caves and rock crevices (Horácek, 1975; Swift, 1998; Dietz and Keifer, 2016). Hibernation sites are usually located within less than 30km from summer roosts, but distances may range between 5km and 61km (Hutterer, 2005; Ijäs et al., 2017). Based on this evidence from continental Europe, and the broad wing structure which is inefficient for long-distance flight (Norberg and Rayner, 1987a), the species is considered to be sedentary.

Ecological niche modelling suggests that the distribution of the grey long-eared bat in the UK is mainly limited by low winter temperatures, high summer rainfall, and the availability of grasslands. Suitable environmental conditions do not appear to extend much beyond the current distribution (Razgour et al., 2011b). However, climate-change may alter this situation.

Status

Native.

Conservation Status

- IUCN Red List (GB: EN; England: [EN]; Scotland: n/a; Wales: n/a; Global: LC).
- National Conservation Status (Article 17 overall assessment 2013. Annex IV; UK: Declining; England: Declining; Scotland: n/a; Wales: n/a).

Species' distribution

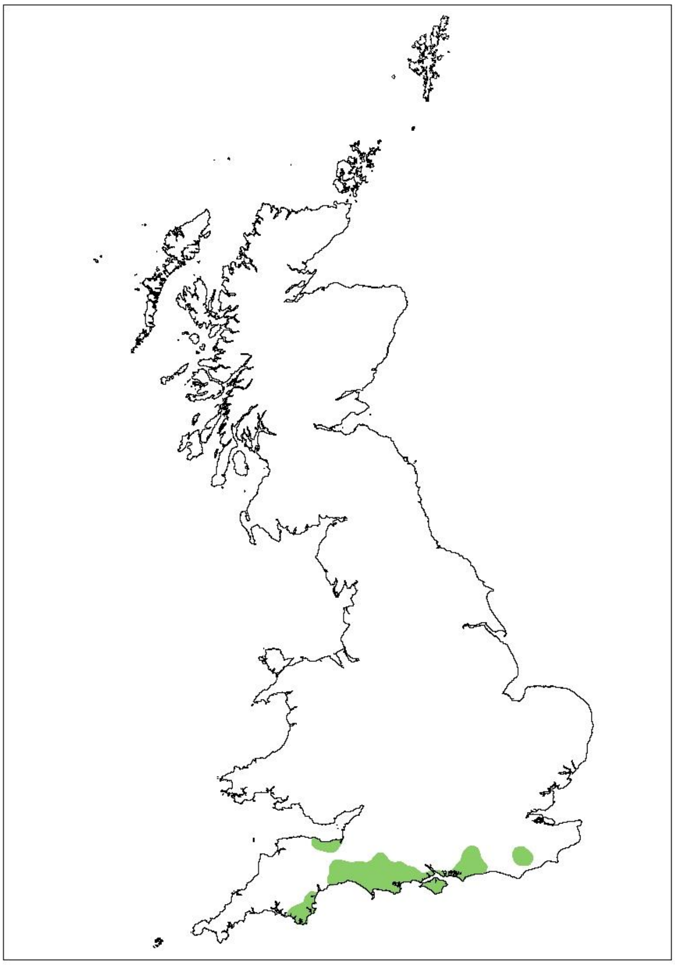


Figure 10.18a Current range of the grey long-eared bat in Britain. Range is based on presence data collected between 1995 and 2016. Areas that contain very isolated records may not have been included in the area of distribution — see Methods, Section 2.5, for more details.

Species-specific methods

There has been relatively little research on this species in the UK. The assessment is, therefore, largely based on the report on the conservation status of the grey long-eared bat in the UK by Razgour et al. (2013), on additional observations made too late for inclusion in that report (Fiona Mathews, *pers. obs.*), and on further molecular surveillance (Barlow and Briggs, 2012).

Maternity roosts in Great Britain typically include 7-34 adults, with the median being approximately 20 (Razgour et al., 2013; George Bemment, *pers. comm.*; Fiona Mathews, *pers. obs.*). This is similar to reports elsewhere in Europe (Dietz and Keifer, 2016).

Expert opinion suggests that maternity colonies are comprised almost entirely of female bats. No data were available in the literature or elsewhere on the sex ratio of the population overall. There are also estimates of roost density. The species appears to have a highly localised distribution, and it is possible that, within these areas, true density is higher than that currently recorded. Some evidence is provided by the identification of occasional grounded bats distant from the nearest known colonies, e.g., in Dorset and east Devon (Sally Humphreys, *pers. comm.*). However, molecular surveillance in these regions has, to date, yielded few additional maternity roost records (Barlow and Briggs, 2012).

Habitable area was, for the purpose of the current review, considered to be all habitats within the geographical range. Because of the landscape-wide movements of bats and their dependency on a matrix of habitats and roosting locations, it is not currently possible to make more refined estimates of the area of suitable habitat within the range.

Results

The values used to derive the density estimates are shown in Table 10.18a.

Table 10.18a Values used to derive bat density estimates.

	Value (plausible intervals)
Roost size	20 (7-34)
Sex ratio	n/a
Maternity roost density	n/a

Population estimation and range

There are thought to be about 10 maternity colonies in Great Britain (8 sites studied by Razgour (2012), which were visits to previously identified maternity colonies, and 2 new colonies identified in Devon). There have also been a small number of additional sites confirmed by molecular analysis (including 2 in east Devon, 1 in south Devon, 1 in north Somerset, and 1 in Pembrokeshire; Barlow & Briggs 2012; Fiona Mathews, *pers. obs.*; Carol Williams, *pers. comm.*). However, it is currently unclear whether these are maternity sites.

Using the median value of 20 adult female bats per roost, this would suggest a pre-breeding female population of 200, or a total population of 400 adult bats. Effective population size, that is, the number of individuals in a population that contribute offspring to the next generation, has been estimated as 184 (95% Credible Intervals = 107-537), based on molecular data from 8 maternity colonies (Razgour, 2012)). Estimations of the effective size of colonies within England varied between a mean of 16 for the Devon colony (95%CI = 15-20), 24 for the two Isle of Wight colonies combined (95%CI = 21-36) and 54 for the Dorset colony (95%CI = 34-180). Effective colony sizes were so low that, except for the Dorset population, it is likely that all colonies run the risk of inbreeding unless gene flow is improved (Razgour et al., 2014). Although the inbreeding risk is not imminent, the extreme isolation means that is a high probability that chance events will send colonies to extinction in the near future owing to the limited opportunities for immigration from surrounding populations.

Plausible estimates for the adult population size could be as high as 1,000-3,000 bats (Razgour, 2012), based on the broad principle that, for mammals generally, effective population sizes are approximately 10 times lower than true population sizes for populations in Hardy-Weinberg equilibrium (Frankham, 2010). However, for this species, most adult females appear to breed in a given year (Fiona Mathews, *pers. obs.*), so there is no evidence of staging of reproduction, and it is unclear whether the population is in equilibrium. In addition, the estimated effective population size takes into account the genetic contribution of migrants: given the genetic connectivity between the bats in England and France, this influence may be quite high. So whilst the molecular data are consistent with a population of around 1,000 individuals, the true number may be much lower, especially as high survey effort has not revealed more colonies (Orly Razgour, *pers. comm.*). The molecular and survey data indicate that local populations are small and highly fragmented, and that the total population is in decline. It is concluded that the conservation status of this species is precarious.

Table 10.18b Area of suitable habitat within the species' range, and total population size estimates with plausible upper and lower intervals for England, Scotland, Wales, and the whole of Britain.

Country	Area within range (km^2)	Bat density (adults/km^2)			Adult population size		
		Estimate	Plausible intervals		Estimate	Plausible intervals	
			Lower	Upper		Lower	Upper
England	7,250	n/a	n/a	n/a	1,000	400	3,000
Scotland	0	n/a	n/a	n/a	0	0	0
Wales	0	n/a	n/a	n/a	0	0	0
Britain	**7,250**	**n/a**	**n/a**	**n/a**	**1,000**	**400**	**3,000**

The Article 17 Report on grey long-eared bat population status 2006-2011 is shown below (Table 10.18c; Joint Nature Conservation Committee (2013b)).

Table 10.18c Article 17 Report on grey long-eared bat population size 2006-11.

Country	Minimum	Maximum
England	n/a	1,000
Scotland	n/a	n/a
Wales	n/a	n/a
Britain	**n/a**	**1,000**

The current distribution estimate for the species is based on known records since 1995, and is shown in Table 10.18d. The recent isolated record from Pembrokeshire is excluded.

Table 10.18d Geographical ranges reported by the current review and the most recent Article 17 Report (Joint Nature Conservation Committee, 2013a).

Country	Extent of occurrence (km^2)	Surface estimate in JNCC Article 17 Report 2007-2012 (km^2)
England	0	n/a
Scotland	0	n/a
Wales	7,250	n/a
Britain	**7,250**	**14,300**

Critique

The estimates are derived from direct counts of all known maternity roosts and population genetic surveillance.

The main potential source of error is under-recording of roosts, particularly given the difficulty of distinguishing the species from the much more common brown long-eared bat. Effort has been put into encouraging molecular identification of droppings from suspected long-eared bat roosts within priority areas identified from habitat suitability modelling (Barlow and Briggs, 2012). With this initiative, one new grey long-eared bat roost was identified from 44 long-eared roosts surveyed. There needs to be greater survey effort deployed at buildings not subject to European Protected Species Licensing.

Table 10.18e Reliability assessment for grey long-eared bats. Scores are based on the availability of roost location data, roost count data, and data on sex ratio. These scores are summed to give a total reliability score.

Measure	Score	Details	Score
Availability of robust roost density estimates*	0	Limited (1 to 3)	0
	1	A few (4 to 6)	
	2	More than 6	
Sample size for roost size estimates	0	<100 roosts	0
	1	<150 roosts	
	2	>200 roosts	
Sex ratio data available	0	No	
	1	Yes	1†
		Overall reliability score	**1**

* Either from literature or expert opinion with high confidence scores.
†Very limited data are available.

Changes through time

Comparison to Harris et al. (1995), Arnold (1993) and JNCC (2013)

The range of the species remains similar to that reported previously.

Harris estimated a pre-breeding population of approximately 1,000 individuals, all in England. However, these estimates were scored as being very subjective, and were based on expert opinion only. The current population estimate also suggests a very low population size, identifying this species as one of the rarest mammals in Great Britain.

Other evidence of changes through time

Most of the sites historically recorded as having grey long-eared bat roosts no longer had any evidence of the species when they were revisited by Razgour (2012).

The range given in this report is smaller than that given in the Article 17 Report (Joint Nature Conservation Committee, 2013b): 7,250km^2 compared with 14,300km^2 . This is likely to reflect methodological differences.

Table 10.18f Trends in population size and geographical range. Population size trends were identified by comparison with Harris et al. (1995). Trends in range size were identified by comparing point maps of current data (not presented) with those from Arnold (1993), unless otherwise stated.

<table>
<tr><td colspan="2"></td><td colspan="4">Range</td></tr>
<tr><td colspan="2"></td><td>Increase</td><td>Stable</td><td>Decrease</td><td>Data deficient</td></tr>
<tr><td rowspan="4">Population size</td><td>Increase</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Stable</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Decrease</td><td></td><td></td><td></td><td></td></tr>
<tr><td>Data deficient</td><td></td><td></td><td>England
Wales</td><td></td></tr>
</table>

* Although the number of records has increased since the period 1960-1992, there have been no recent records in previously positive tetrads on the edge of the species' range, thereby causing an overall decline in range size.

Drivers of change

Table 10.18g Drivers of population change between 1995 and the present. Drivers are limited to those likely to affect the population at a national level.

Driver	Mechanism	Source	Direction of effect
Loss of viable roosts during barn and other building conversions.	Reduction in roost suitability, particularly a reduction in loft area.	Waring et al. (2013)	Negative
Change of habitat, particularly loss of wet and species-rich meadows.	Decline in moth populations.	Conrad et al. (2006a); Conrad et al. (2006b)	Negative
Urban development encroaching on traditional roosts.	Loss of foraging habitat and increased isolation of roosts in the landscape. The species is thought to be poor dispersers owing to wing morphology.	Entwistle et al. (2000)	Negative
Impact of road casualties on local populations.	Collisions with vehicles.	Fensome and Mathews (2016)	Negative
Artificial night lighting.	The species is extremely light-shy. Lighting potentially severs commuting routes and reduces moth availability.	Plummer et al. (2016) and inferences from other light-shy species	Negative

Data deficiencies

Table 10.18h Areas where further research is required to improve the reliability of population size estimates.

Data deficiencies	Habitat	Details
Density of roosts across range.	n/a	Further efforts are needed to identify maternity colonies, particularly in areas where single individuals (e.g., grounded female bats) have been found.
Impacts of loft insulation and the installation of breathable roofing membranes.	n/a	The species appears to have quite defined requirements for maternity roosts. The impacts of changes to lofts on the availability of potentially suitable sites for population expansion is unknown.
Effects of cumulative pressures of land use change, lighting, etc., on local populations, particularly through the fragmentation of habitat, which may restrict access to core foraging areas.	Woodland edge, riparian corridors	No data available (the species is light-sensitive).
Impact of agri-environment schemes on moth abundance and foraging activity by the species.	Farmland	Concern that widely-reported declines in abundance of many moth species will have a negative impact. Although the species is commonly thought of as a woodland bat, much foraging occurs outside woodland, particularly in declining habitats such as wet meadows.
Impacts of insecticides on prey abundance.	Farmland	

Future prospects

Table 10.18i An assessment of the future prospects for the grey long-eared bat, in terms of whether the population size, range and habitat quality are likely to increase, decrease or remain stable. This assessment is based on the current trends, current drivers of change, and potential future drivers of change. For a full assessment of future prospects, see Appendix 7.

Trend	Status
Population	Decline
Range	Unknown
Habitat	Decline

11 Overall research priorities

1. **Distributions were poorly defined for many species. Uncertainty about whether a lack of observer effort or true absence accounted for gaps in the distribution was a recurring problem, particularly towards the peripheries of geographical ranges. Delimiting ranges and understanding the potential impacts of climate change are vital in planning for ecosystem resilience.**

   Current mammal monitoring depends very largely on citizen science initiatives and casual recording. Additional effort needs to be directed to surveying i) towards the edges of known distributions; ii) in areas considered likely to be suitable because of habitat suitability assessments but where the species is not known to be established; and iii) in areas with isolated records that could represent pioneer or remnant populations. Existing citizen science schemes such as the National Bat Monitoring Programme (NBMP) and the National Dormouse Monitoring Programme (NDMP) are not designed to delineate species' distributions; and with the exception of a small number of species that are difficult to misidentify (the badger, fox, hedgehog and rabbit), the data from other established schemes are insufficiently robust for inclusion in this review.

2. **Time trend analyses of both distribution and population size were severely compromised by a lack of systematic monitoring.**

   The establishment of a network of sites that are repeatedly monitored at relevant time intervals (3-5 years) using standardised protocols will address this issue. It is crucial that the peripheries of known distributions are monitored systematically. This has been a recognised objective for many years (e.g., through the Tracking Mammals Partnership), but has been hindered by a lack of resources and/or methodological weaknesses in the methods applied. For some species, particularly those that are cryptic or difficult to observe, genetic estimations of population sizes and trends are likely to prove a much more robust and cost-effective approach to monitoring than count-based techniques.

3. **Occupancy data are lacking for most species and habitats. The assumption that all areas of potentially suitable habitat within the range are occupied may severely overestimate population sizes. This problem is particularly acute for species which are likely to be patchily distributed among suitable habitat within their range, such as the Bechstein's bat and the red deer.**

   This issue should be addressed through widespread presence/absence surveys, which require much less resource than comprehensive monitoring of population size. Effort for each species should focus on those habitats that contribute the greatest proportion of the population in the current estimates.

4. **Estimates of mammal densities are often derived from studies in areas considered likely *a priori* to hold good populations, and are usually in restricted geographical areas rather than in areas of representative habitat quality in each country. Any changes in density with latitude or habitat quality are therefore poorly defined, limiting the ability to plan strategically for the maintenance of ecosystem function and services.**

   This issue is partly a consequence of the fact that density estimation is generally a secondary objective of projects designed to address a different issue (e.g., behavioural ecology or epidemiology). Even where relevant data are collected, there is often a lack of academic interest in publication, so the information remains in project reports and theses that are difficult to access. In addition, some parts of Great Britain — remote areas of Scotland and Wales, for example — are much less studied than others. Stratified randomised sampling, prioritising habitats that contribute most to the current overall population size estimate, provides an efficient and cost-effective means of addressing this difficulty. This network of sites can align with those used in (2). The density data for each species-habitat combination should be stored in an open-access repository.

5. **There has been very little survey effort deployed on abundant species, despite their likely importance to ecosystem services and function: survey effort is instead strongly skewed towards rare animals.**

   This bias has arisen partly as a consequence of protected species legislation and the focus of conservation effort on key species. Brexit, and the departure from the Common Agricultural Policy, provide an opportunity to improve the monitoring of population trends and estimates for other key species. Monitoring should include invasive common species such as the grey squirrel and the brown rat, which are likely to have significant ecological impacts. In addition, several abundant and naturalised species are very poorly quantified. For example, the available evidence for the rabbit suggests that the species is in decline, most notably in Scotland. This decline may be temporary if disease outbreaks are the major driver, but monitoring is required to verify this assumption. Robust population estimates are also lacking for many of the most abundant bats, including the common and soprano pipistrelle. This information is necessary to understand the impact of current threats (such as wind turbines or roost loss), and to design appropriate and proportionate monitoring and mitigation strategies.

6. **Current estimates are crude as they depend on applying a single density estimate to land-cover types (or in the case of bats, regional roost density estimates). It is known that, for many mammals, density and distribution are strongly affected by habitat quality as well as land class. There is evidence that the quality of habitats for wildlife is in decline, even where total availability is constant (e.g., decline of species-rich grassland, or decline of hedgerow quality (Countryside Survey 2007)). Evidence of the impact of such changes is needed for a wide range of species, including common species vital to ecosystem function such as the field vole.**

Effort should be deployed in understanding the associations between habitat quality (including configuration and linkages) and mammal abundance and distribution. Wild mammals make extensive use of marginal habitats within agricultural landscapes, such as hedgerow bottoms and unmanaged field corners; these areas are very poorly estimated by the Land Cover Map. This exercise needs to be aligned with data that can permit extrapolation on a national scale. Examples of suitable datasets include the Countryside Survey and LiDAR. It may also be possible to integrate citizen science mapping or assessment of habitat quality with the surveys described in (2) where species have particular habitat requirements that are not well-captured by remote survey methods (e.g., the availability of tree holes or of habitats free from light pollution, both of which may determine bat presence or abundance).

7. **Some species groups, including many that are of conservation concern and some invasive animals, are notable for the poor quality of data available to determine population size or distribution. Reliability scores were extremely**

poor (score <=1) for all shrews and most bats (16/18 species), and for some species, data were entirely lacking for habitats in which the species is known to occur (for example, brown rats in riparian habitats, or dormice in hedgerows). The following non-bat species (40% of the total) had habitat-specific density and occupancy scores of <1: mole, all shrews, rabbit, edible dormouse, Orkney vole, harvest mouse, black rat, otter (for this species there are excellent occupancy data but information on density is very poor), stoat, weasel, mink, sika deer and Chinese water deer.

Resource needs to be deployed to collect evidence on these species. Several are inherently difficult to study (e.g., the small mustelids), and consideration should be given to the development of alternative monitoring techniques, such as non-invasive genetic sampling.

8. **All bats lack robust density data, with the exception of greater and lesser horseshoe bats. There were insufficient data to permit population size estimation at all for the whiskered, Brandt's and Alcathoe bats (cryptic species); barbastelle bat; Leisler's bat; and the potentially migratory Nathusius' pipistrelle bat. One other bat, the noctule, also had a score of zero for population estimate reliability. For this species, estimates could be computed but they were based on very restricted data, resulting in correspondingly large confidence intervals.**

   Resource needs to be invested in obtaining robust data for these species. Although acoustic techniques can contribute to occupancy data for some species (with the caveat that there is high potential for identification error or under-recording for many groups, such as the *Myotis spp.*, Nyctaloid and long-eared bats), this approach cannot, at present, yield density information. Consideration should be given to genetic approaches to monitor population size trends.

9. **The importance of trees and woodland to bats is extremely poorly understood. Population estimates were impossible for several species particularly associated with woodland. Confidence intervals around estimates for some widespread species, including the noctule, Natterer's and brown long-eared bat, were unacceptably high owing to an almost total reliance on data from buildings to estimate population size. Without information on tree roosts, it is**

not possible to make informed decisions about whether developments are likely to have a material impact on local populations.

There is an urgent need to establish roost densities in woodland and also in other trees (e.g., parkland and mature hedgerow trees). Roost sizes in trees also need to be established for most species. Genetic identification of droppings in rural buildings and those at the rural/suburban interface should be undertaken to improve roost identification for the small *Myotis*.

10. **The sex ratio of pre-breeding roosts is not known for bats. This has a major impact on the population estimates.**

This evidence gap could be rapidly and economically addressed through co-ordinated effort of local bat groups and researchers.

11. **The scale and nature of the impact associated with many potential future threats (e.g., major infrastructure developments; new housing allocations; increased traffic volume; and changes to farming practice in the face of climate-change and altered subsidy scenarios) are extremely poorly characterised, and many of the approaches currently used to monitor them are not suitable for answering these questions. Almost nothing is known about the cumulative effects of such threats, with the loss of foraging habitat, decreased habitat connectivity, and increased light pollution being of particular concern. Most mitigation activities lack a robust evidence base, meaning that resource may be wasted on ineffective actions.**

This information is vital to planning sustainable development in the UK, particularly in the context of the current pressure for new housing and infrastructure. Without it, survey and mitigation methods are unlikely to be either suitable or proportionate. Methods to improve the capture, sharing and standardised interpretation of ecological data are urgently required. The large-scale changes to the agricultural landscape anticipated over the next 20 years are subject to much less legislative control than the changes to the built environment. Given the correspondingly fewer opportunities to take advantage of data collected by industry, there is a need for strategic research, which should include assessment of the effectiveness of new agri-environmental schemes.

Appendix 1: Comparison of habitat classifications

Habitats from Harris et al., 1995 (Table 3) were matched, as closely as possible, to the Land Cover Map 2007 habitat categories and sub-categories for the current review.

LCM2007 broad habitats	LCM2007 sub-habitats	Harris et al.,1995
Broadleaved woodland	**Deciduous**	Semi-natural broadleaf woodland
	Recent (<10yrs)	Broadleaved plantation
	Mixed	Semi-natural mixed woodland Mixed plantation
	Scrub	Tall scrub
Coniferous woodland	**Conifer**	Semi-natural coniferous woodland Coniferous plantation
	Larch	
	Recent (<10yrs)	Young plantation
	Evergreen	
	Felled	Recently felled woodland
Arable and horticulture	**Arable bare**	
	Arable unknown	Arable land
	Arable Orchard	
	Arable barley	
	Arable wheat	
	Arable stubble	
Improved grassland	**Improved grassland** **Ley**	Improved grassland Semi-improved grassland Parkland/amenity grassland
	Hay	
	Neutral grassland	
	Calcareous grassland	
	Acid grassland	
Rough grassland (here considered equivalent to unimproved grassland)	**Rough / unmanaged grassland**	Upland unimproved grassland Lowland unimproved grassland
Fen, marsh and swamp	**Fen / swamp**	

LCM2007 broad habitats	LCM2007 sub-habitats	Harris et al., 1995
Dwarf shrub heath	**Heather & dwarf shrub**	Heather moorland Lowland heaths Low scrub Bracken
	Burnt heather	
	Gorse heather	
	Dry heath	
	Heather grass	
Bog	**Bog**	Raised bog
	Blanket bog	Blanket bog
	Bog (Grass dominated)	
	Bog (Heather dominated)	
Montane habitats	**Montane habitats**	
Inland rock	**Inland rock**	Unquarried inland cliffs
	Land	
Salt water	**Sea**	
	Estuary	
Freshwater	**Flooded**	Standing man-made water
	Lake	Standing natural water
	River	Running natural water Running canalised water
Supra-littoral rock	**Supra littoral rocks**	
Supra-littoral sediment	**Sand dune**	Coastal sand dunes
	Sand dune with shrubs	
	Shingle	Coastal shingle or boulder beaches
	Shingle vegetated	
Littoral rock	**Littoral rock**	
	Littoral rock / algae	
Littoral sediment	**Littoral mud**	
	Littoral mud / algae	
	Littoral sand	
Saltmarsh	**Saltmarsh**	Coastal marsh Coastal sand or mud flats
	Saltmarsh grazing	

LCM2007 broad habitats	LCM2007 sub-habitats	Harris et al. 1995
Built up areas and gardens	**Bare**	Bare ground
	Urban	Built land
	Industrial	
	Suburban	
Hedgerows	**Hedgerows**	Hedgerows
Treelines	**Treelines**	Treelines
No match	**No match**	Ditches and drains Marginal Inundation Wet Ground Vertical Coastal Cliffs Sloping Coastal Cliffs

Appendix 2: Extent of occurrence.

Total area (km^2) (including unsuitable habitat) within range based on alpha hull approach.

Genus	Species	England	Scotland	Wales	Britain
Erinaceomorpha	**Hedgehog**	129,914	73,279	20,643	223,836
Soricomorpha	**Mole**	129,901	69,705	20,643	220,249
	Common shrew	127,995	52,938	19,424	200,358
	Pygmy shrew	118,980	24,563	18,708	162,251
	Water shrew	117,78316	25,8330	17,5300	161,14616
	Lesser white-toothed shrew	16	0	0	16
Lagomorpha	**European rabbit**	129,916	75,612	20,643	226,172
	Brown hare	129,439	55,012	20,633	205,083
	Mountain hare	2,423	57,411	0	59,834
Rodentia	**Red squirrel**	18,449	55,060	3,192	76,701
	Grey squirrel	129,135	33,831	19,658	182,623
	Beaver	244	5,016	0	5,261
	Hazel dormouse	67,601	0	14,677	82,277
	Edible dormouse	2,368	0	0	2,368
	Bank vole	125,389	32,206	20,037	177,632
	Field vole	128,942	63,098	18,996	211,036
	Orkney vole	0	706	0	706
	Water vole	109,996	43,930	14,512	168,437
	Harvest mouse	101,637	0	5,042	106,680
	Wood mouse	127,593	55,946	20,051	203,590
	Yellow-necked mouse	55,974	0	6,795	62,769
	House mouse	105,477	12,806	9,146	127,429
	Brown rat	127,511	36,835	18,653	183,000
	Black rat	DD	DD	DD	DD

Genus	Species	England	Scotland	Wales	Britain
Carnivora	**Wildcat**	0	26,700	0	26,700
	Fox	129,901	69,721	20,643	220,265
	Badger	129,901	64,552	20,643	215,096
	Otter	125,672	76,479	20,643	222,794
	Pine marten	12,358	61,049	9,544	82,952
	Stoat	128,226	56,350	16,416	200,992
	Weasel	129,390	54,012	19,563	202,965
	Polecat	85,377	n/a	20,552	105,929
	Mink	128,900	51,308	20,411	200,619
Artiodactyla	**Wild boar**	6,889	1,149	309	8,347
	Red deer	97,559	62,966	8,956	169,481
	Sika deer	26,183	41,366	1,398	68,947
	Fallow deer	114,602	14,291	18,479	147,371
	Roe deer	128,604	70,294	16,804	215,701
	Chinese water deer	18,152	0	0	18,152
	Muntjac deer	111,130	1,530	11,382	124,042
Chiroptera	**Greater horseshoe bat**	29,567	0	13,230	42,797
	Lesser horseshoe bat	33,552	0	19,549	53,101
	Alcathoe bat	5,040	0	0	5,040
	Bechstein's bat	23,344	0	155	23,499
	Brandt's bat*	109,201	2,012	20,488	131,700
	Whiskered bat*	109,201	2,012	20,488	131,700
	Daubenton's bat	129,146	44,417	20,377	193,941
	Greater mouse-eared bat	DD	0	0	DD
	Natterer's bat	126,502	16,172	20,611	163,286

Species	England	Scotland	Wales	Britain
Serotine bat	78,082	0	12,499	90,580
Leisler's bat	68,353	4,978	6,739	80,070
Noctule bat	126,913	9,485	20,627	157,025
Common pipistrelle bat	129,914	60,792	20,601	211,307
Soprano pipistrelle bat	128,458	52,223	20,643	201,324
Nathusius' pipistrelle bat	70,285	4,214	6,921	81,421
Barbastelle bat	67,610	0	6,386	73,996
Brown long-eared bat	129,683	49,139	20,643	199,464
Grey long-eared bat	7,247	0	0	7,247

* Geographical range calculated jointly for the whiskered and Brandt's bat. DD Data deficient

Appendix 3: Population size estimates, reliability scores and 25-year trends.

Genus	Species	Country	Population size	-95%CI	+95%CI	Population	Range	Reliability score
Erinaceomorpha	Hedgehog	England	[597,000]	n/a	n/a	Decrease	Stable	2
		Scotland	[196,000]	n/a	n/a			
		Wales	[86,800]	n/a	n/a			
		Britain	[879,000]	n/a	n/a			
Soricomorpha	Mole	England	[24,300,000]	n/a	n/a	Unknown	Stable	1
		Scotland	[12,200,000]	n/a	n/a			
		Wales	[4,900,000]	n/a	n/a			
		Britain	[41,400,000]	n/a	n/a			
	Common shrew	England	[11,000,000]	3,520,000	29,500,000	Unknown	Stable[1]	1
		Scotland	[7,690,000]	1,980,000	22,900,000			
		Wales	[2,330,000]	1,010,000	6,120,000			
		Britain	[21,100,000]	6,520,000	58,500,000			
	Pygmy shrew	England	[3,690,000]	552,000	27,900,000	Unknown	Stable[1]	0.5
		Scotland	[1,430,000]	217,000	6,040,000			
		Wales	[1,170,000]	231,000	4,970,000			
		Britain	[6,300,000]	999,000	38,900,000			
	Water shrew	England	[458,000]	147,000	1,228,000	Unknown	Stable[2]	0
		Scotland	[118,000]	30,000	353,000			
		Wales	[137,000]	60,000	361,000			
		Britain	[714,000]	237,000	1,942,000			
	Lesser white toothed shrew	England	[14,000]	n/a	n/a	Stable	Stable	0
		Britain	[14,000]	n/a	n/a			
Lagomorpha	European rabbit	England	[21,300,000]	n/a	n/a	Decrease	Stable	1
		Scotland	[11,800,000]	n/a	n/a			
		Wales	[2,910,000]	n/a	n/a			
		Britain	[36,000,000]	n/a	n/a			
	Brown hare	England	454,000	336,000	1,480,000	Unknown	Stable	3
		Scotland	87,700	64,000	342,000			
		Wales	37,000	27,000	171,000			
		Britain	579,000	427,000	1,990,000			
	Mountain hare	England	2,500	1,500	9,500	Unknown	Increase[3]	2
		Scotland	132,000	79,500	516,000			
		Britain	135,000	81,000	526,000			
Rodentia	Red squirrel	England	38,900	29,500	91,000	Decrease	Decrease[4]	2
		Scotland	239,000	181,000	444,000			
		Wales	9,200	7,000	18,200			
		Britain	287,000	218,000	553,000			
	Grey squirrel	England	1,940,000	957,000	2,560,000	Increase	Stable[5]	1.7
		Scotland	478,000	249,000	808,000			
		Wales	283,000	139,000	423,000			
		Britain	2,700,000	1,340,000	3,790,000			

Genus	Species	Country	Population size	-95%CI	+95%CI	Population	Range	Reliability score
Rodentia	Beaver	England	10	n/a	n/a	Increase	Increase	n/a
		Scotland	158	n/a	n/a			
		Britain	168	n/a	n/a			
	Hazel dormouse	England	757,000	298,000	2,110,000	Decline	Stable	2
		Scotland	0	0	0			
		Wales	172,000	90,700	529,000			
		Britain	930,000	389,000	2,640,000			
	Edible dormouse	England	[23,000]	9,800	82,000	Unknown	Increase	1
		Britain	[23,000]	9,800	82,000			
	Bank vole	England	19,100,000	10,400,000	35,600,000	Unknown	Stable[1]	1.7
		Scotland	5,390,000	3,130,000	11,900,000			
		Wales	2,930,000	1,560,000	6,560,000			
		Britain	27,400,000	15,100,000	54,100,000			
	Field vole	England	28,600,000	16,900,000	44,000,000	Unknown	Stable	2
		Scotland	21,500,000	13,600,000	24,500,000			
		Wales	9,760,000	6,430,000	11,800,000			
		Britain	59,900,000	37,000,000	80,300,000			
	Orkney vole	Scotland	n/a	n/a	n/a	Decrease[1]	Stable	0
		Britain	n/a	n/a	n/a			
	Water vole	England	77,000	58,000	193,000	Decline	Stable[5]	3
		Scotland	50,000	38,000	125,000			
		Wales	4,500	3,400	11,300			
		Britain	132,000	99,000	329,000			
	Harvest mouse*	England	[532,000]	[272,000]	[879,000]	Unknown	Unknown	0
		Wales	[34,000]	[17,000]	[56,000]			
		Britain	[566,000]	[288,000]	[934,000]			
	Wood mouse	England	22,700,000	11,600,000	37,800,000	Stable	Stable	2
		Scotland	12,300,000	6,510,000	18,800,000			
		Wales	4,600,000	2,240,000	7,680,000			
		Britain	39,600,000	20,400,000	64,300,000			
	Yellow necked mouse	England	1,360,000	426,000	3,940,000	Unknown	Increase	2.5
		Wales	140,000	40,600	423,000			
		Britain	1,500,000	467,000	4,360,000			
	House mouse	England	[4,340,000]	n/a	n/a	Stable	Stable[1]	2
		Scotland	[523,900]	n/a	n/a			
		Wales	[339,000]	n/a	n/a			
		Britain	[5,203,000]	n/a	n/a			
	Brown rat	England	[4,730,000]	n/a	n/a	Unknown	Stable[1]	1
		Scotland	[1,060,000]	n/a	n/a			
		Wales	[1,280,000]	n/a	n/a			
		Britain	[7,070,000]	n/a	n/a			

Genus	Species	Country	Population size	-95%CI	+95%CI	Population	Range	Reliability score
Rodentia	Black rat	Britain	n/a	n/a	n/a	Decrease	Decrease	n/a
Carnivora	Wildcat	Scotland	200	30	430	Decrease	Decrease	2
		Britain	200	30	430			
	Red fox	England	255,000	65,200	464,000	Unknown	Stable[5]	2.5
		Scotland	74,000	30,100	132,000			
		Wales	27,700	9,260	50,000			
		Britain	357,000	104,000	646,000			
	Badger	England	384,000	259,000	711,000	Increase	Stable	4
		Scotland	115,000	85,000	198,000			
		Wales	47,000	47,000	104,000			
		Britain	562,000	391,000	1,014,000			
	Otter	England	[2,900]	n/a	n/a	Increase	Increase	1
		Scotland	[7,100]	n/a	n/a			
		Wales	[1,000]	n/a	n/a			
		Britain	[11,000]	n/a	n/a			
	Pine marten	Scotland	3,700	1,600	8,900	Increase	Increase	2
		Wales	39	n/a	n/a			
		Britain	3,700	1,600	8,900			
	Stoat	England	[260,000]	n/a	n/a	Unknown	Stable	1
		Scotland	[140,000]	n/a	n/a			
		Wales	[37,600]	n/a	n/a			
		Britain	[438,000]	n/a	n/a			
	Weasel	England	[308,000]	n/a	n/a	Unknown	Stable[1]	0
		Scotland	[106,000]	n/a	n/a			
		Wales	[36,000]	n/a	n/a			
		Britain	[450,000]	n/a	n/a			
	Polecat	England	66,000	54,000	79,000	Increase	Increase[6]	4
		Wales	17,000	14,000	20,000			
		Britain	83,000	68,000	99,000			
	Mink	England	[62,400]	n/a	n/a	Decrease	Increase[2]	1
		Scotland	[46,600]	n/a	n/a			
		Wales	[12,900]	n/a	n/a			
		Britain	[122,000]	n/a	n/a			
Artiodactyla	Wild boar	England	500	30	1,500	Increase	Increase	2
		Scotland	2,000	100	6,500			
		Wales	150	<10	500			
		Britain	2,600	200	8,400			
	Red deer	England	80,000	31,000	124,000	Increase	Increase	4
		Scotland	256,000	176,000	376,000			
		Wales	10,000	4,000	16,000			
		Britain	346,000	212,000	516,000			

Genus	Species	Country	Population size	-95%CI	+95%CI	Population	Range	Reliability score
Artiodactyla	Sika deer	England	[45,000]	8,000	107,000	Increase	Increase	0.5
		Scotland	[54,000]	17,900	149,000			
		Wales	[3,600]	900	9,300			
		Britain	[103,000]	27,000	266,000			
	Fallow deer	England	188,000	138,000	245,000	Increase	Increase	3
		Scotland	57,00	42,000	74,000			
		Wales	19,000	14,000	24,800			
		Britain	264,000	194,000	343,000			
	Roe deer	England	120,000	97,900	135,000	Increase[4]	Increase[4]	4
		Scotland	122,000	99,000	136,000			
		Wales	22,000	18,000	25,000			
		Britain	265,000	215,000	296,000			
	Chinese water deer	England	[3,600]	200	143,000	Increase	Increase	1
		Britain	[3,600]	200	143,000			
	Muntjac deer	England	112,000	100,000	128,000	Increase	Increase	4
		Scotland	16,000	15,000	19,000			
		Britain	128,000	115,000	147,000			
Chiroptera	Greater horseshoe bat	England	10,200	7,300	14,600	Increase	Increase	4
		Scotland	0	0	0			
		Wales	2,700	1,930	3,850			
		Britain	12,900	9,200	18,500			
	Lesser horseshoe bat	England	19,600	13,900	27,700	Increase	Increase	3
		Scotland	0	0	0			
		Wales	30,700	22,700	45,300			
		Britain	50,300	36,600	73,000			
	Alcathoe bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Whiskered bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Brandt's bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Bechstein's bat	England	21,600	10,200	55,000	Unknown	Unknown	2
		Scotland	0	0	0			
		Wales	250	120	630			
		Britain	21,800	10,300	55,600			
	Daubenton's bat	England	[682,000]	18,100	2,950,000	Unknown	Stable	1
		Scotland	[235,000]	6,220	1,020,000			
		Wales	[108,000]	2,860	466,000			
		Britain	[1,030,000]	27,000	4,440,000			
	Greater mouse-eared bat	Britain	n/a	n/a	n/a	Stable	Stable	n/a

Genus	Species	Country	Population size	-95%CI	+95%CI	Population	Range	Reliability score
Chiroptera	Natterer's bat	England	[321,000]	11,700	2,040,000	Unknown	Unknown	2
		Scotland	[41,000]	1,500	260,000			
		Wales	[52,300]	1,900	332,000			
		Britain	[414,000]	15,100	2,630,000			
	Serotine bat	England	117,000	6,300	356,000	Unknown	Unknown	3
		Scotland	0	0	0			
		Wales	18,700	1,000	57,000			
		Britain	136,000	7,300	413,000			
	Leisler's bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Noctule bat	England	[565,000]	17,700	1,872,000	Unknown	Unknown	0
		Scotland	[not published]	not published	not published			
		Wales	[91,900]	2,900	304,000			
		Britain	n/a	n/a	n/a			
	Common pipistrelle bat	England	1,870,000	609,000	4,620,000	Unknown	Unknown	2
		Scotland	875,000	285,000	2,160,000			
		Wales	297,000	96,600	732,000			
		Britain	3,040,000	991,000	7,510,000			
	Soprano pipistrelle bat	England	2,980,000	1,260,000	5,360,000	Unknown	Unknown	2
		Scotland	1,210,000	512,000	2,180,000			
		Wales	478,000	202,000	862,000			
		Britain	4,670,000	1,970,000	8,400,000			
	Nathusius' pipistrelle bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Barbastelle bat	Britain	n/a	n/a	n/a	Unknown	Unknown	0
	Brown long-eared bat	England	607,000	34,000	1,430,000	Unknown	Unknown	2
		Scotland	230,000	13,000	543,000			
		Wales	97,000	5,400	228,000			
		Britain	934,000	52,000	2,200,000			
	Grey long-eared bat	England	[1,000]	400	3,000	Unknown	Decrease	1
		Britain	[1,000]	400	3,000			

[1] Scotland decline (possible artefact of recording effort); [2] England and Wales increase (possible artefact of recording effort); [3] England decrease; [4] Scotland stable; [5]Scotland increase (possible artefact of recording effort); [6] Wales stable. * Geographical range calculated jointly for the whiskered and Brandt's bat. DD: data deficient.

Reliability scores were not produced where there was no population size estimate; where population size was based on a total count, or where other necessary evidence was unavailable. Values are shown in square brackets where the reliability score is <=1; where the upper confidence limit is more than 5 times larger than the central estimate; or where it was not possible to compute confidence intervals (except for beaver where total counts are assumed to include most of the population).

Appendix 4: Expert opinion questionnaire

Example of the questionnaires sent to experts during consultation to gain expert opinion on roost size and density (bats) and population density (all other mammals). Surveys were sent via email in most cases, or in conducted in person where appropriate.

Questionnaire – Bats

1	**What is your name?**	

2 **We could not find any estimates of occupied PRE-BREEDING maternity roost size in the UK for Leislers bats.**

If you have experience of working with this species which would enable you to provide an estimate of maternity roost size, please provide this below:
We define 'pre-breeding' as the period after the maternity roost has formed, but before the young are born.

Data source	Pre-breeding maternity roost size	Typical range		No. roost counts (N)
		lower	upper	
NBMP data	101	34	169	2
Greenaway and Hutson (1990)	...	20	50	...
Your estimate				

Please note:
* The typical range 'lower' and 'upper' columns refer to the likely pre-breeding maternity roost size in poor and ideal quality habitat, respectively

* We have set a minimum roost size NBMP and licence return data at 20 to ensure that the roosts surveyed likely to be maternity roosts. Only data collected before the end of June were used to increase the probability that young were not volant and estimates represent adults only. Mean roost size was estimated using the peak emergence count for each roost between 1995 and 2016.

* The estimates provided do not account for adults not observed during emergence counts, however we would expect this number to be relatively low before breeding.

* If you do not feel able to suggest an estimate, please write 'no comment'

Reference: Greenaway, F., & Hutson, A. M. (1990). A Field Guide to British Bats: Bruce Coleman Books.

2b	**How did you arrive at the figures above?** *E.g. Based in unpublished estimates / field experience - please give as much detail as possible.*	

3	**What percentage of adults in a maternity roost is likely to be female for this species?**	%	*Comments...*

4	**What percentage of females, in a typical year, will roost away from the main maternity roost on any given night at the START of the breeding season?** (i.e. BEFORE the young are born)	**%**	***Comments…***

5	**Do you have evidence that the overall ratio of males: females in the population (not just maternity roost) differs markedly from 1:1?**	

6	**We would like to identify the areas that could help improve bat population estimates. Please list the environmental or demographic variables, or other types of information, that you would consider most useful to collect alongside count data in future. Examples might include surveys of potential tree roost availability in target regions.**	

7	**Reasonable estimates of roost density and colony size may not be possible for all species. Identifying which species are data-poor is an important part of this project. Please indicate, on a scale of 1 to 10, the confidence you have in the information relevant to population size possessed by the community of British bat workers and researchers for this species (1=no confidence at all; 10=high confidence).**	**Maternity roost density =**	***Comments…***
		Colony size =	***Comments…***

8 How many years of experience do you have of carrying out surveys / analysing data on this species? *Please tick the appropriate box*	
One to two	
Three to five	
More than five	

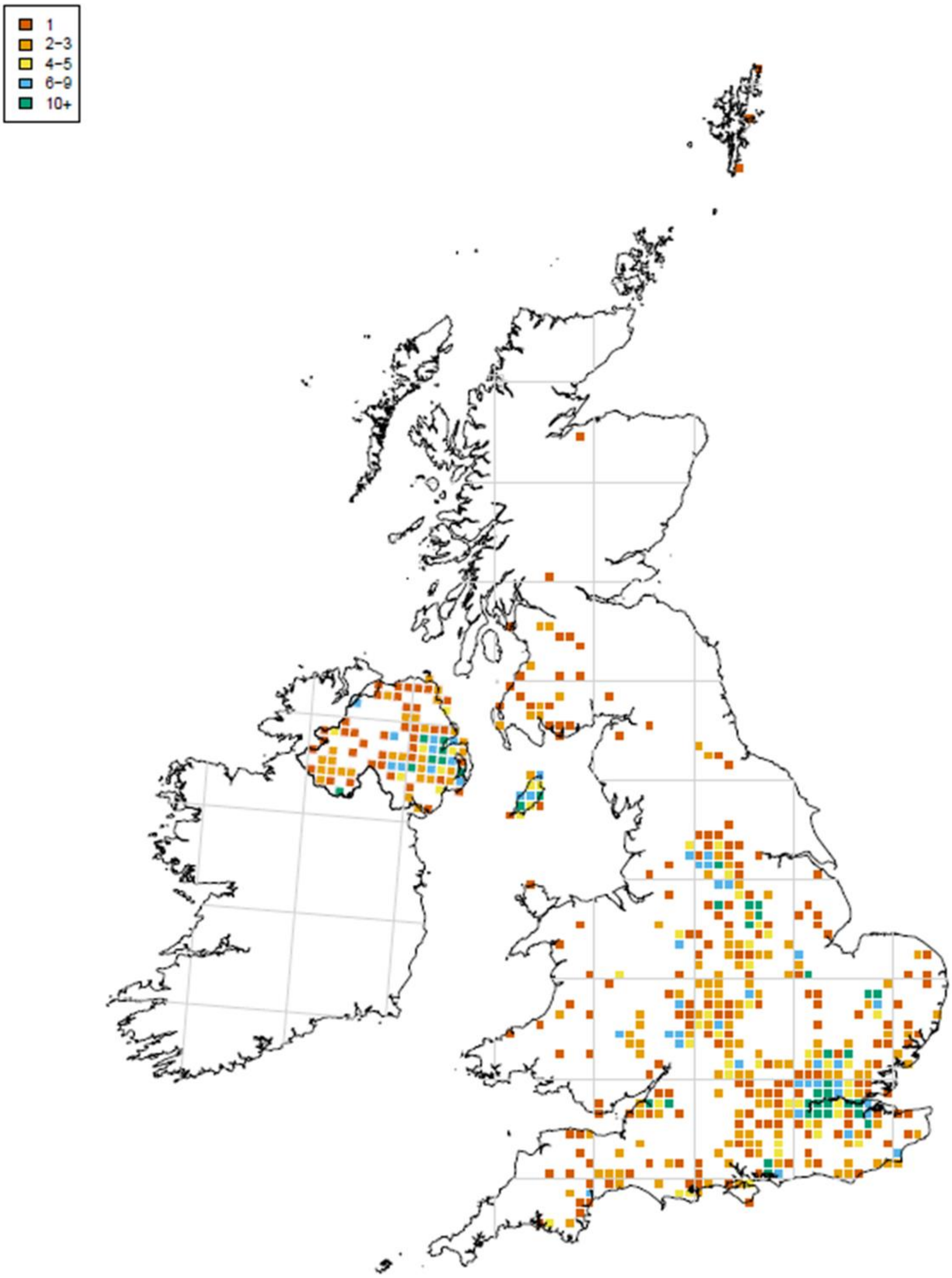
9 How frequently do you carry out surveys / analyse data on this species? *Please tick the appropriate box*	
Less than one per season	
One or two per season	
Several per season	

10 Please now review the DISTRIBUTION map below. The map was produced using records from LRCs, the NBN gateway and a range of other surveys. Each 10km square has been highlighted if it contains one or more records (see the map key). To review the map, please follow the instructions below:

a) Please cross out any squares that you think incorrectly show the species to be present

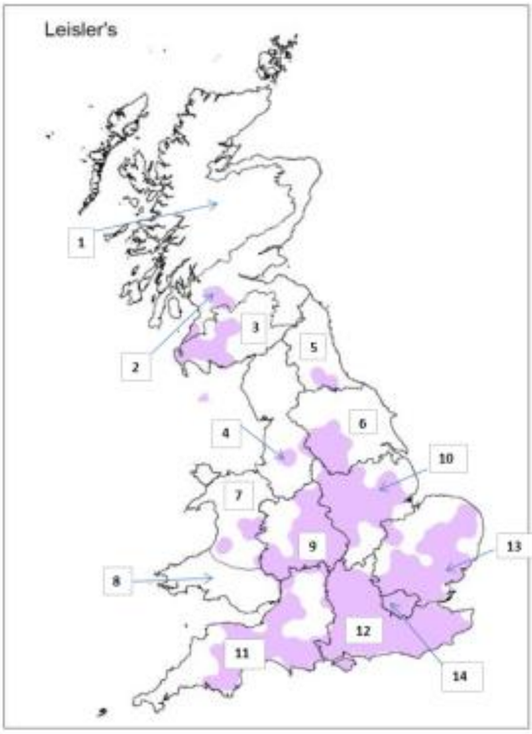
b) To ADD to the species distribution, drag the SQUARES to areas that you think should be included.

Please note - the finalised maps will be smoothed and any blank areas which are surrounded by records will be filled in. You do not, therefore, need to fill in the gaps of the current distribution. Only add new squares where they would expand the current range.


1
2-3
4-5
6-9
10+

11. Please view the maternity roost REGIONAL DENSITY map to the right. The purple area represents the known distribution for this species.

Based on your experience, please suggest the most likely number of maternity roosts for a typical 10x10km square (100km2), a square with poor habitat quality (lower density) and a square with high habitat quality (upper density) within each region on the map, following the guidance below:



a) We wish to establish plausible estimates for the **entire geographical region**. The 'Roost density' relates to a typical habitat within the region. The 'lower' and 'upper' values are for poorer and higher quality habitat but exclude very extreme values.

b) The estimates will be applied to the **known distribution only** (purple areas on the map). Therefore even in poorer quality habitats, the roost density estimates will usually be greater than zero due to the occasional availability of suitable features.

c) If you wish to provide an **estimate for a smaller area**, i.e. an area that you regularly survey, please do so in the row for the relevant region and describe your area in the comments box.

No.	Region	Roost density (per 100km2)	Lower density	Upper density	Comments
1	Scottish Highlands & Islands				
2	Central lowlands & Eastern				
3	Southern Scotland				
4	North West England				
5	North East England				
6	Yorkshire and Humber				
7	North Wales				
8	South Wales				
9	West Midlands				
10	East Midlands				
11	Eastern England				
12	South West England				
13	South East England				
14	London				

Thank you for taking the time to complete our survey.

Questionnaire – All other mammals

What is your name?	

The table below shows the mean pre-breeding population densities for red deer, estimated by meta-analysis from peer-reviewed literature (1995 - 2015).

We would like you to review these estimates, following the instructions below:

Habitat	Pre-breeding estimate (km2)	Typical range (km2)		Pre-breeding estimate (km2)	Typical range (km2)		Reasons for changes to estimates *e.g. Based on unpublished studies, field experience etc.*
		Lower range	Upper range		Lower range	Upper range	
Arable and horticulture							
Improved grassland							
Unimproved grassland							
Broadleaved woodland	10.9	9.3	12.5				
Urban areas / gardens							
Coniferous woodland							
Dwarf shrub heath	8.4	7.1	9.6				
Bog							
Fen marsh and swamp							
Montane habitats							
Sand dunes							
Riparian habitats	(per km)	(per km)	(per km)	(per km)	(per km)	(per km)	
Coastal	(per km)	(per km)	(per km)	(per km)	(per km)	(per km)	
Hedgerows	(per km)	(per km)	(per km)	(per km)	(per km)	(per km)	

a) If you think an estimate is incorrect, please enter an alternative in the boxes below, with a reason for your change.
b) If you agree with our estimate or have no opinion, please leave the box blank.
c) For unsuitable habitats, please enter a zero.
d) If we have not provided an estimate for a habitat you deem suitable, please enter your best estimate, with justification.
e) Please make sure your figures are per square km (or per km for linear features)

Please note: the typical range values roughly equate to 95% confidence intervals. We only require you to make suggests based on your experience, however, and do not expect any calculations.

Over the last 20 years, do you think the national population size of red deer has changed? Please specify a percentage or range of percentages.		
	Increase	%
	No change	%
	Decrease	%
	No opinion	%

4	**If you have indicated a change, why do you think this change has occurred?**	

5	**Which methods you have used to estimate the population density of red deer?** e.g. trapping/distance surveys etc.	Minimum number alive counts	
		Total counts	
		Capture-mark-recapture	
		Distance sampling	
		None - I have conducted presence surveys only	

6	**How many years experience do you have of carrying out surveys / analysing data on red deer?**	One to two	
		Three to five	
		More than five	

7	**How frequently did you carry out surveys / analyse data on red deer?**	Less than one per season	
		One or two per season	
		Several per season	

10 **We have attached a distribution map to your email. The map was produced using records from LRCs, the NBN gateway and other national mammal surveys. Each 10km square has been highlighted if it contains one or more records (see the map key). To review the map, please follow the instructions below:**

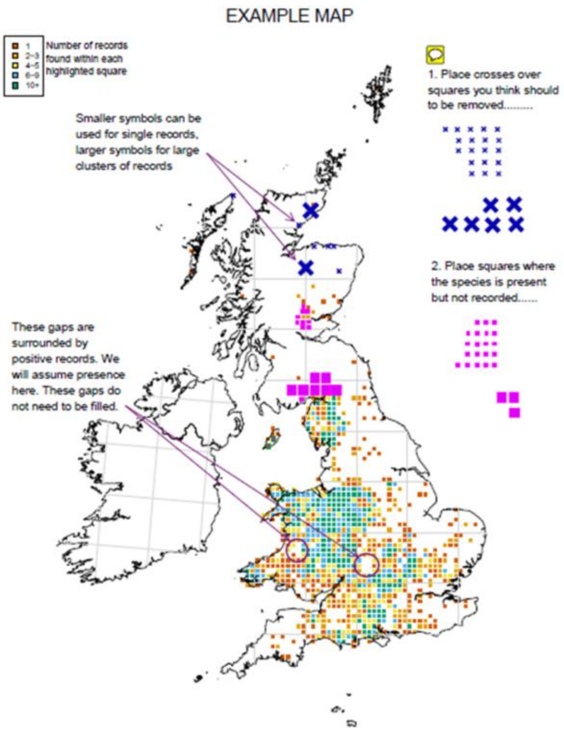
a) To EXCLUDE areas that you think incorrectly show the species to be present, drag CROSSES over any squares (or groups of squares) that you wish to exclude.

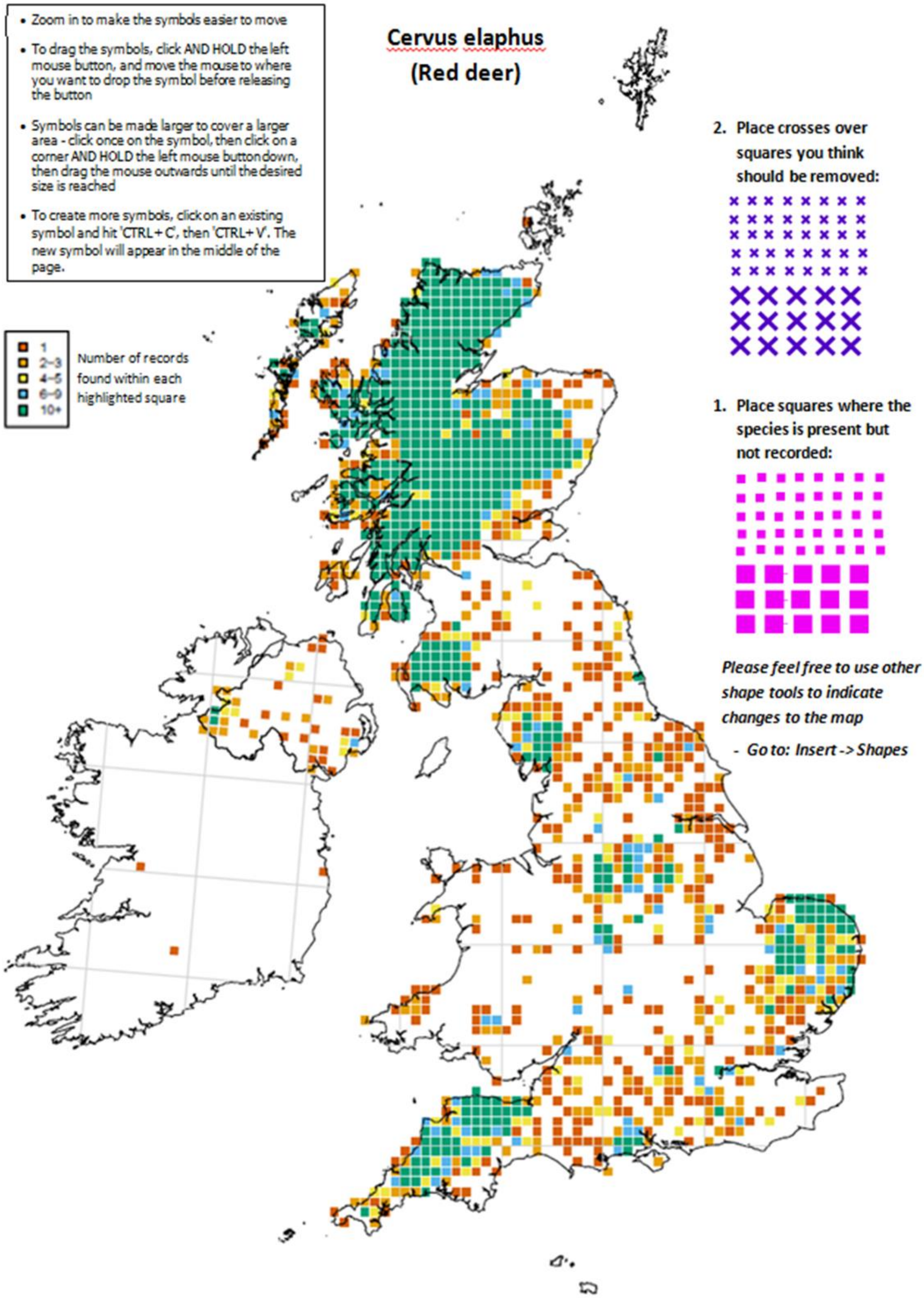
b) To ADD to the species distribution, drag the SQUARES to areas that you think should be included.

c) Feel free to use other symbols / draw on the map in any way to indicate your desired changes. Go to 'Insert - Symbol' for other drawing options.

Please note - the finalised maps will be smoothed and any blank areas which are surrounded by records will be filled in. You do not, therefore, need to fill in the gaps of the current distribution. Only add new squares where they would expand the current range.

For further guidance, see the example map opposite -->





Zoom in to make the symbols easier to move
To drag the symbols, click AND HOLD the left mouse button, and move the mouse to where you want to drop the symbol before releasing the button
Symbols can be made larger to cover a larger area - click once on the symbol, then click on a corner AND HOLD the left mouse button down, then drag the mouse outwards until the desired size is reached
To create more symbols, click on an existing symbol and hit 'CTRL+C', then 'CTRL+V'. The new symbol will appear in the middle of the page.
Cervus elaphus
(Red deer)
1
2–3
4–5
6–9
10+
Number of records found within each highlighted square
2. Place crosses over squares you think should be removed:
1. Place squares where the species is present but not recorded:
Please feel free to use other shape tools to indicate changes to the map
- Go to: Insert -> Shapes

Appendix 5: Data deficiencies for each species considered in the population review

Habitat-specific density data were not available for most bat species and so population sizes had to be computed differently. Horseshoe bats are shown separately from other Chiroptera because different methodologies were used.

Species	Density estimates do not represent within-habitat variability	Density estimates more than 10 years old	Limited density estimates for key habitat	Managed popns	Multiannual population cycles	No density estimates for specified habitat	No occupancy data	Population sizes based on total counts rather than density estimates
Hedgehog	X	X				X		
Mole	X	X					X	
Common shrew	X	X				X	X	
Pygmy shrew	X	X	X			X	X	
Water shrew*						X	X	
Lesser white-toothed shrew	X	X	X			X	X	
European rabbit	X	X	X				X	
Brown hare	(X)	X		X				
Mountain hare	(X)				X	X	X	
Red squirrel	(X)	X	X				X	
Grey squirrel	(X)	X	X	X			X	
Beaver								X
Hazel dormouse			X			X	X	
Edible dormouse						X	X	
Bank vole						X	X	
Field vole					X	X	X	
Orkney vole	(X)		X			X	X	
Water vole	(X)	X						
Harvest mouse			X			X	X	
Wood mouse	X	X				X	X	
Yellow-necked mouse		X	X			X		
House mouse	X	X						
Brown rat	X	X				X		
Black rat	X	X	X	X		X		
Wildcat	X		X					
Fox	[X]						X	
Badger	X		X	X		X	X	X
Otter	X	X	X					
Pine marten						X	X	
Stoat	X	X		X			X	
Weasel*						X	X	
Polecat	[X]	X						
Mink	X		X					

Species	Density estimates do not represent within-habitat variability	Density estimates more than 10 years old	Limited density estimates for key habitat	Manage d popns	Multiannual population cycles	No density estimates for specified habitat	No occup ancy data	Population sizes based on total counts rather than density estimates
Wild boar				X			X	
Red deer			X	X		X	X	
Sika deer	X	X	X	X		x	X	
Fallow deer				X				
Roe deer				X				
Chinese water deer						X	X	
Muntjac deer				X		X		
Horseshoe bats	X	X	X			X	X	X
Other bats	X	X	X			X	X	

*Population estimate calculations relied on ratios with other similar species and, therefore, were not a direct estimate of density.

(X) Despite ranges given and the presence of several density estimates, the species biology and expert opinion suggests that the available values do not fully represent the species density variability. See individual species data deficiency accounts for details.

[X] Population estimates were calculated across habitats combined and it was not possible to ascertain variability between different habitats.

Appendix 6: Species not included in the main review

The following species occur as in Britain only as vagrants, feral animals, island populations, occasional individuals, or managed populations.

Reindeer *Rangifer tarandus*

Status

Naturalised (native in prehistoric times). Managed.

Conservation Status

- IUCN Red List Global: VU

The reindeer is thought to have been present in Britain until approximately 8,000 years ago (Yalden, 1999). Despite a reference to reindeer in a 12th Century Nordic text (*Orkneyinga saga*) , there is no evidence that the species was present in medieval times (Clutton-Brock and MacGregor, 1988). Reindeer were re-introduced to Great Britain in 1952. There is one population in the northern Cairngorms in Scotland, and an additional population, that was established later, near Tomintoul. The herds are free-ranging but are closely managed, with population sizes being maintained at approximately 140-150 individuals. Reindeer use upland heather moorland, and feed primarily on heather, dwarf shrubs, sedges, grasses and lichens. The Scottish populations also receive supplementary food (see Harris and Yalden, 2008).

Feral ferret *Mustela furo*

Status

Non-native

Conservation Status

- IUCN Red List Global: not listed

The ferret is a domesticated form of the polecat *Mustela putorius*. Feral animals include those that have recently been released from captivity, as well as those from more established feral populations (Vincent Wildlife Trust, 2014). The ferret has similar habitat requirements to the polecat, although they are more likely to be found in urban areas (Harris and Yalden, 2008).

The polecat and ferret can interbreed to produce fertile offspring. This presents a hybridisation threat to true polecats in Britain. During a recent survey throughout Britain, including road kill, live sightings, live trapped animals, and camera trap records, 25% of samples (n=187) were classified as polecat-ferret hybrids and 1% (n=10) as ferrets (Croose, 2016). Ferrets are widely kept throughout Britain, so feral ferrets are likely to have a broad geographical range. Records of hybrids are scattered throughout England, although most records are found on the periphery of the true polecat's range. In Wales, very few hybrids were found during 2014-15 Polecat Survey (Vincent Wildlife Trust) whereas they were common in Scotland.

Feral sheep *Ovis aries*

Status

Native

Conservation Status

- IUCN Red List Global: not listed. (However St. Kilda is a World Heritage Site).

There are two breeds of feral sheep in Britain, the Soay sheep and the Boreray sheep. The Soay sheep originates from the Island of Soay, but 107 animals were transferred to Hirta in 1932, following the evacuation of the human population from the island (Clutton-Brock and

Pemberton, 2004). The Hirta population has since remained unmanaged, and includes between 600 and 2,300 individuals, depending on survival rates in a particular year (Regan et al., 2016). Soay sheep have now been introduced to several off-shore Islands, including Lundy, Cardigan Island, Holy Isle (Arran), and Sheep Island (Sanda Island, Kintyre). There is one population of approximately 130 on the mainland in Cheddar Gorge. The Boreray sheep is confined to Boreray, St Kilda (Harris and Yalden, 2008).

Soay sheep on Hirta have been intensively studied over the last 30 years. Despite evolutionary pressures for increased body size, the Soay sheep population has shown a decrease in body size over the period of the study. This counter-intuitive trend is thought to be entirely owing to environmental change, including a shift towards milder winters, which has allowed smaller sheep to survive and breed (Ozgul et al., 2009).

Feral goat *Capra aegagrus hircus*

Status

Non-native (naturalised)

Conservation Status

- IUCN Red List Global: not listed. (Wild *Capra aegagrus* is listed at 'Vulnerable' within its native range.)

The feral goat in Britain is descended from the wild goat *Capra aegagrus*, which was introduced as domestic stock as early as 2,500 BC. It tends to use steep ground for refuge, shelter and foraging, and is largely restricted to mountainous or coastal areas with cliffs. Many populations include scrub and woodland habitats within their home range (Harris and Yalden, 2008).

Population densities range from 1.5 to 12 km^{-2}, with variation between regions and years. The highest recorded densities in Great Britain are in south west Scotland. The most recent population size estimates, which date from 1990-99, suggested that there were 5,000-10,000 (Harris and Yalden, 2008). This is higher than the estimate of 3565 individuals made in 1995 (Harris et al., 1995).

Skomer Vole *Myodes glareolus skomerensis*

Status

Non-native (naturalised)

Conservation Status

- IUCN Red List Global: not listed (island variant of *Myodes glareolus*).

The Skomer vole is an island sub-species of the bank vole, and is likely to have been accidentally introduced to Skomer (Corbet, 1964). The time of this introduction is unknown. Compared with the bank vole in mainland Britain, the home range size of the Skomer vole is much smaller (Loughran, 2014), and the population density much higher (up to four-times greater) (Healing, 1984). This may, in part, be owing to the lack of ground predators on Skomer, although avian predators are still present. The Skomer vole is most numerous in areas with sufficient ground cover. It is found in a range of habitats including scrub, rough grassland and woodland, and it can make use of the burrows dug by rabbits and Manx shearwaters (Loughran, 2006).

Red necked wallaby *Macropus rufogriseus*

Status

Non-native (naturalised)

Conservation Status

- IUCN Red List Global: LC.

This species was first introduced into zoos and wildlife parks in Great Britain in 1865 (Macdonald and Burnham 2010). Feral populations became established in the peak district and Sussex in 1940s, but are presumed to have since died out (Macdonald and Burnham 2010; Harris et al. 2008). However scattered sightings throughout Great Britain, including in Sussex, suggest that undocumented breeding populations may persist. The only known established colony is on Inchconnachan Island, Loch Lomond, Scotland, where the species has been present since at least 1975 (Harris and Yalden, 2008).

The red necked wallaby tends to feed in open ground and uses scrub and woodlands as resting locations. The population in Inchconnachan uses open birch-oak-pine woodland (Weir et al., 1995). The most recent population estimate, from 1993, suggested that there were approximately 28 individuals (Harris et al., 1995).

Parti-coloured bat *Vespertilio murinus*

Status

Vagrant

Conservation Status

- IUCN Red List Global: LC.

The parti-coloured bat forages on beetles and moths in open areas. It uses a variety of habitat types, ranging from forests and agricultural land, to around urban street lights. Summer roosts are usually in houses or buildings, but are occasionally in hollow trees (Harris and Yalden, 2008).

British records of the species are scarce. Only four individuals were recorded up until 1980, although it is becoming a more frequent. Since that time, records have gradually become more common, and there are now one or two per year. There is good potential for breeding colonies to become established in Great Britain given the frequency of summer records and the species' long- distance migration behaviour: the maximum migration distance recorded is 1780 km; Hutterer 2005.

Kuhl's pipistrelle *Pipistrellus kuhlii*

Status

Vagrant

Conservation Status

- IUCN Red List Global: LC

Relatively abundant in the Mediterranean and Middle East, the Kuhl's pipistrelle forages over a variety of habitats, including agricultural fields and around urban street lights. It feeds on Diptera, Psocoptera, Coleoptera and other small insects (Bogdanowicz, 2004). Maternity colonies are generally in buildings, whilst winter hibernacula are found in rock crevices or cellars (Harris and Yalden, 2008). Records of the Kuhl's pipistrelle are rare in Great Britain, with the first record being made in 1991 in Suffolk. Since then, around 10 other individuals have been reported. However there is potential for under-reporting because of the likelihood of confusion with other *Pipistrellus* species.

Raccoon *Procyon lotor*

Status

Non-native

Conservation Status

- IUCN Red List Global: LC

The raccoon is an opportunistic feeder, and so is able to survive in a wide variety of habitats. Despite several recorded escapes from captivity, there is as yet no evidence of a breeding population in Great Britain, though little recent information is available (Harris and Yalden, 2008). Given the high potential for this species to become invasive, and the adverse ecological effects it has generated elsewhere in Europe, it is important that monitoring for this species is conducted. The importation of racoons was made illegal in 2017 under the EU Invasive Alien Species (IAS) Regulation (1143/2014).

Appendix 7: Future prospects

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Hedgehog	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. Precise estimates are difficult because of the way in which grassland is classified. The amount of land used for rough grazing fell by 4.9% between 1998 and 2007, and the rate of loss accelerated to 7.7% between 2007 and 2014 (Khan 2015). (Note that this category overlaps with semi-natural grassland). However, most loss of arable land is because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grassland.
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan, 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat is expected in order to meet housing demand.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Hedgehog	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1997). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also an increase within existing urban areas, largely as a result of paving residential front gardens (Perry & Nawaz 2008). Rural villages currently act as refugia for hedgehogs: small home ranges, from which can be inferred higher densities, are found in rural village green spaces and gardens than in arable area (Pettitt 2017). Further built development (e.g. through urban expansion and increase in impervious surfaces in gardens) may therefore have a negative effect. High rates of supplementary feeding in some urban and suburban areas may increase local carrying capacity.	Green corridors designated as UK BAP Mosaic Habitat in 2010.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Hedgehog	Reduced tree growth in the south. Increased growth in north and west England. Pests / diseases increase with warm winters.	Decline	Population - Decline Range - Stable Habitat - Decline	
	Possible increase in total area as drier weather makes land unsuitable for arable production.			
	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Mole	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009; Chapter 15). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
Common shrew	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Bog	↑ 8.9%	Fluctuating (probably declining)
Pygmy shrew	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. Precise estimates are difficult because of the way in which grassland is classified. The amount of land used for rough grazing fell by 4.9% between 1998 and 2007, and the rate of loss accelerated to 7.7% between 2007 and 2014 (Khan 2015). (Note that this category overlaps with semi-natural grassland). However, most loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grassland.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity [b]**
Mole	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
Common shrew	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	Most types of bog degraded / deteriorating.	Habitat fragmentation likely to increase.
Pygmy shrew	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Mole	Possible increase in total area as drier weather makes land unsuitable for arable production. Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear.	DD	Population - Stable Range - Stable Habitat - Stable/decline	Although data were not comparable between the current and previous population size estimates, the range of this species is well documented. Habitat drivers are not likely to have a significant impact (although further study is advised as this species is particularly deficient in data relating to population density and the drivers of change).
Common shrew	Reduced tree growth in the south. Increased growth in north and west England. Pests / diseases increase with warm winters. Increased summer evaporation will put stress on wetland plant communities in late summer and autumn.	DD	Population -Stable/Decline Range - Stable (possible decline in Scotland) Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is predicted as a result of declines in habitat extent and quality.
Pygmy shrew		DD	Population - Stable/Decline Range - Stable (possible decline in Scotland) Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a fall in population size is predicted because of a decline in habitat extent and quality.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Water shrew	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
Lesser white-toothed shrew	All habitats except built environment		No change in area of suitable habitat. Decline in predation pressure as number of cats falling following eradication of brown rats. Possibly also some decline in interspecific competition from rats.
Rabbit	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009; Chapter 15). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Dwarf shrub heath	↓ 6.5%	Increase in deer numbers likely to result in continued overgrazing.
	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. Precise estimates are difficult because of the way in which grassland is classified. The amount of land used for rough grazing fell by 4.9% between 1998 and 2007, and the rate of loss accelerated to 7.7% between 2007 and 2014 (Khan 2015). (Note that this category overlaps with semi-natural grassland). However, most loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grassland.
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Water shrew	Bank clearance and modification may destroy burrows and alter water supplies. Widescale effects are not known. Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication, and effect of other pollutant (e.g. from road run-off or insecticides) unclear.	Loss of connectivity has occurred through dams, weirs, land drainage, embankments, channel deepening, straightening and widening (Newson 2002).
Lesser white-toothed shrew	No change for Isles of Scilly	No change for Isles of Scilly
Rabbit	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	High grazing pressure has resulted in greatly reduced quality of dwarf shrub heath.	Altered land use and fragmentation can result in the loss of foraging opportunities and shelter, which may be detrimental to survival.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Water shrew	Lakes and rivers are highly sensitive to climate change. Increases in the number of flood events are likely. Acidification and eutrophication are likely to continue, with important consequences for freshwater prey species. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population - Decline Range - Stable Habitat - Decline	Although previous trends in population size and range are unknown for this species, a decline in population size is predicted based on a reduction in habitat quality and connectivity, as well as the effect of pollution on prey species.
Lesser white-toothed shrew		Stable	Population - Stable Range - Stable Habitat - Stable	
Rabbit	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear. Increased biomass production of heathlands. Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear. Possible increase in total area as drier weather makes land unsuitable for arable production.	DD	Population - Decline Range - Stable Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is predicted to occur based on trends from the BTO Breeding Bird Survey and GWCT National Game Bag Census, as well as expert opinion (see main review).

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Brown hare	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). As brown hares select habitat based on structure, rather than the availability of nutrients, this change may be beneficial. Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009; Chapter 15). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
Mountain hare	Montane		
	Dwarf shrub heath	↓ 6.5%	Increase in deer numbers likely to result in continued overgrazing.
Red squirrel	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Brown hare	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing, and brown hare numbers are negatively associated with high intensity grazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
Mountain hare		
	High grazing pressure has resulted in greatly reduced quality of dwarf shrub heath.	Altered land use and fragmentation can result in the loss of foraging opportunities and shelter, which may be detrimental to survival.
Red squirrel	Pathogenic tree disease, e.g. to larch (*Larix spp.*) and Pine (*Pinus spp* .), will affect habitat availability and key food resources. Reduced availability of foraging resource because of damage and competition from grey squirrel.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, impacts on red squirrels unclear.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Brown hare	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear. Possible increase in total area as drier weather makes land unsuitable for arable production.	DD	Population - Stable Range - Stable Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is not thought to be likely in the near future. A decline in habitat quality is not thought to have occurred in relation to this species, as brown hares select habitat on the basis of structure rather than nutrient availability (i.e. the neglect of grassland habitat may provide improved surface resting sites and cover from predators; Smith et al., 2004, Lush et al., 2014)
Mountain hare	Increased biomass production of heathlands.	DD	Population - Decline Range - Stable Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is predicted because of a decline in habitat quality.
Red squirrel	Potential increase in seed availability linked to warming climate; however, also increased pest/disease issues. Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species.	Decline	Population - Decline Range - Decline Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Grey squirrel	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Expansion of urban and suburban habitat will continue in order to meet housing demand.
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
Beaver	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
Hazel dormouse	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Grey squirrel	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, impacts on grey squirrels unclear.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	The area of woodland within or close to urban areas is likely to have increased following initiatives to increase 'urban forests'. In addition, the species uses suitable habitats within urban parks and gardens. High rates of supplementary feeding are also likely to increase the carrying capacity.	Green corridor designated UK BAP Mosaic Habitat in 2010. Grey squirrel densities in urban areas increase with level of urbanisation (Baker and Harris 2007).
	Pathogenic tree disease, e.g. to larch (*Larix spp.*) and pine (*Pinus spp* .) will affect habitat availability and key food resources.	Recent habitat suitability models indicate that that grey squirrels exist in highly fragmented and functionally unconnected landscapes (Stevenson-Holt 2014). Connectivity is therefore unlikely to be important for this species.
Beaver	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts on beavers not yet clear.	
Hazel dormouse	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998), and a reduction in carrying capacity for dormice. Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Roadside plantings have increased available habitat in some areas as well as promoting linear connectivity.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Grey squirrel	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996), so increased seed production linked with warming climate likely to have positive impact. Urban trees threatened, leading to Increased disease / pests. **** Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996). Potential increases in seed availability in warmer climate. However, also increased pest / disease issues.	Increase	Population - Increase Range - Increase Habitat - Stable	There is some possibility of population stabilisation or decline if current field trials of reproductive control prove successful.
Beaver		Increase	Population - Increase Range - Increase Habitat - Stable	
Hazel dormouse	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species. Dormice feed on a succession of flowers and fruits from a variety of species; climate change may alter food availability and masting times. Impacts of climate change on over-winter survival are unclear.	Decline	Population - Decline Range - Stable Habitat - Decline	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Edible dormouse	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures mean that the total area of broadleaved woodland is likely to continue to increase.
Bank vole	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Expansion of urban and suburban habitat will continue in order to meet housing demand.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements owing to agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Unimproved grassland	↓ 7%	Trend based on area of rough grazing (Khan 2015). Likely decline in future because of further intensification of agriculture. However, most loss of arable land has been the result of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grazing and net loss of habitat may be minimal. The extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008).

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Edible dormouse	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, species prefers canopy, so impacts may be limited.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape. Lack of connectivity is likely to have restricted the range expansion of this species.
Bank vole	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	Pathogenic tree disease, e.g. to larch (*Larix spp.*) and pine (*Pinus spp.*) will affect habitat availability and key food resources.	Recent habitat suitability models indicate that grey squirrels exist in highly fragmented and functionally unconnected landscapes (Stevenson-Holt 2014). Connectivity is therefore unlikely to be important for this species.
	Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly owing to urban expansion, but also there is an increase in impervious surfaces within existing urban areas, largely because of the paving of residential front gardens (Perry & Nawaz 2008). However, the area of woodland within or close to urban areas is likely to have increased following initiatives to increase 'urban forests'. There are also high rates of supplementary feeding of wildlife (not intended for bank voles but potentially providing food) in some urban and suburban areas which may increase the carrying capacity of the area.	Green corridors designated as UK BAP Mosaic Habitat since 2010.
	Loss of hedgerows mainly owing to neglect. AES have delivered some local improvements, but these not reflected in national surveys. Treelines generally provide much less suitable for habitat for bank voles; and Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	Increased field size leads to fewer boundary features. Loss of hedgerows continues, mainly owing to neglect. Overall, woody linear features have increased, but quality poor. AES has delivered some local improvements, but changes to schemes means that any future improvements are uncertain.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Edible dormouse	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact. Altered beech masting times as a result of climate change may have a negative impact.	DD	Population - Increase (very slow because of low reproductive rate and habitat barriers) Range - Increase (very slow) Habitat - Increase	Although data were not comparable between the current and previous population size estimates, the range of this species is well documented and known to be increasing slowly. This trend is therefore used as a basis for the predicted slow increase in population size.
Bank vole	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species. Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996). Potential increases in seed availability in warmer climate. However, also increased pest / disease issues. Urban trees threatened - Increased disease / pests *** Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996). Potential changes seed availability linked to warming climate, but impact is likely to be positive. Growth likely to increase in north and west and decline in south. Pest / disease issues likely to increase.	Increase (Scotland) Stable (England & Wales)	Population - Stable Range - Stable Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Field vole	Unimproved grassland	↓ 7%	Trend based on area of rough grazing (Khan 2015). Likely decline in future because of further intensification of agriculture. However, most loss of arable land has been the result of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grazing and net loss of habitat may be minimal. The extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008).
Orkney vole	All natural habitats		Unclear whether trajectory of loss reported up to early 1990s is continuing
	Hedgerows, ditches and verges		Unclear whether trajectory of loss reported up to early 1990s is continuing
	Coniferous woodland		Unknown
Water vole	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways. There are anecdotal reports that mink population sizes may be falling; and in Scotland there have been extensive and co-ordinated mink control operations in some areas.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Field vole	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.
Orkney vole	In GB as a whole there is a decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Unknown for Orkney
	In GB as a whole there is a decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Unknown for Orkney
		Unknown for Orkney
Water vole	Declines in habitat suitability owing to wetland drainage, arable cultivation and watercourse canalisation.	Loss of connectivity has occurred through dams, weirs, land drainage, embankments, channel deepening, straightening and widening (Newson 2002). Conservation actions have led to some improvements. Large core patches of water voles can ensure the long term viability of water vole metapopulations in surrounding landscapes (Macpherson 2011).

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Field vole		DD	Population - Stable Range - Stable Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is predicted owing to a decline in habitat extent and quality.
Orkney vole	Likely to be increase in tree growth. Pests / diseases increase with warm winters	Decline	Population - Decline Range - Stable Habitat - Decline	Predation by stoats, which are rapidly establishing across the islands, together with the long-term decline in habitat availability and quality, is highly likely to result in a decline of this species.
Water vole	Lakes and rivers are highly sensitive to climate change. Increases in the number of flood events are likely. Acidification and eutrophication are likely to continue, with important consequences for freshwater prey species. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	Decrease	Population - Decline Range - Stable Habitat - Stable	The steep decline in population size between 1995 and the present is attributed to a decline in population density throughout the species range, from 17 - 42 per km (Harris et al., 1995) to 3 - 10 per km (see main review). Therefore a decrease in population size is possible without a substantial decline in range, although it is unlikely that the range size has remained stable. The apparent stability in range may be an artefact of an increase in recorder effort between the two time periods, which is highly likely given the vulnerable status and ongoing conservation efforts. A decline in both population size and occupancy is therefore predicted, unless conservation efforts (particularly mink control) have a major impact at the landscape scale.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Harvest mouse	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009; Chapter 15). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Wetland habitats		Countryside Survey data from a sample of 591 1x1 km squares show a small decrease in reed beds since 1990 (Haines-Young et al., 2000). There has been a 12.5% increase in the number of ponds between 1998-2007, which may provide suitable reed bed habitat.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Harvest mouse	Fertiliser application declining since mid-1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies. Change in agricultural practice, e.g. switch to use of winter sown crops harvested in late summer, likely to result in loss of nests and young.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Over 50% of ponds were in 'very poor' condition in 2007 (Carey 2008), although the quality of associated reed bed habitat is uncertain. Restoration projects throughout Britain are likely to have resulted in high quality reed beds although habitat condition in relation to harvest mice is uncertain.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Harvest mouse	Droughts in summer in the south-east and waterlogging because of wetter winters in northern areas are likely to reduce arable farming in GB. Overwinter survival and recruitment is also likely to be enhanced by warmer winters.	DD	Population - Decline Range - Stable Habitat - Decline	Although data were not comparable between the current and previous population size estimates, a decline in population size is predicted as a result of declining habitat extent and quality. Data are urgently required to ensure current distribution maps are accurate.
	Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Wood mouse	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Unimproved grassland	↓ 7%	Trend based on area of rough grazing (Khan 2015). Likely future decline because of further intensification of agriculture. However, most loss of arable land has been the result of a transfer to Neutral Grassland, reflecting less intensive management — therefore some areas may be functionally similar to rough grazing and net loss of habitat may be minimal. The extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008).
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Expansion of urban and suburban habitat will continue in order to meet housing demand.
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Wood mouse	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, species is highly adaptable and tends to persist in degraded habitats, so impacts unclear.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation owing to local and regional loss; and also loss of boundary features with increasing field sizes.
	Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also there is an increase in impervious surfaces within existing urban areas, largely because of the paving of residential front gardens (Perry & Nawaz 2008). However, the area of woodland within or close to urban areas is likely to have increased following initiatives to increase 'urban forests'. There are also high rates of supplementary feeding of wildlife (not intended for wood mice but potentially providing suitable food) in some urban and suburban areas which may increase the carrying capacity of the area.	Green corridors designated as UK BAP Mosaic Habitat since 2010.
	Highly adaptable and opportunistic, likely to adapt to declines in woodland quality.	
	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Treelines generally provide much less suitable for habitat for mice; and Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	Increased field size leads to fewer boundary features. Loss of hedgerows continues, mainly owing to neglect. Overall, woody linear features have increased, but quality poor. Agri-environment schemes have delivered some local improvements, but changes to schemes means that any future improvements are uncertain.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Wood mouse	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species.	Stable	Population - Stable Range - Stable Habitat - Stable	
	Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996). Potential increases in seed availability linked to warming climate. However, also increased pest / disease issues.			
	Overwinter survival is directly related to autumn seed availability (Gurnell 1983, 1996). Potential increases in seed availability linked to warming climate. However, also increased pest / disease issues.			
	Overwinter survival is directly related to autumn seed availability. Potential changes seed availability linked to warming climate; however, impact is likely to be positive. Growth likely to increase in north and west and decline in south. Pest / disease issues likely to increase.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Yellow-necked mouse	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
House mouse	Rural (all habitats)	↓ 4.5%	Urban expansion into rural areas will provide increased number of dwellings to occupy, potentially affecting population size
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Expansion of urban and suburban habitat will continue in order to meet housing demand.
Brown rat	Rural (all habitats)	↓ 4.5%	Urban expansion into rural areas will provide increased number of dwellings to occupy affecting population size
	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Brown rats are highly adaptable and found in both urban and grassland habitats, although population densities in urban (and suburban) areas are likely to be higher than in grassland. Expansion of urban and suburban habitat will continue in order to meet housing demand.
Black rat	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Expansion of urban and suburban habitat will continue in order to meet housing demand.
	Coastal cliffs		No change.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Yellow-necked mouse	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Treelines generally provide much less suitable for habitat for mice; and Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	Increased field size leads to fewer boundary features. Loss of hedgerows continues, mainly owing to neglect. Overall, woody linear features have increased, but quality poor. Agri-environment schemes have delivered some local improvements, but changes to schemes means that any future improvements are uncertain.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, impacts on species unclear.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
House mouse	Newly built houses are likely to be less accessible by the species.	
	Newly built houses are likely to be less accessible by the species.	Populations driven by births and deaths within relatively isolated patches, rather than movement between them (Pocock et a. 2004; Gray et al., 2000). Commensal mice are adapted to being transported by human activities (Baker et al., 1994).
Brown rat	Unclear	
	Change in regulations on rodenticide use may reduce numbers where resistance is present. Increasing availability of waste food/residues from refuse/sewers in urban areas. In rural and suburban areas, increase in food availability from domestic chickens.	Urban expansion and enhanced connectivity of urban environments likely to benefit this species.
Black rat	A decline in the availability of warehouses and other buildings suitable for black rats in ports.	Urban expansion and enhanced connectivity of urban environments, but this may increase competition from brown rats
	No change.	No change.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Yellow-necked mouse	Overwinter survival directly related to autumn seed availability. Potential changes seed availability linked to warming climate; however, impact is likely to be positive. Growth likely to increase in north and west and decline in south. Pest / disease issues likely to increase. Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species.	DD	Population - Potential increase Range - Potential increase Habitat - Stable	The increase in range observed between 1995 and 2016 is relatively small, and it is unclear whether it reflects a true increase in range or increased recorder effort.
House mouse	Recruitment and over-winter survival likely to increase with warmer winters.	Stable	Population - Stable Range - Stable (possible decline) Habitat - Stable (possible decline)	
Brown rat	Recruitment and over-winter survival likely to increase with warmer winters.	DD	Population - Stable/Increase Range - Stable Habitat - Increase	Although data were not comparable between the current and previous population size estimates, there is potential for an increase in population size owing to the increase in urban habitat.
Black rat	Recruitment and over-winter survival likely to increase with warmer winters. No change.	Decline	Population - Decline Range - Decline Habitat - Decline Note species may already be extinct	This species may already be extinct in GB, though there is high potential for confusion with brown rat. There is the possibility of reinforcement in ports by animals entering via shipping, but inland populations are probably extinct with little prospect of recovery.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Wild cat	Coniferous woodland	↓ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures mean that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Unimproved grassland	↓ 7%	Trend based on area of rough grazing (Khan 2015). Likely decline in future because of further intensification of agriculture. However, most loss of arable land has been the result of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grazing and net loss of habitat may be minimal. The extent to which this was the because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008).
Fox	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Urban and suburban habitat expansion will continue in order to meet housing demand.
	Rural (all habitats)	↓ 4.5%	Loss of rural habitats due to urban expansion.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Wild cat	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. In areas where there is a decline in forest cover, the species adapts by making greater use of open areas and increasing group sizes (Hewison 2001). However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation because of local and regional loss; and also loss of boundary features with increasing field sizes.
Fox	Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also there is an increase in impervious surfaces within existing urban areas, largely as a consequence of the paving of residential front gardens (Perry & Nawaz 2008). However, the area of woodland within or close to urban areas is likely to have increased following initiatives to increase 'urban forests'. There are also high rates of supplementary feeding in some urban and suburban areas which may increase the carrying capacity of the area.	Green corridors designated UK BAP Mosaic Habitat since 2010.
	Unclear whether changing quality of agricultural and natural habitats has impact on this species.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Wild cat	Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.	Decrease	Population - Decline Range - Decline Habitat - Stable	
Fox	Recruitment and over-winter survival likely to increase with warmer winters. Recruitment and over-winter survival likely to increase with warmer winters.	DD	Population - Stable Range - Stable Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Badger	Urban	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to urban areas. Urban and suburban habitat expansion will continue in order to meet housing demand.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which generally is likely to be of low value as foraging resource for wildlife - though possibly not for badgers - had increased by 18% and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been because of a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Badger	Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also there is an increase in impervious surfaces within existing urban areas, largely as a consequence of the paving of residential front gardens (Perry & Nawaz 2008). However, the area of woodland within or close to urban areas is likely to have increased following initiatives to increase 'urban forests'. There are also high rates of supplementary feeding in some urban and suburban areas which may increase the carrying capacity of the area.	Green corridors designated UK BAP Mosaic Habitat since 2010.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, species is highly adaptable and tends to persist in degraded habitats, so impacts unclear.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. These issues may have limited impact on badgers.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies. Change in agricultural practice, e.g. increasing production of field maize, may provide an additional foraging resource.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Badger	Recruitment and over-winter survival likely to increase with warmer winters.	Increase	Population - Stable (except in intensive cull areas) Range - Stable Habitat - Stable	Although an increase in population size has been estimated in the past 20 years, trends are based on uncertain data and should be viewed with caution (see main report for full details). Increases are likely to be the result of recovery from persecution and may not be ongoing. The range for this species is well documented and has remained stable over the last 20 years, leading to a prediction of 'stable' for the future trends in population size.
	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Seed production is likely to increase with warming climate, with a positive impact on the species.			
	Possible increase in total area as drier weather makes land unsuitable for arable production.			
	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Otter	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Coastal		Although populations in coastal habitats are not estimated in the current review, they are likely to hold a high proportion of the otter population in Scotland (Harris et al., 1995).
Pine marten	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Otter	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of other pollutants (e.g. from road run-off) are unclear.	Loss of connectivity has occurred through dams, weirs, land drainage, embankments, channel deepening, straightening and widening (Newson 2002).
	The Coastal Margin habitats (i.e. sand dunes, machair, salt marsh, coastal lagoons) have declined in area by an estimated 16.8% over the last 60 years, mainly through development pressures for residential, tourism and industrial use, and agricultural intensification; habitat quality has also deteriorated (Williams 2006).	
Pine marten	Maturing coniferous forests are showing increased structural diversity which may increase the availability of denning sites. Grazing, and loss of understorey, may have a negative effect because of loss of prey species and cover. Pathogenic tree disease, e.g. to larch (*Larix spp*.) and pine (*Pinus spp*.) may affect habitat availability and key food resources.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. The species is flexible and likely to be able to withstand some losses of connectivity: densities are highest where forest cover is 20-35%.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Reductions in open space may limit suitability of habitat for hunting, though this may be counteracted by increased fragmentation.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland *per se* between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Species is adaptable so connectivity may have limited impact.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Otter	Lakes and rivers are highly sensitive to climate change. Increases in the number of flood events are likely. Acidification and eutrophication has important consequences for freshwater ecosystems, and potentially to prey availability. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	Increase	Population - Increase Range - Increase Habitat - Stable	
	Coastal-dwelling otters require a ready supply of fresh water to wash the salt out of their fur. Changes in the volume and timing of freshwater discharge may therefore have some effect on behaviour and distribution. Effects on prey species are likely to have a more notable (although currently uncertain) effect on otter distribution.			
Pine marten	Effects of climate change on woodland habitats are unlikely to have a substantial effect on this species.	Increase	Population - Increase Range - Increase Habitat - Stable	
	Effects of climate change on woodland habitats unlikely to have a substantial effect on this species.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Stoat	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed (105,800ha increase from 1986-1991 compared to 25,000ha increase from 2001-2006), increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment scheme and other farm subsidies.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Stoat	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies, reducing availability of small mammal prey.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Maturing coniferous forests are showing increased structural diversity which may increase the availability of denning sites. Grazing, and loss of understorey, may have a negative effect because of the loss of prey species and cover. Pathogenic tree disease, e.g. to larch (*Larix spp* .) and pine (*Pinus spp* .) may affect habitat availability and key food resources.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape. The species is flexible and likely to be able to withstand some losses of connectivity: densities are highest where forest cover is 20-35%.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Reductions in open space may limit suitability of habitat for hunting, though this may be counteracted by increased fragmentation.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Species is adaptable so connectivity may have limited impact.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Stoat	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear.	DD	Population - Unknown Range - Unknown Habitat - Stable	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the drivers of population change mean that the reported stable range sized is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.
	Effects of climate change on woodland habitats are unlikely to have a substantial effect on this species.			
	Effects of climate change on woodland habitats unlikely to have a substantial effect on this species.			
	Possible increase in total area as drier weather makes land unsuitable for arable production.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Weasel	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.
	Unimproved grassland	↓ 7%	Trend based on area of rough grazing (Khan 2015). Likely decline in future because of further intensification of agriculture. However, most loss of arable land has been the result of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grazing and net loss of habitat may be minimal. The extent to which this was the because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008).
	Improved grassland	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
Polecat	All		
Mink	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Coastal		It has not been possible to derive population estimates for mink in coastal areas. These are likely to form a high proportion of the mink population in Scotland (Harris et al., 1995).

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Weasel	Fertiliser application declining since mid 1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Value of field margins to wildlife may decline with alteration in subsidy policies, reducing availability of small mammal prey.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features frequently associated with this species (Carey et al., 2008).	Habitat fragmentation because of local and regional loss; and also loss of boundary features with increasing field sizes.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected.	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.
Polecat		
Mink	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of other pollutants (e.g. from road run-off unclear)	Loss of connectivity has occurred through dams, weirs, land drainage, embankments, channel deepening, straightening and widening (Newson 2002). However, species is adaptable and makes extensive use of non-riparian habitats, so connectivity may have limited impact.
	The Coastal Margin habitats (i.e. sand dunes, machair, salt marsh, coastal lagoons) have declined in area by an estimated 16.8% over the last 60 years, mainly through development pressures for residential, tourism and industrial use, and agricultural intensification; habitat quality has also deteriorated (Williams 2006).	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Weasel	Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear. Possible increase in total area as drier weather makes land unsuitable for arable production.	DD	Population - Unknown Range - Unknown Habitat - Stable	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the drivers of population change means that the reported stable range sized is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.
Polecat		Increase	Population - Increase Range - Increase Habitat - Stable	
Mink	Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms (therefore prey species). Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities. However, species is a generalist feeder and the effects of these changes are unlikely to be significant. Changes to coastal habitats may affect the availability of prey species, but effect on mink is not likely to be significant because of their generalist feeding behaviour.	DD	Population - Stable / decline (possible future decline owing to control measures) Range - Stable / decline (possible future decline owing to control measures) Habitat - Stable	Although data were not comparable between the current and previous population size estimates, the species is almost at the limit of its range in GB. This limitation, combined with control measures, means that the species is likely to be stable, with possible declines depending on the efficacy of control measures on a landscape scale.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Wild boar	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
Red deer	Dwarf shrub heath	↓ 6.5%	Increase in deer numbers means that overgrazing is likely to continue
	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures mean that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Wild boar	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Species highly adaptable so fragmentation unlikely to have a negative impact.
	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape. The species is highly adaptable and habitat fragmentation is unlikely to have a major impact.
Red deer	Reduction in sheep grazing (15% decline in population since 2005) leads to more grazing available for deer.	Dwarf shrub heath is in decline and is becoming increasingly fragmented in the landscape.
	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. In areas where there is a decline in forest cover, the species adapts by making greater use of open areas and increasing group sizes (Hewison 2001). However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Wild boar	Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Overwinter survival and recruitment likely to be higher with warmer winters.	Increase	Population - Increase Range - Increase Habitat - Stable	
	Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.			
Red deer	Increased biomass production of heathlands.	Increase (England, Wales), Stable (Scotland)	Population - Increase overall (stable in Scotland) Range - Increase overall (stable in Scotland) Habitat - Stable	
	Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
	Coniferous woodland	↑ 6.5%	Although the rate of increased has slowed, increased interest in afforestation as part of climate change mitigation measures mean that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. In areas where there is a decline in forest cover, the species adapts by making greater use of open areas and increasing group sizes (Hewison 2001). However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
	Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Sika deer	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Dwarf shrub heath	↓ 6.5%	Increase in deer numbers means that overgrazing likely to continue.
Fallow deer	Coniferous woodland	↑ 6.5%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Sika deer	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Unclear whether reduced connectivity has negative impact on species: there is evidence that there is increased use of open spaces and larger group sizes in these scenarios (Hewison 2001).
	High grazing pressure leads to greatly reduced quality of dwarf shrub heath. Conversely, reduced grazing leads to deterioration of lowland heath.	
Fallow deer	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. In areas where there is a decline in forest cover, the species adapts by making greater use of open areas and increasing group sizes (Hewison 2001). However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al. 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al. 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al. 2008), with likely negative effects on woodland connectivity across the wider landscape. Unclear whether reduced connectivity has negative impact on species: there is evidence that there is increased use of open spaces and larger group sizes in these scenarios (Hewison 2001).

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Sika deer	Reduced tree growth in the south, and increased growth in north and west England. Pests / diseases likely to increase w i t h warm winters. Overwinter survival and recruitment likely to be higher with warmer winters.	Increase	Range - Increase Habitat - Stable	
	Increased biomass production of heathlands.			
Fallow deer	Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.	Increase	Population - Increase Range - Increase Habitat - Stable	
	Reduced tree growth in the south, and increased growth in north and west England. Pests / diseases likely to increase w i t h warm winters. Overwinter survival and recruitment likely to be higher with warmer winters.			

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Roe deer	Coniferous woodland	↑6.5%	Although the rate of increase has slowed (105,800ha increase from 1986-1991 compared to 25,000ha increase from 2001-2006), increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
Chinese water deer	Fen, marsh and swamp	↓ 8.2%	
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Arable	↓ 8.3%	Most loss of arable land in the last few decades has been caused by a transfer to neutral grassland, although the extent to which this was because of agri-environment schemes, as opposed to neglect, is not known (Carey et al., 2008). Change in the use and extent of arable land results from decisions of individual land managers in the light of markets, policies, the characteristics of the land, environmental conditions, available knowledge and technology, and the attitudes and objectives of the land managers themselves (McIntyre et al., 2009). This means that the extent and vegetation of this habitat may change very rapidly and that changes are difficult to predict.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Roe deer	Maturing coniferous forests are showing increased structural diversity. Grazing, and loss of understorey, may have a negative effect because of loss of prey species and cover. Pathogenic tree disease, e.g. to larch (*Larix spp* .) and pine (*Pinus spp* .) may affect habitat availability and key food resources.	Fragmentation may increase hybridisation as domestic cats prefer open areas / wildcats prefer woodland (Germain et al.,, 2009). The increase in domestic / feral cats where woodland is lost to housing developments in areas important for wildcats may be a significant conservation issue. No data are available on the connectivity of coniferous woodland; however, there was a 7.7% decline in Scotland between 1998 and 2007 (Carey et al., 2008).
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Unclear whether reduced connectivity has negative impact on species: there is evidence that there is increased use of open spaces and larger group sizes in these scenarios (Hewison 2001).
Chinese water deer	Changes in land management may positively affect Chinese water deer.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Unclear whether reduced connectivity has negative impact on species: there is evidence that there is increased use of open spaces and larger group sizes in these scenarios (Hewison 2001).
	Fertiliser application declining since mid-1990s. Increase in pest / pathogens expected. Changes in pesticides expected following regulatory changes, but impact on prey species unclear. Soil invertebrates and other aspects of soil health likely to decline as a result of continuing changes in agricultural practice. Increased planting of winter crops may provide additional foraging resource (Quine 2004).	Increased field size and fewer boundary features. Some recent improvements from agri-environment schemes, though little evidence for positive impact on measured receptor species.

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Roe deer	Effects of climate change on woodland habitats are unlikely to have a substantial effect on this species. Reduced tree growth in the south, and increased growth in north and west England. Pests / diseases likely to increase with warm winters. Overwinter survival and recruitment likely to be higher with warmer winters.	Increase (England, Wales), Stable (Scotland)	Population - Stable Range - Stable Habitat - Stable	
Chinese water deer	Reduced tree growth in the south, and increased growth in north and west England. Pests / diseases likely to increase with warm winters. Overwinter survival and recruitment likely to be higher with warmer winters. Droughts in summer in the south-east, and waterlogging owing to wetter winters in northern areas, are likely to affect arable farming, although the effect of these changes on the species is unclear.	DD	Population - Increase Range - Increase Habitat - Stable	Although data were not comparable between the current and previous population size estimates, changes to the species habitat are unlikely to have a significant detrimental effect. The species is therefore predicted to increase based on previous observed increase in range size.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Muntjac deer	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Coniferous woodland	↑ 6.4%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Muntjac deer	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. However, these changes are unlikely to affect this species.	The UK Biodiversity Indicator Report (JNCC 2012) suggests little or no change in the connectivity of broadleaved woodland per se between 1990 and 2007; with increases in the area of woodland being likely to improve connectivity. Conversely, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effects on woodland connectivity across the wider landscape. Unclear whether reduced connectivity has negative impact on species: there is evidence that there is increased use of open spaces and larger group sizes in these scenarios (Hewison 2001).
	Maturing coniferous forests are showing increased structural diversity. Overgrazing may influence habitat quality.	No data are available on the connectivity of coniferous woodland. However, the area of coniferous woodland has remained stable between 1998 and 2007 in each of the GB countries except Scotland, where there has been a 7.7% decline. Major changes in connectivity within this habitat type are therefore unlikely. In areas where there is a decline in forest cover, the species adapts by making greater use of open areas and increasing group sizes (Hewison 2001). However, between 1998 and 2007, there was a 6.1% decline in hedgerows, a 1.7% decline in total woody linear features, and a loss of parkland trees in GB (Carey et al., 2008), with likely negative effect on woodland connectivity across the wider landscape.

<table>
<tr><th rowspan="2">Species</th><th></th><th rowspan="2">Change in population size (1995 - 2016)</th><th rowspan="2">Future prospects rating</th><th rowspan="2">Explanatory notes</th></tr>
<tr><th>Climate change (from National Ecosystem Assessment)</th></tr>
<tr><td rowspan="2">Muntjac deer</td><td>Reduced tree growth in the south, and increased growth in north and west. Pests / diseases likely to increase with warm winters. Overwinter survival and recruitment likely to be higher with warmer winters.</td><td rowspan="2">Increase</td><td rowspan="2">Population - Increase
Range - Increase
Habitat - Stable</td><td rowspan="2"></td></tr>
<tr><td>Increased tree pest / disease issues with warming climate. Overwinter survival and recruitment likely to be higher with warmer winters.</td></tr>
</table>

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Greater horseshoe bat	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. Precise estimates are difficult because of the way in which grassland is classified. The amount of land used for rough grazing fell by 4.9% between 1998 and 2007, and the rate of loss accelerated to 7.7% between 2007 and 2014 (Khan 2015). (Note that this category overlaps with semi-natural grassland). However, most loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grassland.
	Improved grassland (foraging)	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Urban (roosts in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases because of the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand. This has significant issues for hibernation and also some maternity sites which are situated in increasingly urbanised environments.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Greater horseshoe bat	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008). Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Connectivity of woodland varies regionally but is low overall. Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB, with likely negative effects on connectivity.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. Decline in boundary features may have negative impact on foraging opportunities.	
	Loss of hedgerows mainly because of neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity (Carey et al., 2008). Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. May affect suitability of foraging habitat.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Greater horseshoe bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	Increase	Population - Increase Range - Increase Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Lesser horseshoe bat	Broadleaved woodland (foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban (roosts in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand. This has significant issues for hibernation and also some maternity sites which are situated in increasingly urbanised environments.
	Riparian (foraging)	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
Alcathoe bat	Unknown		**DD**

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [(b)]
Lesser horseshoe bat	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. May affect suitability of foraging habitat.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB, with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface.	
	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of these, and other forms of pollution (e.g. polyaromatic hydrocarbons from road run-off), on prey species unclear.	
Alcathoe bat	**DD**	**DD**

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Lesser horseshoe bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	Increase	Population - Increase Range - Increase Habitat - Stable	
Alcathoe bat	**DD**	DD	Population - Unknown Range - Unknown Habitat - Unknown	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Whiskered bat	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Broadleaved woodland (roosts and foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Whiskered bat	Loss of hedgerows mainly owing to neglect. AES have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability o f roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of paving residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging habitat may be falling.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Whiskered bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population- Unknown Range- Unknown Habitat - Unknown	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Brandt's bat	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Broadleaved woodland (roosts and foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Brandt's bat	Loss of hedgerows mainly owing to neglect. AES have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability o f roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of paving residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging habitat may be falling.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Brandt's bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population- Unknown Range- Unknown Habitat - Unknown	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Bechstein's bat	Broadleaved woodland (roosts and foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Unimproved grassland (foraging adjacent to woodland)	↓ 7%	Likely decline in future as agricultural intensification continues. The amount of land used for rough grazing has declined by 15% over the last 15 years (Khan 2015). However, most loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas have some functional similarity to rough grassland. However, ploughing of permanent pasture is likely to be particularly detrimental to key prey species.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Bechstein's bat	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging and roosting habitats may be falling. Strong evidence of substantial declines in abundance of larger moths in Southern Britain (40% between 1968-2007) (Fox et al., 2013).	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation by green bridges, but these are unlikely to have a population wide impact). Isolated populations may be particularly negatively affected by further loss of connectivity. Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008) with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface.	
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008). Strong evidence of substantial declines in abundance of larger moths in Southern Britain (40% between 1968-2007) (Fox et al., 2013).	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Bechstein's bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population- Unknown Range-Stable Habitat - Decline	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change mean that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Daubenton's bat	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Broadleaved woodland (roosts and, to some extent, foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Daubenton's bat	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread diffuse particulate pollution and eutrophication. Impacts of these, and other forms of pollution (e.g. polyaromatic hydrocarbons from road run-off), on prey species unclear.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability o f roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of paving residential front gardens (Perry & Nawaz 2008). Improvements in some aspects of water quality occurred following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impact of these, and other pollutants (such as polychlorinated hydrocarbons), carried in storm water run-off, on prey species abundance is unclear. The area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging and roosting opportunities.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of habitat for roosting may therefore be falling.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Daubenton's bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population - Unknown Range - Stable Habitat - Unknown	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change mean that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain; and evidence on different habitat types is conflicting.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Greater mouse-eared bat	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
Natterer's bat	Riparian margins (foraging)	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Broadleaved woodland (foraging and roosts)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Greater mouse-eared bat	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. However, this species is likely to benefit from lack of vegetation on woodland floor. Suitability of foraging and roosting habitats may be falling. Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population wide impact).
Natterer's bat	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of these and other forms of pollution (e.g. polyaromatic hydrocarbons from road run-off) on prey species unclear.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging and roosting habitats may be falling. Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains. Dung flies are key prey item and are likely to be negatively affected by use of anthelmintics in livestock.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Greater mouse-eared bat	Overwinter survival and may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. May be a range shift northwards in the species across Europe which could potentially increase GB population.	Stable	Population- Decline Range- Decline Habitat - Stable	Only a single individual is known in GB so it is difficult to draw inferences, particularly because it may be migratory. Species is likely to be affected by same factors as other *Myotis spp.* in GB, and is likely to become extinct unless additional individuals are identified and efforts made to create suitable linked habitat.
Natterer's bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population- Unknown Range- Stable Habitat - Decline	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change mean that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.

Species	Habitat drivers		
	Priority habitats	**Change in area 1990 - 2007** (a)	**Current trajectory of trend**
Serotine bat	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. The amount of land used for rough grazing has declined by 15% over the last 15 years (Khan 2015). However, the majority of loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas have some functional similarity to rough grassland. However, ploughing of permanent pasture is likely to be particularly detrimental to key prey species.
	Improved grassland (foraging)	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases resulting from the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Broadleaved woodland	↑ 4.7%	Although the area of woodland is increasing, planting rates have declined in the last 20 years so the rate of increase in woodland area has also declined. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Serotine bat	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008).	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. Decline in boundary features, and recent increases in temporary grassland and losses of permanent grassland (Khan 2015), may have negative impact on foraging opportunities.	
	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of paving residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging habitat may be falling.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Serotine bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect, and this species is particularly vulnerable to high juvenile mortality if weather in summer is poor. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population - Unknown Range - Increase Habitat - Decline	

Species	Habitat drivers		
	Priority habitats	**Change in area 1990 - 2007 (a)**	**Current trajectory of trend**
Leisler's bat	Improved grassland (foraging)	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. The amount of land used for rough grazing has declined by 15% over the last 15 years (Khan 2015). However, most loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas have some functional similarity to rough grassland. However, ploughing of permanent pasture is likely to be particularly detrimental to key prey species.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Leisler's bat	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. Decline in boundary features may have negative impact on foraging opportunities.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008). Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging and roosting habitats may be falling. Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Leisler's bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population- Unknown Range- Unknown Habitat - Stable	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change mean that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Noctule bat	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. The amount of land used for rough grazing has declined by 15% over the last 15 years (Khan 2015). However, the majority of loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas have some functional similarity to rough grassland. However, ploughing of permanent pasture is likely to be particularly detrimental to key prey species.
	Improved grassland (foraging)	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Broadleaved woodland	↑ 4.7%	Although the area of woodland is increasing, planting rates have declined in the last 20 years so the rate of increase in woodland area has also declined. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Noctule bat	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008).	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. Decline in boundary features, and recent increases in temporary grassland and losses of permanent grassland (Khan 2015), may have negative impact on foraging opportunities.	
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging habitat may be falling.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Noctule bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect, and this species is particularly vulnerable to high juvenile mortality if weather in summer is poor. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population - Unknown Range - Unknown Habitat - Unknown	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Common pipistrelle bat	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [b]
Common pipistrelle bat	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability o f roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a result of paving of residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts on prey species unclear.	
	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Common pipistrelle bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population - Unknown Range - Stable Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Soprano pipistrelle bat	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Broadleaved woodland (foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases owing to the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Hedgerows & total woody linear features	↓ 5.9% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).

<table>
<tr><th rowspan="2">Species</th><th colspan="2">National Ecosystem Assessment (2011) unless stated otherwise</th></tr>
<tr><th>Quality</th><th>Connectivity [b]</th></tr>
<tr><td rowspan="4">Soprano pipistrelle bat</td><td>Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of these, and other forms of pollution (e.g. polyaromatic hydrocarbons from road run-off), on prey species abundance is unclear.</td><td rowspan="4">Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.</td></tr>
<tr><td>The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. May affect suitability of foraging habitat.</td></tr>
<tr><td>Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of the paving of residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.</td></tr>
<tr><td>Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.</td></tr>
</table>

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Soprano pipistrelle bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population - Unknown Range - Stable Habitat - Stable	

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Nathusius' pipistrelle bat	Riparian	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.
	Broadleaved woodland	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Hedgerows & total woody linear features	↓ 5.7% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
Barbastelle bat	Broadleaved woodland (roosts and foraging)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban (roosts frequently in buildings, and impact of urban expansion on roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases because of the conversion of grassland to urban areas. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Riparian margins (foraging)	↓ 2.7%	Changes in habitat quality are more likely than substantial changes in length of waterways.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	Connectivity [(b)]
Nathusius' pipistrelle bat	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread issues of diffuse particulate pollution and eutrophication. Impacts of these, and other forms of pollution (e.g. polyaromatic hydrocarbons from road run-off) on prey species unclear.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. May affect suitability of foraging habitat.	
	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	
Barbastelle bat	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. May be loss of suitable foraging and roosting habitats. Strong evidence of substantial declines in abundance of larger moths in southern Britain (40% between 1968-2007) (Fox et al., 2013).	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface.	
	Improvements in some aspects of water quality following the banning of organochlorine pesticides, but widespread diffuse particulate pollution and eutrophication. Impacts of these, and other pollution (e.g. polyaromatic hydrocarbons from road run-off), on prey species abundance is unclear.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Nathusius' pipistrelle bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase. Lakes and rivers are highly sensitive to climate change. Increase in the number of flood events likely. Acidification and eutrophication have major consequences for freshwater organisms. Reduced summer rainfall and increased summer evaporation will put stress on wetland plant communities.	DD	Population - Unknown Range - Unknown Habitat - Stable	
Barbastelle bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population - Unknown Range - Unknown Habitat - Decline	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change means that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.

Species	Habitat drivers		
	Priority habitats	Change in area 1990 - 2007 (a)	Current trajectory of trend
Brown long-eared bat	Broadleaved woodland (foraging and roosts)	↑ 4.7%	Increased interest in afforestation as part of climate change mitigation measures means that the total area of broadleaved woodland is likely to continue to increase. However, the current trajectory of increase is modest once the loss of existing woodlands is taken into account; and the available statistics do not adjust for woodland recently converted into another land use (Forestry Commission 2017, Forestry Commission 2016). The rate of new planting of woodland (conifer and broadleaved combined) has fallen over the past 20 years, whilst the rate of restocking has remained approximately stable in all countries.
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases because of the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.
	Coniferous woodland	↑ 6.4%	Although the rate of increase has slowed, increased interest in afforestation as part of climate change mitigation measures means that the total area of woodland is likely to continue to increase.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Brown long-eared bat	The key threats to semi-natural woodland are overgrazing, habitat fragmentation and isolation, invasion by non-native species, unsympathetic forestry practices, lack of appropriate management, air pollution and new pests and diseases. The abandonment of coppicing has resulted in increased shadiness, reductions in understorey and open space, and increases in deadwood (Kirby et al., 1998). Substantial declines (26%) in ancient woodland indicator species between 1998 and 2007 in GB (Carey et al., 2008), indicative of declining woodland quality. Suitability of foraging and roosting habitats may be falling. Strong evidence of substantial declines in abundance of larger moths (28% between 1968-2007) (Fox et al., 2013). Losses in southern Britain were greater (40%), whereas in northern Britain losses were offset by gains.	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB (Carey et al., 2008), with likely negative effects on connectivity.
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface. Area of urban greenspace declined from 1980-2000, thereby reducing the quality of 'urban areas' as a habitat. Rate of decline probably slowing (not monitored). The level of impervious cover in urban areas has increased. This is partly because of urban expansion, but also because of an increase in impervious surfaces within existing urban areas, largely as a consequence of paving residential front gardens (Perry & Nawaz 2008). This may affect availability of foraging areas. However, the area of woodland within or close to urban areas is likely to have increased following initiatives to create 'urban forests' which may provide foraging opportunities.	
	Maturing coniferous forests are showing increased structural diversity. Grazing, and loss of understorey, may have a negative effect because of loss of prey species and cover. Pathogenic tree disease, e.g. to larch (*Larix spp* .) and pine (*Pinus spp* .) may affect habitat availability and key food resources.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Brown long-eared bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population - Unknown Range - Stable Habitat - Stable	Although there has not been a change in estimated range size in the last 20 years, a lack of data on population densities, size, and the conflicting effects of drivers of population change means that the reported stable range size is not considered to be sufficient evidence for a stable population. The future prospects for population size for this species are therefore uncertain.

Species	Habitat drivers		
	Priority habitats	**Change in area 1990 - 2007** (a)	**Current trajectory of trend**
Grey long-eared bat	Unimproved grassland	↓ 7%	Likely decline in future as agricultural intensification continues. Precise estimates are difficult because of the way in which grassland is classified. The amount of land used for rough grazing fell by 4.9% between 1998 and 2007, and the rate of loss accelerated to 7.7% between 2007 and 2014 (Khan 2015). (Note that this category overlaps with semi-natural grassland). However, the majority of loss of arable land has been because of a transfer to Neutral Grassland, reflecting less intensive management - therefore some areas may be functionally similar to rough grassland.
	Improved grassland (foraging)	↓ 1.4%	Livestock numbers declining, therefore further increase in the area of improved grassland unlikely unless climate change induces a move away from arable production and back to livestock. However, the trend of change from semi-improved to improved grassland, and hence declining habitat suitability for many species, is likely to continue (note: in our analyses, the category 'improved' grassland includes semi-natural grasslands). In addition, between 2007 and 2014, the amount of temporary grassland, which is likely to be of low value as foraging resource for wildlife, had increased by 18%, and the amount of permanent improved grassland had declined by 2.4% (Khan 2015). This reverses the previous trends and possibly reflects changes to agri-environment schemes and other farm subsidies.
	Hedgerows & total woody linear features	↓ 5.9% hedgerows ↑ 9.9% woody linear features	In the most recent decade for which data are available (1998-2007), 6.1% decline in hedgerows and 1.7% decline in other woody linear features in GB, implying that losses are likely to continue. Livestock numbers declining, reducing need for impermeable field boundaries. Loss of hedgerows may reduce foraging opportunities. Some local improvements because of agri-environment schemes, but overall trend is for decline in hedgerow structure and quality, future trends uncertain because of changes to subsidy schemes. Long-term decline (now stabilised) in quality of plant communities associated with hedgerow bottoms (Carey et al., 2008).
	Urban (roosts frequently in buildings, and impact of urban expansion on suitability of roosting/foraging areas)	↑ 4.5%	Regeneration of brownfield sites may limit rate of expansion of urban areas, although this is unlikely to outweigh increases because of the conversion of grassland to built environments. Expansion of urban and suburban habitat expected in order to meet housing demand.

Species	National Ecosystem Assessment (2011) unless stated otherwise	
	Quality	**Connectivity** [b]
Grey long-eared bat	Decline in diversity of unimproved grassland with likely knock-on effects for foraging resources. Also declines in structural complexity of rough grassland and in species richness and structural complexity of hedgerows, ditches and other marginal features that may be important to prey abundance (Carey et al., 2008). Strong evidence of substantial declines in abundance of larger moths in southern Britain (40% between 1968-2007) (Fox et al., 2013).	Artificial night lighting potentially severs commuting routes and delays emergence time. Habitat fragmentation owing to new roads/infrastructure disrupts commuting routes (some mitigation with the construction of green bridges, but these are unlikely to have a population-wide impact). Between 1998 and 2007 there was a 6.1% decline in hedgerows and 1.7% decline in total woody linear features in GB, with likely negative effects on connectivity.
	23% - 42% of designated habitats in 'favourable' condition (note this does not apply to non-designated areas and so only applies to a small proportion of available habitat). Main issue is overgrazing. Livestock numbers declining, but intensity of grazing and use of external inputs in remaining areas may be increasing (Samson 1999). Increase in pests / pathogens expected. Decline in boundary features may have negative impact on foraging opportunities.	
	Loss of hedgerows mainly owing to neglect. Agri-environment schemes have delivered some local improvements, but these not reflected in national surveys. Decline in structural quality and associated vegetation diversity. Countryside Survey 2007 notes decline in structural quality and associated vegetation diversity. Trend of frequent cutting and flailing reduces availability of foraging resources.	
	Urban expansion may reduce roosting opportunities. Also may affect availability of foraging areas and suitability of roosts at current urban-rural interface.	

Species	Climate change (from National Ecosystem Assessment)	Change in population size (1995 - 2016)	Future prospects rating	Explanatory notes
Grey long-eared bat	Overwinter survival and recruitment may be affected by climate change. Directions of impacts unclear. Cold wet springs likely to have negative effect. Reduced tree growth in the south, and increases in north and west. Pests / disease issues affecting habitat likely to increase.	DD	Population - Decline Range - Unknown Habitat - Decline	It is considered that the population is likely to continue to decline in the future based on past trajectory and declining habitat quality.

(a) Information taken from Countryside Survey tables 2.2; 3.2; 4.1; 4.2 (Carey et al., 2008); except for unimproved grassland which is taken from Khan (2015). For Improved grassland, the percentage change was calculated for improved neutral, calcareous and acid grassland combined for consistency with main Review.

(b) Because bats use the landscape on a broad spatial scale, assessments are made across all habitat types combined.

References are found within the main review or refer to the Millenium Ecosystem Assessment with the following additions:

Forestry Commission (2017). *Forestry Statistics 2017*. Forest Research, Edinburgh.

Forestry Commission (2016). *Preliminary estimates of the changes in canopy cover in British woodlands between 2006 and 2015*. National Forest Inventory, Forest Research, Edinburgh.

Fox, R., Parsons, M.S., Chapman, J.W., Woiwod, I.P., Warren, M.S. & Brooks, D.R. (2013) *The State of Britain's Larger Moths 2013*. Butterfly Conservation Trust and Rothamsted Research, Wareham, Dorset, UK.

Khan, J., Powell, T., Harwood, A. (2015) *Land Use in the UK*. A report for the Office of National Statistics and the University of East Anglia.

Kirby, K.J., Reid, C.M., Thomas, R.C. and Goldsmith, F.B. 1997. Preliminary estimates of fallen dead wood and standing dead trees in managed and unmanaged forests in Britain. *Journal of Applied Ecology* 35: 148–155.

References:

AARS, J., LAMBIN, X., DENNY, R. & GRIFFIN, A. C. 2001. Water vole in the Scottish uplands: distribution patterns of disturbed and pristine populations ahead and behind the American mink invasion front. *Animal Conservation,* 4(3), 187-194.

ABBOTT, I. M., SLEEMAN, D. P. & HARRISON, S. 2009. Bat activity affected by sewage effluent in Irish rivers. *Biological Conservation,* 142(12), 2904-2914.

AEBISCHER, N. J., DAVEY, P. D. & KINGDON, N. G. 2011. *National Gamebag Census: Mammal Trends to 2009*. Fordingbridge: Game & Wildlife Conservation Trust.

AEGERTER, J. N. 2003. *Maximising the biodiversity value of farm woodlands to the agri-environment,* London, HMSO.

ALTRINGHAM, J. & SENIOR, P. 2005. Social systems and ecology of bats. *In:* RUCKSTUHL, K. E. & NEUHAUS, P. (eds.) *Sexual Segregation in Vertebrates: Ecology of the Two Sexes.* New York: Cambridge University Press.

AMAR, A., REDPATH, S. & THIRGOOD, S. 2003. Evidence for food limitation in the declining hen harrier population on the Orkney Islands, Scotland. *Biological Conservation,* 111(3), 377-384.

AMSTRUP, S. C., MCDONALD, T. & MANLY, B. F. J. 2005. *Handbook of capture-recapture analyses,* Princeton and Oxford, Princeton University Press.

ANCILLOTTO, L., CISTRONE, L., MOSCONI, F., JONES, G., BOITANI, L. & RUSSO, D. 2015. The importance of non-forest landscapes for the conservation of forest bats: lessons from barbastelles (*Barbastella barbastellus*). *Biodiversity and Conservation,* 24(1), 171-185.

ANDERSON, M. E. & RACEY, P. A. 1991. Feeding behaviour of captive brown long-eared bats, *Plecotus auritus*. *Animal Behaviour,* 42(-), 489-493.

ANDERSON, M. E. & RACEY, P. A. 1993. Discrimination between fluttering and non-fluttering moths by brown long-eared bats, *Plecotus auritus*. *Animal Behaviour,* 46(6), 1151-1155.

ANDERSON, P. & YALDEN, D. W. 1981. Increased sheep numbers and the loss of heather moorland in the Peak District, England. *Biological Conservation,* 20(3), 195-213.

ANGELL, R. L., BUTLIN, R. K. & ALTRINGHAM, J. D. 2013. Sexual Segregation and Flexible Mating Patterns in Temperate Bats. *PLOS ONE,* 8(1), e54194.

ARLETTAZ, R. 1996. Feeding behaviour and foraging strategy of free-living mouse-eared bats, *Myotis myotis* and *Myotis blythii*. *Animal Behaviour,* 51(1), 1-11.

ARMITAGE, P., WEST, B. & STEEDMAN, K. 1984. New evidence of black rat in Roman London. *The London Archaeologist,* 4(14), 375-383.

ARNOLD, H. R. 1993. *Atlas of Mammals in Britain,* London, Joint Nature Conservation Committee / Institute of Terrestrial Ecology. HMSO.

ASKEW, N. P., SEARLE, J. B. & MOORE, N. P. 2007. Agri-environment schemes and foraging of barn owls *Tyto alba*. *Agriculture, Ecosystems & Environment,* 118(1-4), 109-114.

ATTERBY, H., AEGERTER, J. N., SMITH, G. C., CONYERS, C. M., ALLNUTT, T. R., RUEDI, M. & MACNICOLL, A. D. 2010. Population genetic structure of the Daubenton's bat (*Myotis daubentonii*) in western Europe and the associated occurrence of rabies. *European Journal of Wildlife Research,* 56(1), 67-81.

ATTUQUAYEFIO, D. K., GORMAN, M. L. & WOLTON, R. J. 1986. Home range sizes in the Wood mouse *Apodemus sylvaticus*: habitat, sex and seasonal differences. *Journal of Zoology (London),* 210(1), 45-53.

AUGUST, T. A., NUNN, M. A., FENSOME, A. G., LINTON, D. M. & MATHEWS, F. 2014. Sympatric woodland *Myotis* bats form tight-knit social groups with exclusive roost home ranges. *PLOS ONE,* 9(10), p.e112225.

BAKER, P. J. & HARRIS, S. 2007. Urban mammals: what does the future hold? An analysis of the factors affecting patterns of use of residential gardens in Great Britain. *Mammal Review,* 37(4), 297-315.

BAKKER, E. S., REIFFERS, R. C., OLFF, H. & GLEICHMAN, J. M. 2005. Experimental manipulation of predation risk and food quality: effect on grazing behaviour in a central-place foraging herbivore. *Oecologia,* 146(1), 157-167.

BALHARRY, D. 1993. *Factors affecting the distribution and population density of pine martens (Martes martes l.) in Scotland.* PhD, University of Aberdeen.

BARLOW, K. & BRIGGS, P. 2012. *Grey long-eared bat surveillance 2012.* JNCC Report No. 478.

BARLOW, K., HARGREAVES, D. & MATHEWS, F. 2016. *Understanding the ecology, current status and conservation threats for Nathusius' pipistrelle in Great Britain - a pilot study. Final Report to the People's Trust for Endangered Species.* People's Trust for Endangered Species.

BARLOW, K. E. 1997. The diets of two phonic types of the bat *Pipistrellus pipistrellus* in Britain. *Journal of Zoology,* 243(3), 597-609.

BARLOW, K. E. & JONES, G. 1999. Roosts, echolocation calls and wing morphology of two phonic types of *Pipistrellus pipistrellus*. *Zeitschrift für Säugetierkunde,* 64(5), 257-268.

BARRATT, E. M., DEAVILLE, R., BURLAND, T. M., BRUFORD, M. W., JONES, G., RACEY, P. A. & WAYNE, R. K. 1997. DNA answers the call of pipistrelle bat species. *Nature,* 387(6629), 138-139.

BARRETO, G. R. & MACDONALD, D. W. 2000. The decline and local extinction of a population of water voles, *Arvicola terrestris*, in southern England. *Zeitschrift Fur Saugetierkunde-International Journal of Mammalian Biology,* 65(2), 110-120.

BARRETO, G. R., MACDONALD, D. W. & STRACHAN, R. 1998a. The tightrope hypothesis:an explanation for plummeting water vole numbers in the Thames catchment. *In:* SHERWOOD, B. & BAILEY, R. (eds.) *United Kingdom Floodplains.*

BARRETO, G. R., RUSHTON, S. P., STRACHAN, R. & MACDONALD, D. W. 1998b. The role of habitat and mink predation in determining the status and distribution of water voles in England. *Animal Conservation,* 1(2), 129-137.

BAT CONSERVATION TRUST 2016. The National Bat Monitoring Programme Annual Report 2015. London. Available at http://www.bats.org.uk/pages/nbmp_annual_report.html: Bat Conservation Trust.

BATTERSBY, J. 1999. *A comparison of the roost ecology of the brown long-eared bat Plecotus auritus and the serotine bat Eptesicus serotinus.* PhD, University of Sussex.

BATTERSBY, J. 2005. *UK mammals: species status and population trends. First report by tracking mammals partnership*. Peterborough: Joint Nature Conservation Committee, Tracking Mammals Partnership.

BATTERSBY, J. E. & GREENWOOD, J. J. D. 2004. Monitoring terrestrial mammals in the UK: past, present and future, using lessons from the bird world. *Mammal Review,* 34(1-2), 3-29.

BAUEROVA, Z. 1982. Contribution to the trophic ecology of the grey long-eared bat, *Plecotus austriacus*. *Folia Zoologica,* 31(2), 113-122.

BEAUMONT, M., BARRATT, E. M., GOTTELLI, D., KITCHENER, A. C., DANIELS, M. J., PRITCHARD, J. K. & BRUFORD, M. W. 2001. Genetic diversity and introgression in the Scottish wildcat. *Molecular Ecology,* (10), 319-336.

BELLAMY, P. E., SHORE, R. F., ARDESHIR, D., TREWEEK, J. R. & SPARKS, T. H. 2000. Road verges as habitat for small mammals in Britain. *Mammal Review,* 30(2), 131-139.

BENTLEY, E. 1959. The distribution and status of *Rattus rattus L.* in the United Kingdom in 1951 and 1956. *Journal of Animal Ecology,* 28(2), 299-308.

BENTLEY, E. 1964. A further loss of ground by *Rattus rattus L*. in the United Kingdom during 1956-61. *Journal of Animal Ecology,* 33(2)**,** 371-373.

BERGE, L. 2007. *Resource partitioning between the cryptic species Brandt's bat (Myotis brandtii) and the whiskered bat (M. mystacinus) in the UK.* PhD, University of Bristol.

BERGER, K. M. 2006. Carnivore-livestock conflicts: Effects of subsidized predator control and economic correlates on the sheep industry. *Conservation Biology,* 20(3)**,** 751-761.

BERRY, R. J. 1996. Small mammal differentiation on islands. *Philos Trans R Soc Lond B Biol Sci,* 351(1341)**,** 753-64.

BERRY, R. J. & TRICKER, B. J. K. 1969. Competition and extinction: the mice of Foula, with notes on those of Fair Isle and St. Kilda. *Journal of Zoology,* 158(2)**,** 247-265.

BERTHIER, K., PIRY, S., COSSON, J. F., GIRAUDOUX, P., FOLTÊTE, J. C., DEFAUT, R., TRUCHETET, D. & LAMBIN, X. 2014. Dispersal, landscape and travelling waves in cyclic vole populations. *Ecology Letters,* 17(1)**,** 53-64.

BILLINGTON, G. & RAWLINGSON, M. D. 2006. *A review of horseshoe bats flight lines and feeding areas*. *CCW Science Report No. 755.* Bangor: CCW.

BIRKS, J. 1999. *Ecology of the polecat in lowland England*. *In:* BIRKS, J. & KITCHENER, A. C. (eds.) *The distribution and status of the polecat (Mustela putorius) in Britain in the 1990s.* Ledbury: Vincent Wildlife Trust.

BIRKS, J. D. S. 2015. *Polecats*, Whittet Books Ltd.

BIRKS, J. D. S. & KITCHENER, A. 1999. *The distribution and status of the polecat (Mustela putorius) in Britain in the 1990s,* London, The Vincent Wildlife Trust.

BIRKS, J. D. S. & KITCHENER, A. C. 2008. Carnivores: Order Carnivora. *In:* HARRIS, S. & YALDEN, D. (eds.) *Mammals of the British Isles: Handbook.*

BLACKMORE, M. 1956. An occurrence of the mouse-eared bat, *Myotis myotis*, (Borkhausen) in England. *Journal of Zoology,* 127(2)**,** 201-203.

BLANDFORD, P. 1986. *Behavioural ecology of the polecat Mustela putorius in Wales.* University of Exeter.

BOATMAN, N. D., PARRY, H. R., BISHOP, J. D. & CUTHBERTSON, A. G. S. 2007. Impacts of Agricultural Change on Farmland Biodiversity in the UK. *Biodiversity Under Threat.* The Royal Society of Chemistry.

BOGDANOWICZ, W. 2004. Pipistrellus kuhlii (Kuhl, 1817) - Weissrandfledermaus, Band 4, Fledertiere, Tiel II: Chiroptera II: Vespertilionidae 2, Molossidae, Nycteridae. *In:* KRAPP, F. (ed.) *Handbuch der Säugetiere Europas.* Wiesbaden (In German): AULA-Verlag.

BONESI, L. & MACDONALD, D. W. 2004. Differential habitat use promotes sustainable coexistence between the specialist otter and the generalist mink. *Oikos,* 106(3)**,** 509-519.

BONESI, L., STRACHAN, R. & MACDONALD, D. W. 2006. Why are there fewer signs of mink in England? Considering multiple hypotheses. *Biological Conservation,* 130(2)**,** 268-277.

BONNINGTON, C., GASTON, K. & EVANS, K. 2014. Squirrels in suburbia: influence of urbanisation on the occurrence and distribution of a common exotic mammal. *Urban Ecosystems,* 17(2)**,** 533-546.

BONTADINA, F., SCHOFIELD, H. & NAEF-DAENZER, B. 2002. Radio-tracking reveals that lesser horseshoe bats (*Rhinolophus hipposideros*) forage in woodland. *Journal of Zoology,* 258(3)**,** 281-290.

BOONMAN, A. M., BOONMAN, M., BRETSCHNEIDER, F. & VAN DE GRIND, W. A. 1998. Prey Detection in Trawling Insectivorous Bats: Duckweed Affects Hunting Behaviour in Daubenton's Bat, *Myotis daubentonii*. *Behavioral Ecology and Sociobiology,* 44(2)**,** 99-107.

BORCHERS, D. L., STEVENSON, B. C., KIDNEY, D., THOMAS, L. & MARQUES, T. A. 2015. A Unifying Model for Capture–Recapture and Distance Sampling Surveys of Wildlife Populations. *Journal of the American Statistical Association,* 110(509)**,** 195-204.

BOSTON, E. S. M., IAN MONTGOMERY, W., HYNES, R. & PRODÖHL, P. A. 2015. New insights on postglacial colonization in western Europe: the phylogeography of the

Leisler's bat (*Nyctalus leisleri*). *Proceedings of the Royal Society B: Biological Sciences,* 282(1804), 20142605.

BOUGHEY, K. L., LAKE, I. R., HAYSOM, K. A. & DOLMAN, P. M. 2011. Effects of landscape-scale broadleaved woodland configuration and extent on roost location for six bat species across the UK. *Biological Conservation,* 144(9), 2300-2310.

BOYD, I. L. & STEBBINGS, R. E. 1989. Population Changes of Brown Long-Eared Bats (*Plecotus auritus*) in Bat Boxes at Thetford Forest. *Journal of Applied Ecology,* 26(1), 101-112.

BOYE, P. & DIETZ, M. 2005. *Development of good practice guidelines for woodland management for bats*. *Research Report No 661.* Peterborough: English Nature.

BRADSHAW, E. L. 1993. *Social and ecological determinants of food availability in the brown hare, Lepus europaeus Pallas.* DPhil DPhil Thesis, University of Oxford.

BRIGGS, P. 2000. *A Study of Barn Conversions in Hertfordshire Commissioned by Hertfordshire BRC and Hertfordshire County Council.*

BRIGHT, P. & MACPHERSON, D. 2002. *Hedgerow management, dormice and biodiversity. Research Report no 454,* Peterborough, Natural England.

BRIGHT, P. & MORRIS, P. 2005. *The Dormouse,* London, The Mammal Society.

BRIGHT, P., MORRIS, P. & MITCHELL-JONES, T. 2006. *The dormouse conservation handbook*, English Nature.

BRIGHT, P. W., MITCHELL, P. & MORRIS, P. A. 1994. Dormouse Distribution: Survey Techniques, Insular Ecology and Selection of Sites for Conservation. *Journal of Applied Ecology,* 31(2), 329-339.

BRIGHT, P. W. & MORRIS, P. A. 1996. Why are dormice rare? A case study in conservation biology. *Mammal Review,* 26(4), 157-187.

BRIGHT, P. W., MORRIS, P. A. & MITCHELL-JONES, A. J. 1996. A new survey of the dormouse *Muscardinus avellanarius* in Britain, 1993-4. *Mammal Review,* 26(4), 189-195.

BRIGHT, P. W. & SMITHSON, T. J. 1997. *Species Recovery Programme for the pine marten in England: 1995-96. Research Report no 240,* Peterborough, English Nature.

BRIGHT, P. W. & SMITHSON, T. J. 2001. Biological invasions provide a framework for reintroductions: selecting areas in England for pine marten releases. *Biodiversity and Conservation,* 10(8), 1247-1265.

BROUGHTON, R. K., SHORE, R. F., HEARD, M. S., AMY, S. R., MEEK, W. R., REDHEAD, J. W., TURK, A. & PYWELL, R. F. 2014. Agri-environment scheme enhances small mammal diversity and abundance at the farm-scale. *Agriculture Ecosystems & Environment,* 192(1st July), 122-129.

BROWN, P. 2016. *The cryptic group of small Myotis bats (M. mystacinus, M. brandtii and M. alcathoe) and habitat use by woodland bats species in Britain.* MSc Thesis, University of Bristol.

BRYCE, J., CARTMEL, S. & QUINE, C. 2005. Habitat use by red and grey squirrels: results of two recent studies and implications for management *Forestry Commission Information Note 76. .* Edinburgh: Forestry Commission.

BRYCE, R., OLIVER, M. K., DAVIES, L., GRAY, H., URQUHART, J. & LAMBIN, X. 2011. Turning back the tide of American mink invasion at an unprecedented scale through community participation and adaptive management. *Biological Conservation,* 144(1), 575-583.

BUCKLE, A. 2013. Anticoagulant resistance in the United Kingdom and a new guideline for the management of resistant infestations of Norway rats (*Rattus norvegicus,* Berk.). *Pest Management Science,* 69(3), 334-341.

BUNCE, R. G. H., BARR, C. J., CLARKE, R. T., HOWARD, D. C. & LANE, A. M. J. 1996. ITE Merlewood Land Classification of Great Britain. *Journal of Biogeography,* 23(5), 625-634.

BUNCE, R. G. H., BARR, C. J. & WHITTAKER, H. A. 1981a. *An integrated system of land classification*, Annual Report of the Institute of Terrestrial Ecology.

BUNCE, R. G. H., BARR, C. J. & WHITTAKER, H. A. 1981b. Land classes in Great Britain: preliminary descriptions for users of the Merlewood method of land classification. *Merlewood research and development paper No. 86.* Grange-over-Sands: Institute of Terrestrial Ecology.

BURGESS, M., MORRIS, P. & BRIGHT, P. 2003. Population Dynamics of the Edible Dormouse (*Glis glis*) in England. *Acta Zoologica Academiae Scientiarum Hungaricae,* 49(1)**,** 27-31.

BURLAND, T., BARRATT, E., BEAUMONT, M. & RACEY, P. 1999. Population genetic structure and gene flow in a gleaning bat, *Plecotus auritus*. *Proceedings of the Royal Society of London B: Biological Sciences,* 266(1422)**,** 975-980.

BURLAND, T., BARRATT, E., NICHOLS, R. & RACEY, P. A. 2001. Mating patterns, relatedness and the basis of natal philopatry in the brown long-eared bat, *Plecotus auritus*. *Molecular Ecology,* 10(5)**,** 1309-1321.

BURTON, M. 1969. *The hedgehog*, André Deutsch, London.

CAMPBELL, R. D., HARRINGTON, A., ROSS, A. & HARRINGTON, L. 2012. *Distribution, population assessment and activities of beavers in Tayside: Scottish Natural Heritage Commissioned Report,* Edinburgh, Scottish Natural Heritage.

CAMPBELL, S. & HARTLEY, G. Wild boar distribution in Scotland. Poster Presentation, 8th International Symposium on Wild Boar and Other Suids, 2010.

CANTY, A. & RIPLEY, B. 2012. boot: Bootstrap R (S-Plus) functions: R package version.

CAPIZZI, D. & LUISELLI, L. 1996. Ecological relationship between small mammals and age of coppice in a oak-mixed forest in central Italy. *Revue d Ecologie,* 51(3)**,** 227-91.

CAPREOLUS WILDLIFE CONSULTANCY 2005. *The ecology and conservation of water voles in upland habitats: Scottish Natural Heritage Commissioned Report 99,* Edinburgh, Scottish Natural Heritage.

CAREY, P. D., WALLIS, S., CHAMBERLAIN, P. M., COOPER, A., EMMETT, B. A., MASKELL, L. C., MCCANN, T., MURPHY, J., NORTON, L. R., REYNOLDS, B., SCOTT, W. A., SIMPSON, I. C., SMART, S. M. & ULLYETT, J. M. 2008. Countryside Survey: UK Results from 2007. NERC Centre for Ecology & Hydrology. *CEH Project Number: C03259.*

CARTER, P. & CHURCHFIELD, S. 2006. *The Water Shrew Handbook,* London, The Mammal Society.

CARTMEL, S. 2000. *Squirrel ecology in a conifer forest in North Wales.* PhD PhD Thesis, Queen Mary, Universty of London.

CARYL, F. M. 2008. *Pine marten diet and habitat use within a managed coniferous forest.* PhD PhD Thesis, University of Stirling.

CARYL, F. M., QUINE, C. P. & PARK, K. J. 2012. Martens in the matrix: the importance of nonforested habitats for forest carnivores in fragmented landscapes. *Journal of Mammalogy,* 93(2)**,** 464-474.

CATTO, C. 1993. *Aspects of ecology and behaviour of the serotine bat (Eptesicus serotinus).* PhD PhD Thesis, University of Aberdeen.

CATTO, C. M., HUTSON, A. M. & RACEY, P. A. 1994. The diet of *Eptesicus serotinus* in southern England. *Folia Zoologica,* 43(4)**,** 307-14.

CATTO, C. M. C., HUTSON, A. M., RACEY, P. A. & STEPHENSON, P. J. 1996. Foraging behaviour and habitat use of the serotine bat (*Eptesicus serotinus*) in southern England. *Journal of Zoology,* 238(4)**,** 623-633.

CHADWICK, H. A., HODGE, J. S. & RATCLIFFE, P. R. 1997. Foxes and Forestry. *Forestry Commission Technical papers 23.* Forestry Commission.

CHANIN, P. 2003. *Monitoring the Otter Lutra lutra. Conserving Natura 2000 Rivers Monitoring Series No. 10,* Peterborough, English Nature.

CHANIN, P. & GUBERT, L. 2011. Surveying hazel dormice (*Muscardinus avellanarius*) with tubes and boxes: a comparison. *Mammal Notes,* 4(-)**,** 1-6.

CHANIN, P. & WOODS, M. 2003. *Surveying dormice using nest tubes: Research Report 524*. Peterborough.

CHANNON, D., CHANNON, E., ROBERTS, T. & HAINES, R. 2006. Hotspots: are some areas of sewer network prone to re-infestation by rats (*Rattus norvegicus*) year after year? *Epidemiology and Infection,* 134(1)**,** 41-48.
CHANTREY, J., DALE, T. D., READ, J. M., WHITE, S., WHITFIELD, F., JONES, D., MCINNES, C. J. & BEGON, M. 2014. European red squirrel population dynamics driven by squirrelpox at a gray squirrel invasion interface. *Ecology and Evolution,* 4(19)**,** 3788-3799.
CHAPMAN, N., HARRIS, S. & STANFORD, A. 1994. Reeves' Muntjac *Muntiacus reevesi* in Britain: their history, spread, habitat selection, and the role of human intervention in accelerating their dispersal. *Mammal Review,* 24(3)**,** 113-160.
CHAUVENET, A. L., HUTSON, A. M., SMITH, G. C. & AEGERTER, J. N. 2014 Demographic variation in the UK serotine bat: filling gaps in knowledge for management. *Ecology and Evolution,* 4(19)**,** 3820-9.
CHURCHFIELD, S., HOLLIER, J. & BROWN, V. K. 1995. Population dynamics and survivorship patterns in the common shrew *Sorex araneus* in southern Scotland. *Acta Theriologica,* 40(1)**,** 53-68.
CLAPHAM, S. J. 2011. *The abundance and diversity of small mammals and birds in mature crops of the perennial grasses Miscanthus x giganteus and Phalaris arundinacea grown for biomass energy.* PhD, Cardiff University.
CLARKE, G. P., WHITE, P. C. & HARRIS, S. 1998. Effects of roads on badger *Meles meles* populations in south-west England. *Biological Conservation,* 86(2)**,** 117-124.
CLUTTON-BROCK, J. & MACGREGOR, A. 1988. An end to medieval reindeer in Scotland. *Proceedings of the Royal Society of Antiquaries of Scotland,* 118(23-35.
CLUTTON-BROCK, T. H. & ALBON, S. D. 1989. *Red deer in the Highlands,* Oxford, BSP Professional Books.
CLUTTON-BROCK, T. H. & PEMBERTON, J. M. 2004. *Soay sheep: dynamics and selection in an island population*, Cambridge University Press.
COLLEN, B., PETTORELLI, N., BAILLIE, J. E. M. & DURANT, S. M. 2013. *Biodiversity monitoring and conservation: bridging the gap between global commitment and local action*, Wiley-Blackwell / Zoological Society of London.
COLLIN, P. N. 1995. Leisler's bats from Galloway. *Scottish Bats,* 3(8)**,** -.
CONNORS, E. 2016. *UK Natural Capital: Ecosystem accounts for farmland (Experimental Statistics),* London, Office for National Statistics.
CONRAD, K. F., PERRY, J. N., WOIWOD, I. P. & ALEXANDER, C. J. 2006a. Large-scale temporal changes in spatial pattern during declines of abundance and occupancy in a common moth. *Journal of Insect Conservation,* 10(1)**,** 53-64.
CONRAD, K. F., WARREN, M. S., FOX, R., PARSONS, M. S. & WOIWOD, I. P. 2006b. Rapid declines of common, widespread British moths provide evidence of an insect biodiversity crisis. *Biological Conservation,* 132(3)**,** 279-291.
COOKE, A. 2011. Deer in Huntingdonshire and the Soke of Peterborough: . *Huntingdonshire Fauna & Flora Society Report* 63(-)**,** 51-61.
CORBET, G. B. 1964. Regional variation in the bank vole *Clethriomimys glareolus* in the British Isles. *Proceedings of the Zoological Society of London,* 143(2)**,** 191-219.
COSTA, M., FERNANDES, C., BIRKS, J. D., KITCHENER, A. C., SANTOS-REIS, M. & BRUFORD, M. W. 2013. The genetic legacy of the 19th-century decline of the British polecat: evidence for extensive introgression from feral ferrets. *Molecular Ecology,* 22(20)**,** 5130-47.
COWAN, D. P. 1991. Rabbit *Oryctolagus cuniculus*. *In:* CORBET, G. B. & HARRIS, S. (eds.) *Mammals of the British Isles: Handbook (4th Edn), pp201-210.* Southampton: Mammal Society.
CRAWFORD, A. 2003. *Fourth Otter Survey of England 2000-2002*. Bristol: Environment Agency.
CRAWFORD, A. 2010. Fifth Otter Survey of England 2009–2010. Bristol: Environment Agency.

CRESSWELL, P., HARRIS, S. & JEFFERIES, D. J. 1990. *The history, distribution, status and habitat requirements of the badger in Britain,* Peterborough, Nature Conservancy Council.

CROFT, S., CHAUVENET, A. L. & SMITH, G. C. 2017. A systematic approach to estimate the distribution and total abundance of British mammals. *PloS one,* 12(6), e0176339.

CROIN-MICHIELSEN, N. 1966. Intraspecific and interspecific competition in the shrews *Sorex araneus* L. and *S. minutus* L. *Archives Néelandaises de Zoologie,* 17(-), 73-174.

CROOSE, E. 2016. *The distribution and status of the polecat (Mustela putorius) in Britain 2014-2015,* Ledbury, The Vincent Wildlife Trust.

CROOSE, E., BIRKS, J. D. S., O'REILLY, C., TURNER, P., MARTIN, J. & MACLEOD, E. T. 2015. Sample diversity adds value to non-invasive genetic assessment of a pine marten (*Martes martes*) population in Galloway Forest, southwest Scotland. *Mammal Research,* 61(2), 131-139.

CROOSE, E., BIRKS, J. D. S. & SCHOFIELD, H. W. 2013. *Expansion zone survey of pine marten (Martes martes) distribution in Scotland: Scottish Natural Heritage Commissioned Report no 520,* Edinburgh, Scottish Natural Heritage.

CROOSE, E., BIRKS, J. D. S., SCHOFIELD, H. W. & O'REILLY, C. 2014. *Distribution of the pine marten (Martes martes) in southern Scotland in 2013: Scottish Natural Heritage Commissioned Report no 740,* Edinburgh, Scottish Natural Heritage.

DAAN, S. 1973. Activity during natural hibernation in three species of Vespertilionid bats. *Netherlands Journal of Zoology,* 23(-), 1-71.

DALLAS, J. F., COXON, K. E., SYKES, T., CHANIN, P. R., MARSHALL, F., CARSS, D. N., BACON, P. J., PIERTNEY, S. B. & RACEY, P. A. 2003. Similar estimates of population genetic composition and sex ratio derived from carcasses and faeces of Eurasian otter *Lutra lutra*. *Molecular Ecology,* 12(1), 275-282.

DANIELS, M. J. 2006. Estimating red deer *Cervus elaphus* populations: an analysis of variation and cost-effectiveness of counting methods. *Mammal Review,* 36(3), 235-247.

DANILKIN, A. 1996. *Behavioural ecology of Siberian and European roe deer,* Netherlands, Chapman and Hall.

DANKO, Š., KRIŠTÍN, A. & KRIŠTOFÍK, J. 2010. Myotis alcathoe in eastern Slovakia: occurrence, diet, ectoparasites and notes on its identification in the field. *Vespertilio,* 13(14), 77-91.

DAVIDSON-WATTS, I. & JONES, G. 2006. Differences in foraging behaviour between *Pipistrellus pipistrellus* (Schreber, 1774) and *Pipistrellus pygmaeus* (Leach, 1825). *Journal of Zoology,* 268(1), 55-62.

DAWO, B., KALKO, E. K. V. & DIETZ, M. 2013. Spatial Organization Reflects the Social Organization in Bechstein's Bats. *Annales Zoologici Fennici,* 50(6), 356-370.

DEBROT, S. & MERMOD, C. 1983. The spatial and temporal distribution pattern of the stoat (*Mustela erminea* L.). *Oecologia,* 59(1), 69-73.

DEFRA 2008. *Feral wild boar in England: An action plan*. London: Department for Environment, Food and Rural Affairs.

DELAHAY, R., DAVISON, J., POOLE, D., MATTHEWS, A., WILSON, C., HEYDON, M. & ROPER, T. 2009. Managing conflict between humans and wildlife: trends in licensed operations to resolve problems with badgers *Meles meles* in England. *Mammal Review,* 39(1), 53-66.

DEPARTMENT FOR COMMUNITIES AND LOCAL GOVERNMENT 2015. English House Condition Survey, 2007. 3rd edition ed.: UK Data Service.

DICKINSON, J. L., ZUCKERBERG, B. & BONTER, D. N. 2010. Citizen Science as an Ecological Research Tool: Challenges and Benefits. *Annual Review of Ecology, Evolution, and Systematics,* 41(1), 149-172.

DICKMAN, C. R. 1986. Habitat utilization and diet of the harvest mouse, *Micromys minutus*, in an urban environment. *Acta Theriologica,* 31(19), 249-256.

DIETZ, C. & KEIFER, A. 2016. *Bats of Britain and Europe,* London, Bloomsbury.

DIETZ, M. & PIR, J. 2009. Distribution and habitat selection of *Myotis bechsteinii* in Luxembourg: implications for forest management and conservation. *Folia Zoologica,* 58(-), 327-340.
DIETZ, M. & PIR, J. 2011. *Distribution, Ecology and Habitat Selection by Bechstein's Bat (Myotis bechsteinii) in Luxembourg,* Luxembourg, Laurenti.
DONCASTER, C., DICKMAN, C. & MACDONALD, D. 1990. Feeding ecology of red foxes (*Vulpes vulpes*) in the city of Oxford, England. *Journal of Mammalogy,* 71(2), 188-194.
DONCASTER, C. P. 1994. Factors regulating local variations in abundance: field tests on hedgehogs, *Erinaceus europaeus*. *Oikos,* 69(2), 182-192.
DONCASTER, C. P. & MACDONALD, D. W. 1991. Drifting territoriality in the red fox *Vulpes vulpes*. *The Journal of Animal Ecology,* 60(2), 423-439.
DONDINI, G., RUTKOWSKI, T., VERGARI, S. & WOJTASZYN, G. 2013. Long distance migration of female Leisler's bat (*Nyctalus leisleri*) from Italy to Poland. *2013,* 23(2), 2.
DOWNS, N. C. & SANDERSON, L. J. 2010. Do Bats Forage Over Cattle Dung or Over Cattle? *Acta Chiropterologica,* 12(2), 349-358.
DUNSTONE, N. & BIRKS, J. D. S. 1985. Comparative ecology of coastal, riverine and lacustrine mink *Mustela vison* in Britain. *Zeitschrift für Angewandte Zoologie,* 72(1), 59-70.
DUNSTONE, N. & MACDONALD, D. W. 2008. Carnivores: Order Carnivora. *In:* HARRIS, S. & YALDEN, D. (eds.) *Mammals of the British Isles: Handbook.* Southampton: The Mammal Society.
DURRANT, C. J., BEEBEE, T. J. C., GREENAWAY, F. & HILL, D. A. 2009. Evidence of recent population bottlenecks and inbreeding in British populations of Bechstein's bat, *Myotis bechsteinii*. *Conservation Genetics,* 10(2), 489-496.
DUTTON, J. S., CLAYTON, H. T. & EVANS, S. M. 2015. The social aspects of wild boar in the Forest Of Dean. Unpublished report for the Forestry Commission by the University of Worcester.
DUVERGE, P. L. & JONES, G. 1994. Greater horseshoe bats - activity, foraging behaviour and habitat use. *British Wildlife,* 6(-), 69-77.
EASTERBEE, N., HEPBURN, L. V. & JEFFERIES, D. J. 1991. *Survey of the status and distribution of the wildcat in Scotland, 1983-1987*. Nature Conservancy Council for Scotland.
EDWARDS, C. A. & LOFTY, J. R. 1972. *Biology of Earthworms*, Springer US.
EDWARDS, T. & KENYON, W. 2013. Wild Deer in Scotland. *SPICe Briefing.* The Scottish Parliament: SPICe The Information Centre.
EFFORD, M. 2004. Density estimation in live-trapping studies. *Oikos,* 106(3), 598-610.
EKLÖF, J. & JONES, G. 2003. Use of vision in prey detection by brown long-eared bats, *Plecotus auritus*. *Animal Behaviour,* 66(5), 949-953.
EKMAN, M. & JONG, J. D. 1996. Local patterns of distribution and resource utilization of four bat species (*Myotis brandti, Eptesicus nilssoni, Plecotus auritus* and *Pipistrellus pipistrellus*) in patchy and continuous environments. *Journal of Zoology,* 238(3), 571-580.
ENCARNAÇÃO, J. A. 2012. Mating at summer sites: indications from parentage analysis and roosting behaviour of Daubenton's bats (*Myotis daubentonii*). *Conservation Genetics,* 13(4), 1161-1165.
ENTWISTLE, A. C., RACEY, P. A. & SPEAKMAN, J. R. 1996. Habitat Exploitation by a Gleaning Bat, *Plecotus auritus*. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 351(1342), 921-931.
ENTWISTLE, A. C., RACEY, P. A. & SPEAKMAN, J. R. 2000. Social and population structure of a gleaning bat, *Plecotus auritus*. *Journal of Zoology,* 252(1), 11-17.
ERLINGE, S. 1977. Spacing strategy in stoat *Mustela erminea*. *Oikos,* 28(1), 32-42.
EVEREST, D. J., SHUTTLEWORTH, C. M., STIDWORTHY, M. F., GRIERSON, S. S., DUFF, J. P. & KENWARD, R. E. 2014. Adenovirus: an emerging factor in red squirrel *Sciurus vulgaris* conservation. *Mammal Review,* 44(3-4), 225-233.

FENSOME, A. G. & MATHEWS, F. 2016. Roads and bats: a meta-analysis and review of the evidence on vehicle collisions and barrier effects. *Mammal Review,* 46(4), 311-323.

FERNANDO, A. S. F., EVANS, P. R. & DUNSTONE, N. 1994. Local Variation in Rodent Communities of Sitka Spruce Plantations: The Interplay of Successional Stage and Site-Specific Habitat Parameters. *Ecography,* 17(4), 305-313.

FEZZI, C., BATEMAN, I., ASKEW, T., MUNDAY, P., PASCUAL, U., SEN, A. & HARWOOD, A. 2014. Valuing provisioning ecosystem services in agriculture: the impact of climate change on food production in the United Kingdom. *Environmental and Resource Economics,* 57(2), 197-214.

FINDLAY, M., ALEXANDER, L. & MACLEOD, C. 2015. Site condition monitoring for otters (*Lutra lutra*) in 2011-12: Scottish Natural Heritage Commissioned Report no 521. Edinburgh.

FLOWERDEW, J. R. 1985. The population dynamics of wood mice and yellow-necked mice. *Symposia of the Zoological Society of London,* 55(-), 315-338.

FLOWERDEW, J. R. & ELLWOOD, S. A. 2001. Impacts of woodland deer on small mammal ecology. *Forestry,* 74(3), 277-287.

FLOWERDEW, J. R., SHORE, R. F., POULTON, S. M. C. & SPARKS, T. H. 2004. Live trapping to monitor small mammals in Britain. *Mammal Review,* 34(1-2), 31-50.

FLUX, J. E. C. 1970. Life history of the Mountain hare (*Lepus timidus scoticus*) in north-east Scotland. *Journal of Zoology,* 161(1), 75-123.

FORESTRY COMMISSION 2014. 50-year forecast of softwood timber availability.

FRANKHAM, R. 2010. Challenges and opportunities of genetic approaches to biological conservation. *Biological Conservation,* 143(9), 1919-1927.

FRASER, E., LAMBIN, X., MCDONALD, R. & SM, R. 2015a. *Stoat (Mustela erminea) on the Orkney Islands–assessing the risks to native species. Scottish Natural Heritage Commissioned Report No 871.,* Edinburgh, Scottish Natural Heritage.

FRASER, E. J., LAMBIN, X., TRAVIS, J. M. J., HARRINGTON, L. A., PALMER, S. C. F., BOCEDI, G. & MACDONALD, D. W. 2015b. Range expansion of an invasive species through a heterogeneous landscape – the case of American mink in Scotland. *Diversity and Distributions,* 21(8), 888-900.

FUENTES-MONTEMAYOR, E., GOULSON, D., CAVIN, L., WALLACE, J. M. & PARK, K. J. 2013. Fragmented woodlands in agricultural landscapes: The influence of woodland character and landscape context on bats and their insect prey. *Agriculture, Ecosystems & Environment,* 172(1st June), 6-15.

FUNMILAYO, O. 1979. Food consumption, preferences and storage in the mole. *Acta Theriologica,* 24(27), 379-389.

FURMANKIEWICZ, J. 2008. Population size, catchment area, and sex-influenced differences in autumn and spring swarming of the brown long-eared bat (*Plecotus auritus*). *Canadian Journal of Zoology,* 86(3), 207-216.

FURMANKIEWICZ, J. & ALTRINGHAM, J. 2007. Genetic structure in a swarming brown long-eared bat (*Plecotus auritus*) population: evidence for mating at swarming sites. *Conservation Genetics,* 8(4), 913-923.

GARRATT, C. M., MINDERMAN, J. & WHITTINGHAM, M. J. 2012. Should we stay or should we go now? What happens to small mammals when grass is mown, and the implications for birds of prey. *Annales Zoologici Fennici,* 49(1-2), 113-122.

GAYWOOD, M., STRINGER, A., BLAKE, D., HALL, J., HENNESSY, M., TREE, A., GENNEY, D., MACDONALD, I., TONHASCA, A., BEAN, C., MCKINNELL, J., COHEN, S., RAYNOR, R., WATKINSON, P., BALE, D., TAYLOR, K., SCOTT, J. & BLYTH, S. 2015. *Beavers in Scotland: a report to the Scottish Government,* Edinburgh, Scottish Natural Heritage.

GAYWOOD, M. J. 2018. Reintroducing the Eurasian beaver Castor fiber to Scotland. *Mammal Review,* 48(1), 48-61.

GAYWOOD, M. J., BOON, P. J., THOMPSON, D. B. A. & STRACHAN, I. M. E. 2016. *The Species Action Framework Handbook,* Battleby, Perth., Scottish Natural Heritage.

GELLING, M., MACDONALD, D. W. & MATHEWS, F. 2007. Are hedgerows the route to increased farmland small mammal density? Use of hedgerows in British pastoral habitats. *Landscape Ecology,* 22(7), 1019-1032.

GERARD, J. F., CARGNELUTTI, B., SPITZ, F., VALET, G. & SARDIN, T. 1991. Habitat use of wild boar in a French agroecosystem from late winter to early summer. *Acta Theriologica,* 36(1-2), 119-129.

GIBBS, J. P., SNELL, H. L. & CAUSTON, C. E. 1999. Effective monitoring for adaptive wildlife management: Lessons from the Galapagos Islands. *Journal of Wildlife Management,* 63(4), 1055-1065.

GILL, R. & WAEBER, K. 2016. *Feral Wild Boar and Deer in the Forest of Dean - Survey and Population Projections in the Public Forest Estate 2016* Forest Research Report.

GILL, R. M. A. 2014. *Wild Boar and Deer in the Forest of Dean, Survey Results 2013*, Forest Research Report.

GILL, R. M. A. & FULLER, R. J. 2007. The effects of deer browsing on woodland structure and songbirds in lowland Britain. *Ibis,* 149(s2), 119-127.

GILL, R. M. A., JOHNSON, A. L., FRANCIS, A., HISCOCKS, K. & PEACE, A. J. 1996. Changes in roe deer (*Capreolus capreolus* L) population density in response to forest habitat succession. *Forest Ecology and Management,* 88(1-2), 31-41.

GILL, R. M. A. & MORGAN, G. 2009. The effects of varying deer density on natural regeneration in woodlands in lowland Britain. *Forestry,* 83(1), 53-63.

GLENDELL, M. & VAUGHAN, N. 2002. Foraging activity of bats in historic landscape parks in relation to habitat composition and park management. *Animal Conservation,* 5(4), 309-316.

GLOVER, A. M. & ALTRINGHAM, J. D. 2008. Cave selection and use by swarming bat species. *Biological Conservation,* 141(6), 1493-1504.

GOODMAN, S. J., BARTON, N. H., SWANSON, G., ABERNETHY, K. & PEMBERTON, J. M. 1999. Introgression Through Rare Hybridization: A Genetic Study of a Hybrid Zone Between Red and Sika Deer (*Genus Cervus*) in Argyll, Scotland. *Genetics,* 152(1), 355-371.

GOODMAN, S. J., TAMATE, H. B., WILSON, R., NAGATA, J., TATSUZAWA, S., SWANSON, G. M., PEMBERTON, J. M. & MCCULLOUGH, D. R. 2001. Bottlenecks, drift and differentiation: the population structure and demographic history of sika deer (*Cervus nippon*) in the Japanese archipelago. *Molecular Ecology,* 10(6), 1357-1370.

GOODWIN, C. E. D., HODGSON, D. J., AL-FULAIJ, N., BAILEY, S., LANGTON, S. & MCDONALD, R. A. 2017. Voluntary recording scheme reveals ongoing decline in the United Kingdom hazel dormouse *Muscardinus avellanarius* population. *Mammal Review,* 47(3), 183-197.

GORMAN, M. L. & AHMAD, Z. A. B. M. 1993. A comparative study of the ecology of woodmice *Apodemus sylvaticus* in two contrasting habitats: deciduous woodland and maritime sand-dunes. *Journal of Zoology (London),* 229(3), 385-396.

GORMAN, M. L. & REYNOLDS, P. 1993. The impact of land-use change on voles and raptors. *Mammal Review,* 23(3-4), 121-126.

GORMAN, M. L. & REYNOLDS, P. 2003. The impact of changing land-use on the Orkney vole. In. *Conservation and conflict: mammals and farming in Britain. Linnean Society Occasional Publications 4.* London: Linnean Society.

GORMAN, M. L. & STONE, R. D. 1990. *The natural history of moles*, Cornell University Press.

GORMLEY, A. M., FORSYTH, D. M., GRIFFIOEN, P., LINDEMAN, M., RAMSEY, D. S. L., SCROGGIE, M. P. & WOODFORD, L. 2011. Using presence-only and presence-absence data to estimate the current and potential distributions of established invasive species. *Journal of Applied Ecology,* 48(1), 25-34.

GORTÁZAR, C., DELAHAY, R. J., MCDONALD, R. A., BOADELLA, M., WILSON, G. J., GAVIER-WIDEN, D. & ACEVEDO, P. 2012. The status of tuberculosis in European wild mammals. *Mammal Review,* 42(3), 193-206.

GOSLING, L. M. & BAKER, S. J. 2008. Rodents: Order Rodentia. *In:* HARRIS, S. & YALDEN, D. (eds.) *Mammals of the British Isles: Handbook.* Southampton: The Mammal Society.

GOULDING, M. J., SMITH, G. & BAKER, S. J. 1998. Current status and potential impact of wild boar in the English countryside: a risk assessment. *Central Science Laboratory Report to the Ministry of Agriculture Fisheries and Food.* York, UK.
GOW, D. 2008. Water vole reintroduction projects - the lessons and the success factors. *ECOS,* 28(1)**,** 98.
GREEN, J. & GREEN, R. 1987. *Otter Survey of Scotland 1984-1985*. London: Vincent Wildlife Trust.
GREENAWAY, F. 2001. The Barbastelle in Britain. *British Wildlife,* 12(5)**,** 327-334.
GREENWOOD, A., CHURCHFIELD, S. & HICKEY, C. 2002. Geographical distribution and habitat occurrence of the Water Shrew (*Neomys fodiens*) in the Weald of South-East England. *Mammal Review,* 32(1)**,** 40-50.
GURNELL, J. 1983. Introduction to the symposium on the biology of squirrels. *Mammal Review,* 13(2-4)**,** 55-56.
GURNELL, J. 1996. The effects of food availability and winter weather on the dynamics of a grey squirrel population in southern England. *Journal of Applied Ecology,* 33(2)**,** 325-338.
GURNELL, J., BLACKETT, T., BUTLER, H., LURZ, P. W. W., MAGRIS, L. & SHUTTLEWORTH, C. M. 2015a. British red squirrel strongholds: challenges for conservation. . *In:* SHUTTLEWORTH, C., LURZ, P. & HAYWOOD, M. (eds.) *Red Squirrels: Ecology, Conservation & Management in Europe.* Woodbridge, Suffolk UK: European Squirrel Initiative.
GURNELL, J., GURNELL, A. M., DEMERITT, D., LURZ, P. W. W., SHIRLEY, M. D. F., RUSHTON, S. P., FAULKES, C. G., NOBERT, S. & HARE, E. J. 2008. *The feasibility and acceptability of reintroducing the European beaver to England,* Sheffield, UK, Natural England / People's Trust for Endangered Species.
GURNELL, J., HICKS, M. & WHITBREAD, S. 1992. The effects of coppice management on small mammal populations. *In:* BUCKLEY, G. P. (ed.) *Ecology and Management of Coppice Woodlands.* Dordrecht: Springer Netherlands.
GURNELL, J., LURZ, P. & BERTOLDI, W. 2014. The changing patterns in the distribution of red and grey squirrels in the North of England and Scotland between 1991 and 2010 based on volunteer surveys. *Hystrix-Italian Journal of Mammalogy,* 25(2)**,** 83-89.
GURNELL, J., LURZ, P. & WAUTERS, L. 2015b. Years of interactions and conflict in Europe: competition between Eurasian red squirrels and North American grey squirrels. *In:* SHUTTLEWORTH, C., LURZ, P. & HAYWARD, M. (eds.) *Red Squirrels: Ecology, Conservation & Management in Europe.* Woodbridge, Suffolk UK: European Squirrel Initiative.
GURNELL, J., LURZ, P. W. W., SHIRLEY, M. D. F., CARTMEL, S., GARSON, P. J., MAGRIS, L. & STEELE, J. 2004a. Monitoring red squirrels *Sciurus vulgaris* and grey squirrels *Sciurus carolinensis* in Britain. *Mammal Review,* 34(1-2)**,** 51-74.
GURNELL, J., WAUTERS, L. A., LURZ, P. W. W. & TOSI, G. 2004b. Alien species and interspecific competition: effects of introduced eastern grey squirrels on red squirrel population dynamics. *Journal of Animal Ecology,* 73(1)**,** 26-35.
HALLEY, D., ROSELL, F. & SAVELJEV, A. 2012. Population and distribution of Eurasian beaver (*Castor fiber*). *Baltic Forestry,* 18(1)**,** 168-175.
HALLEY, D. J. & ROSELL, F. 2003. Population and distribution of European beavers (*Castor fiber*). *Lutra,* 46(2)**,** 91–101.
HALLIWELL, E. C. 1997. *The Ecology of Red Squirrels in Scotland in relation to Pine Marten Predation.* PhD, University of Aberdeen.
HANSSON, L. 1985. The food of bank voles, wood mice and yellow-necked mice. *Symposia of the Zoological Society of London,* 55(-)**,** 141-168.
HARBUSCH, C. & RACEY, P. A. 2006. The sessile serotine: the influence of roost temperature on philopatry and reproductive phenology of *Eptesicus serotinus* (Schreber, 1774) (*Mammalia: Chiroptera*). *Acta Chiropterologica,* 8(1)**,** 213-229.
HARE, E. J. 2009. *Island syndrome in rodents; a comparative study on island forms of the bank vole, Myodes glareolus.* PhD, University of London.

HARRINGTON, L. A., HARRINGTON, A. L. & MACDONALD, D. W. 2008. Estimating the relative abundance of American mink *Mustela vison* on lowland rivers: evaluation and comparison of two techniques. *European Journal of Wildlife Research,* 54(1)**,** 79-87.

HARRINGTON, L. A., HARRINGTON, A. L., YAMAGUCHI, N., THOM, M. D., FERRERAS, P., WINDHAM, T. R. & MACDONALD, D. W. 2009. The impact of native competitors on an alien invasive: temporal niche shifts to avoid interspecific aggression. *Ecology,* 90(5)**,** 1207-1216.

HARRIS, G. 2014. *The Bat Assemblage of the Cotswold Water Park; an analysis of data collected 2005 to 2013*. http://www.waterpark.org/wp-content/uploads/2014/07/CWP-Bat-Atlas-v6-FINAL-sm61.pdf: The Cotswold Water Park Bat Initiative.

HARRIS, S. 1979a. Breeding season, litter size and nestling mortality of the Harvest mouse, *Micromys minutus* (*Rodentia: Muridae*), in Britain. *Journal of Zoology,* 188(4)**,** 437-442.

HARRIS, S. 1979b. History, distribution, status and habitat requirements of the Harvest mouse (*Micromys minutus*) in Britain. *Mammal Review,* 9(4)**,** 159-171.

HARRIS, S. 1984. Ecology of urban badgers *Meles meles:* distribution in Britain and habitat selection, persecution, food and damage in the city of Bristol. *Biological Conservation,* 28(4)**,** 349-375.

HARRIS, S., CRESSWELL, W., REASON, P. & CRESSWELL, P. 1992. An integrated approach to monitoring badger (*Meles meles*) population changes in Britain. *Wildlife 2001: populations.* Springer.

HARRIS, S., MORRIS, P., WRAY, S. & YALDEN, D. 1995. *A review of British mammals: population estimates and conservation status of British mammals other than cetaceans,* Peterborough, JNCC.

HARRIS, S. & RAYNER, J. 1986. Urban fox (*Vulpes vulpes*) population estimates and habitat requirements in several British cities. *The Journal of Animal Ecology,* 55(2)**,** 575-591.

HARRIS, S. & YALDEN, D. 2008. *Mammals of the British Isles: Handbook. 4th Edition,* Southampton, The Mammal Society.

HARRIS, S. J., MASSIMINO, D., NEWSON, S. E., EATON, M. A., MARCHANT, J. H., BALMER, D. E., NOBLE, D. G., GILLINGS, S., PROCTER, D. & PEARCE-HIGGINS, J. W. 2016. The Breeding Bird Survey 2015. *British Trust for Ornithology Research Report 687.* Thetford: British Trust for Ornithology.

HAYS, G. C., SPEAKMAN, J. R. & WEBB, P. I. 1992. Why do brown long-eared bats (*Plecotus auritus*) fly in winter? *Physiological Zoology,* 65(3)**,** 554-567.

HEALING, T. D. 1984. *Factors affecting the population dynamics of the Skomer vole (Clethrionomys glareolus skomerensis).* PhD, University of London.

HELYAR, A. F. 2005. *The ecology of American mink (Mustela vison) : response to control.* PhD, University of York.

HEMAMI, M. R., WATKINSON, A. R. & DOLMAN, P. M. 2005. Population densities and habitat associations of introduced muntjac *Muntiacus reevesi* and native roe deer *Capreolus capreolus* in a lowland pine forest. *Forest Ecology and Management,* 215(1-3)**,** 224-238.

HEMAMI, M. R., WATKINSON, A. R., GILL, R. M. A. & DOLMAN, P. M. 2007. Estimating abundance of introduced Chinese muntjac *Muntiacus reevesi* and native roe deer *Capreolus capreolus* using portable thermal imaging equipment. *Mammal Review,* 37(3)**,** 246-254.

HETHERINGTON, D. A. & CAMPBELL, R. D. 2012. The Cairngorms Wildcat Project Final Report. *Report to Cairngorms National Park Authority, Scottish Natural Heritage, Royal Zoological Society of Scotland, Scottish Gamekeepers Association and Forestry Commission Scotland.*

HEWISON, A. J. M., J P VINCENT, J. P., JOACHIM, J., ANGIBAULT, J. M., CARGNELUTTI, B. & CIBIEN, C. 2001. The effects of woodland fragmentation and human activity on roe deer distribution in agricultural landscapes. *Canadian Journal of Zoology,* 79(4)**,** 679-689.

HEWSON, R. & HINGE, M. D. C. 1990. Characteristics of the Home Range of Mountain Hares *Lepus timidus*. *Journal of Applied Ecology,* 27(2)**,** 651-666.

HEYDON, M. J., REYNOLDS, J. C. & SHORT, M. J. 2000. Variation in abundance of foxes (*Vulpes vulpes*) between three regions of rural Britain, in relation to landscape and other variables. *Journal of Zoology,* 251(2), 253-264.

HOF, A. R. & BRIGHT, P. W. 2010. The value of agri-environment schemes for macro-invertebrate feeders: hedgehogs on arable farms in Britain. *Animal Conservation,* 13(5), 467-473.

HOF, A. R. & BRIGHT, P. W. 2012. Factors affecting hedgehog presence on farmland as assessed by a questionnaire survey. *Acta Theriologica,* 57(1), 79-88.

HOF, A. R. & BRIGHT, P. W. 2016. Quantifying the long-term decline of the West European hedgehog in England by subsampling citizen-science datasets. *European Journal of Wildlife Research,* 62(4), 407-413.

HOLDERIED, M. W., JONES, G. & VON HELVERSEN, O. 2006. Flight and echolocation behaviour of whiskered bats commuting along a hedgerow: range-dependent sonar signal design, Doppler tolerance and evidence for 'acoustic focussing'. *Journal of Experimental Biology,* 209(10), 1816-26.

HOPE, P. R. & JONES, G. 2012. Warming up for dinner: torpor and arousal in hibernating Natterer's bats *(Myotis nattereri*) studied by radio telemetry. *Journal of Comparative Physiology. B, Biochemical, systemic, and environmental physiology,* 182(4), 569-78.

HOPKIRK, A. & RUSS, J. 2004. Pre-hibernal and hibernal activity and dispersal patterns of Leisler’s bat *Nyctalus leisleri* in Northern Ireland. *Final Report to the Environment and Heritage Service of Northern Ireland.*

HORÁCEK, I. 1975. Notes on the ecology of bats of the genus *Plecotus* Geoffroy, 1818 (Mammalia: *Chiroptera*). *Vestnik Ceskolovenske Zoologicke Spolecnostii,* 39(-), 195-210.

HORWOOD, M. T. & MASTERS, E. H. 1970. *Sika deer,* Reading, United Kingdom, British Deer Society.

HOWARD, R. W. 1995. *Auritus: a natural history of the brown long-eared bat,* York, William Sessions Ltd.

HOWIE, A. & STOKES, K. 2003. *Cornwall Water Shrew Survey*. Cornwall Wildlife Trust, Truro.

HUCK, M., DAVISON, J. & ROPER, T. J. 2008. Predicting European badger *Meles meles* sett distribution in urban environments. *Wildlife Biology,* 14(2), 188-198.

HUI, C., FOXCROFT, L. C., RICHARDSON, D. M. & MACFADYEN, S. 2011. Defining optimal sampling effort for large-scale monitoring of invasive alien plants: a Bayesian method for estimating abundance and distribution. *Journal of Applied Ecology,* 48(3), 768-776.

HUIJSER, M. P. & BERGERS, P. J. M. 2000. The effect of roads and traffic on hedgehog (*Erinaceus europaeus*) populations. *Biological Conservation,* 95(1), 111-116.

HUTCHINGS, M. R. & HARRIS, S. 1996. *The current status of the brown hare (Lepus europaeus) in Britain,* Peterborough, Joint Nature Conservation Committee.

HUTSON, A. M., SPITZENBERGER, F., JUSTE, J., AULAGNIER, S., PALMEIRIM, J., KARATAS, A. & PAUNOVIC, M. 2008. *Pipistrellus nathusii. In: IUCN 2013. IUCN Red List of Threatened Species. Version 2013.1* [Online]. [Accessed 10 January 2017].

HUTTERER, R. 2005. *Bat migrations in Europe; a review of banding data and literature. Vol. 28,* Bonn, Federal Agency for Nature Conservation.

IASON, G. R., MANSO, T., SIM, D. A. & HARTLEY, F. G. 2002. The functional response doesnot predict the local distribution of European rabbits (*Oryctolagus cuniculus*) ongrass swards: experimental evidence. *Functional Ecology,* 16(3), 394-402.

IJÄS, A., KAHILAINEN, A., VASKO, V. V. & LILLEY, T. M. 2017. Evidence of the Migratory Bat, *Pipistrellus nathusii,* Aggregating to the Coastlines in the Northern Baltic Sea. *Acta Chiropterologica,* 19(1), 127-139.

IOSSA, G., SOULSBURY, C. D., BAKER, P. J., EDWARDS, K. J. & HARRIS, S. 2009. Behavioral changes associated with a population density decline in the facultatively social red fox. *Behavioral Ecology,* 20(2), 385-395.

IUCN 2001. IUCN Red List Categories and Criteria: Version 3.1. IUCN, Gland and Cambridge: IUCN Species Survival Commission.

JACKSON, D. B. 2007. Factors affecting the abundance of introduced hedgehogs (*Erinaceus europaeus*) to the Hebridean island of South Uist in the absence of natural predators and implications for nesting birds. *Journal of Zoology,* 271(2)**,** 210-217.

JACKSON, D. L. 2000. *Guidance on the interpretation of the Biodiversity Broad Habitat Classification (terrestrial and freshwater types): Definitions and the relationship with other classifications. Report No. 307,* Peterborough, Joint Nature Conservation Committee

JACKSON, J. E. 1994. The edible or fat dormouse (*Glis glis*) in Britain. *Quarterly Journal of Forestry,* 88(-)**,** 119 - 125.

JAN, C. M. I., FRITH, K., GLOVER, A. M., BUTLIN, R. K., SCOTT, C. D., GREENAWAY, F., RUEDI, M., FRANTZ, A. C., DAWSON, D. A. & ALTRINGHAM, J. D. 2010. *Myotis alcathoe* Confirmed in the UK from Mitochondrial and Microsatellite DNA. *Acta Chiropterologica,* 12(2)**,** 471-483.

JARMAN, P. J. & CAPARANO, S. M. 1997. Use of Rock-wallaby faecal pellets for detecting and monitoring populations and examining habitat use. *Australian Mammalogy,* 19(2)**,** 257-264.

JEFFERIES, D. J., STRACHAN, C. & STRACHAN, R. 2003. Estimating numbers of the three interacting riparian mammals in Britain using survey data. *In:* JEFFERIES, D. J. (ed.) *The water vole and mink survey of 1996- 1998 with a history of the long-term changes in the status of both species and their causes.* Ledbury: Vincent Wildlife Trust.

JENKINS, E. V., LAINE, T., MORGAN, S. E., COLE, K. R. & SPEAKMAN, J. R. 1998. Roost selection in the pipistrelle bat, *Pipistrellus pipistrellus (Chiroptera: Vespertilionidae)*, in northeast Scotland. *Animal Behaviour,* 56(4)**,** 909-917.

JOINT NATURE CONSERVATION COMMITTEE. 2007. *Supporting documentation for making conservation status assessments: Technical Note I AlphaShapes range calculation tool. Second Report by the UK under Article 17 on the implementation of the Habitats Directive from January 2001 to December 2006* [Online]. Peterborough. Available: http://jncc.defra.gov.uk/pdf/FCS2007_techI_alphashapes.pdf [Accessed 28.02.2018 2018].

JOINT NATURE CONSERVATION COMMITTEE. 2013a. *Individual Species Reports - 3rd UK Habitats Directive Reporting 2013* [Online]. Available: http://jncc.defra.gov.uk/page-6391 [Accessed 25th February 2018 2018].

JOINT NATURE CONSERVATION COMMITTEE. 2013b. *Species Conservation Status Reports - 3rd UK Habitats Directive Reporting 2013* [Online]. Available: http://jncc.defra.gov.uk/page-6564 [Accessed 28.02.2018 2018].

JOINT NATURE CONSERVATION COMMITTEE. 2013c. *The UK Approach to Assessing Conservation Status for the 2013 EU Habitats Directive Article 17 Reporting* [Online]. Available: http://jncc.defra.gov.uk/page-6563 [Accessed 28.02.2018 2018].

JONES, G. 1991. Hibernal ecology of whiskered bats (*Myotis mystacinus*) and Brandt's bats (*Myotis brandti*) sharing the same roost site. *Myotis,* 29(-)**,** 121-128.

JONES, G. 1995. Flight performance, echolocation and foraging behaviour in noctule bats *Nyctalus noctula*. *Journal of Zoology,* 237(2)**,** 303-312.

JONES, G., JACOBS, D. S., KUNZ, T. H., WILLIG, M. R. & RACEY, P. A. 2009. Carpe noctem: the importance of bats as bioindicators. *Endangered Species Research,* 8(1-2)**,** 93-115.

JONES, G. & RAYNER, J. M. V. 1988. Flight performance, foraging tactics and echolocation in free-living Daubenton's bats *Myotis daubentoni (Chiroptera: Vespertilionidae)*. *Journal of Zoology,* 215(1)**,** 113-132.

JONES, G. & RYDELL, J. 1994. Foraging Strategy and Predation Risk as Factors Influencing Emergence Time in Echolocating Bats. *Philosophical Transactions of the Royal Society of London. Series B: Biological Sciences,* 346(1318)**,** 445-455.

JONES, K. E., ALTRINGHAM, J. D. & DEATON, R. 1996. Distribution and population densities of seven species of bat in northern England. *Journal of Zoology,* 240(3)**,** 788-798.

JUDGE, J., WILSON, G. J., MACARTHUR, R., DELAHAY, R. J. & MCDONALD, R. A. 2014. Density and abundance of badger social groups in England and Wales in 2011-2013. *Scientific Reports,* 4(Jan 23)**,** 3809.

JUDGE, J., WILSON, G. J., MACARTHUR, R., MCDONALD, R. A. & DELAHAY, R. J. 2017. Abundance of badgers (*Meles meles*) in England and Wales. *Scientific Reports,* 7(1)**,** 276.

JUSKAITIS, R., BALCIAUSKAS, L., BALTRUNAITE, L. & AUGUTE, V. 2015. Dormouse (*Gliridae*) populations on the northern periphery of their distributional ranges: a review. *Folia Zoologica,* 64(4)**,** 302-309.

JUSKAITIS, R. & BÜCHNER, S. 2013. *The Hazel Dormouse: Muscardinus avellanarius*, Wolf, Verlagskg.

KAŇUCH, P., KRIŠTÍN, A. & KRIŠTOFÍK, J. 2005. Phenology, diet, and ectoparasites of Leisler's bat (*Nyctalus leisleri*) in the Western Carpathians (Slovakia). *Acta Chiropterologica,* 7(2)**,** 249-257.

KEAN, E., LYONS, G. & CHADWICK, E. A. 2013. *Persistent organic pollutants and indicators of otter health: A CHEM Trust Report*.

KELLS, A. R. & GOULSON, D. 2003. Preferred nesting sites of bumblebee queens (*Hymenoptera: Apidae*) in agroecosystems in the UK. *Biological Conservation,* 109(2)**,** 165-174.

KENWARD, R. E., HODDER, K. H., ROSE, R. J., WALLS, C. A., PARISH, T., HOLM, J. L., MORRIS, P. A., WALLS, S. S. & DOYLE, F. I. 1998. Comparative demography of red squirrels (*Sciurus vulgaris*) and grey squirrels (*Sciurus carolinensis*) in deciduous and conifer woodland. *Journal of Zoology,* 244(1)**,** 7-21.

KERTH, G. & KONIG, B. 1999. Fission, Fusion and Nonrandom Associations in Female Bechstein's Bats (*Myotis bechsteinii*). *Behaviour,* 136(9)**,** 1187-1202.

KERTH, G., MAYER, F. & KÖNIG, B. 2000. Mitochondrial DNA (mtDNA) reveals that female Bechstein's bats live in closed societies. *Molecular Ecology,* 9(6)**,** 793-800.

KERTH, G., MAYER, F. & PETIT, E. 2002. Extreme sex-biased dispersal in the communally breeding, nonmigratory Bechstein's bat (*Myotis bechsteinii*). *Molecular Ecology,* 11(8)**,** 1491-8.

KERTH, G. & MELBER, M. 2009. Species-specific barrier effects of a motorway on the habitat use of two threatened forest-living bat species. *Biological Conservation,* 142(2)**,** 270-279.

KILSHAW, K. 2011. *Scottish wildcats: Naturally Scottish,* Edinburgh, Scottish Natural Heritage.

KILSHAW, K. 2015. *Introgression and the current status of the Scottish wildcat (Felis silvestris silvestris).* DPhil, University of Oxford / Wildlife Conservation Research Unit.

KILSHAW, K., DRAKE, A., MACDONALD, D. W. & KITCHENER, A. C. 2010. *The Scottish wildcat: a comparison of genetic and pelage characteristics*. Edinburgh: Scottish Natural Heritage Commissioned Report No.356.

KILSHAW, K., JOHNSON, P. J., KITCHENER, A. C. & MACDONALD, D. W. 2015. Detecting the elusive Scottish wildcat *Felis silvestris silvestris* using camera trapping. *Oryx,* 49(2)**,** 207-215.

KILSHAW, K., MONTGOMERY, R. A., CAMPBELL, R. D., HETHERINGTON, D. A., JOHNSON, P. J., KITCHENER, A. C., MACDONALD, D. W. & MILLSPAUGH, J. J. 2016. Mapping the spatial configuration of hybridization risk for an endangered population of the European wildcat (*Felis silvestris silvestris*) in Scotland. *Mammal Research,* 61(1)**,** 1-11.

KIRKPATRICK, L. 2017. *Bat exploitation of Sitka Spruce plantations: impacts of management on bats and their invertebrate prey.* PhD, University of Stirling.

KITCHENER, A. C., YAMAGUCHI, N., WARD, J. M. & MACDONALD, D. W. 2005. A diagnosis for the Scottish wildcat (*Felis silvestris*): a tool for conservation action for a critically-endangered felid. *Animal Conservation,* 8(3)**,** 223-237.

KNIGHT, T. & JONES, G. 2009. Importance of night roosts for bat conservation: roosting behaviour of the lesser horseshoe bat *Rhinolophus hipposideros*. *Endangered Species Research,* 9(1-2)), 79-86.

KNIPE, A., FOWLER, P. A., RAMSAY, S., HAYDON, D. T., MCNEILLY, A. S., THIRGOOD, S. & NEWEY, S. 2013. The effects of population density on the breeding performance of mountain hare *Lepus timidus*. *Wildlife Biology,* 19(4), 473-482.

KOTZAGEORGIS, G. C. & MASON, C. F. 1997. Small mammal populations in relation to hedgerow structure in an arable landscape. *Journal of Zoology,* 242(3), 425-434.

KOWALSKI, M. & LESIŃSKI, G. 1994. Bats occupying nest boxes for birds and bats in Poland. *Nyctalus (NF),* 5(1), 19-26.

KRUUK, H. & CONROY, J. 1987. Surveying otter Lutra lutra populations: a discussion of problems with spraints. *Biological Conservation,* 41(3), 179-183.

KRUUK, H., MOORHOUSE, A., CONROY, J., DURBIN, L. & FREARS, S. 1989. An estimate of numbers and habitat preferences of otters Lutra lutra in Shetland, UK. *Biological Conservation,* 49(4), 241-254.

KRYŠTUFEK, B. & FLAJŠMAN, B. E. 2007. *Dormouse and man*, Ekološki forum LDS & Liberalna akademija, Ljubljana. (in Slovenian).

KUBASIEWICZ, L. M. 2014. *Monitoring European pine martens (Martes martes) in Scottish forested landscapes.* PhD, University of Stirling.

LABUSCHAGNE, L., SWANEPOEL, L. H., TAYLOR, P. J., BELMAIN, S. R. & KEITH, M. 2016. Are avian predators effective biological control agents for rodent pest management in agricultural systems? *Biological Control,* 101(-), 94-102.

LAMBIN, X. 2008. Rodents: Order *Rodentia*, Field Vole. *In:* HARRIS, S. & YALDEN, D. (eds.) *Mammals of the British Isles: Handbook. 4th Edition.* Southampton: The Mammal Society.

LAMBIN, X., PETTY, S. J. & MACKINNON, J. L. 2000. Cyclic dynamics in field vole populations and generalist predation. *Journal of Animal Ecology,* 69(1), 106-118.

LANGBEIN, J. 2007. *National deer-vehicle collisions project: England (2003-2005): Final report to the Highways Agency.* Wrexham: Highways Agency.

LANGLEY, P. J. W. & YALDEN, D. W. 1977. Decline of rarer carnivores in Great Britain during 19th century. *Mammal Review,* 7(3-4), 95-116.

LANGTON, S. D., BRIGGS, P. A. & HAYSOM, K. A. 2010. Daubenton's bat distribution along rivers – developing and testing a predictive model. *Aquatic Conservation: Marine and Freshwater Ecosystems,* 20(S1), S45-S54.

LATHAM, J., STAINES, B. W. & GORMAN, M. L. 1996. The relative densities of red (*Cervus elaphus*) and roe (*Capreolus capreolus*) deer and their relationship in Scottish plantation forests. *Journal of Zoology,* 240(2), 285-299.

LATHAM, J., STAINES, B. W. & GORMAN, M. L. 1999. Comparative feeding ecology of red (*Cervus elaphus*) and roe deer (*Capreolus capreolus*) in Scottish plantation forests. *Journal of Zoology,* 247(3), 409-418.

LAWTON, J. H. & WOODROFFE, G. L. 1991. Habitat and the Distribution of Water Voles: Why are there Gaps in a Species' Range? *Journal of Animal Ecology,* 60(1), 79-91.

LEHNERT, L. S., KRAMER-SCHADT, S., SCHONBORN, S., LINDECKE, O., NIERMANN, I. & VOIGT, C. C. 2014. Wind farm facilities in Germany kill noctule bats from near and far. *PLOS ONE,* 9(8), e103106.

LINDENMAYER, D. B. & LIKENS, G. E. 2010. The science and application of ecological monitoring. *Biological Conservation,* 143(6), 1317-1328.

LINTOTT, P. R., BARLOW, K., BUNNEFELD, N., BRIGGS, P., GAJAS ROIG, C. & PARK, K. J. 2016. Differential responses of cryptic bat species to the urban landscape. *Ecology and Evolution,* 6(7), 2044-2052.

LINTOTT, P. R., BUNNEFELD, N., FUENTES-MONTEMAYOR, E., MINDERMAN, J., MAYHEW, R. J., OLLEY, L. & PARK, K. J. 2014. City life makes females fussy: sex differences in habitat use of temperate bats in urban areas. *Royal Society Open Science,* 1(3), 140200.

LINTOTT, P. R., BUNNEFELD, N. & PARK, K. J. 2015. Opportunities for improving the foraging potential of urban waterways for bats. *Biological Conservation,* 191(November 2015), 224-233.
LITTLEWOOD, N. A., CAMPBELL, R. D., DINNIE, L., GILBERT, L., HOOPER, R., IASON, G., IRVINE, J., KILSHAW, K., KITCHENER, A., LACKOVA, P., NEWEY, S., OGDEN, R. & ROSS, A. 2014. Survey and scoping of wildcat priority areas. *Scottish Natural Heritage Commissioned Report No. 768.*
LIU, C., SIBLY, R. M., GRIMM, V. & THORBEK, P. 2013. Linking pesticide exposure and spatial dynamics: an individual-based model of wood mouse (*Apodemus sylvaticus*) populations in agricultural landscapes. *Ecological Modelling,* 248(-), 92-102.
LOCK, J. 2006. Eradication of brown rats *Rattus norvegicus* and black rats *Rattus rattus* to restore breeding seabird populations on Lundy Island, Devon, England. *Conservation Evidence,* 3(-), 111-113.
LOCKIE, J. D. 1955. The breeding habits and food of short-eared owls after a vole plague. *Bird Study,* 2(2), 53-69.
LOUGHRAN, M. F. E. 2006. Social organisation of female field voles *Microtus agrestis* in a population in Southern England. *Acta Theriologica,* 51(3), 233-242.
LOUGHRAN, M. F. E. 2014. Report on the work carried out on Skomer voles between 2001 - 2013. *In:* BUESCHE, B., STUBBINGS, E., BRAMWELL, A. & MOSS, J. (eds.) *Skomer Island National Nature Reserve Annual Report 2014.* The Wildlife Trust of South and West Wales.
LOVE, R., WEBBON, C., GLUE, D. E. & HARRIS, S. 2000. Changes in the food of British Barn Owls (*Tyto alba*) between 1974 and 1997. *Mammal Review,* 30(2), 107-129.
LUČAN, R. K., ANDREAS, M., BENDA, P., BARTONIČKA, T., BŘEZINOVÁ, T., HOFFMANNOVÁ, A., HULOVÁ, Š., HULVA, P., NECKÁŘOVÁ, J., REITER, A. & SVAČINA, T. 2009. Alcathoe bat (*Myotis alcathoe*) in the Czech Republic: distributional status, roosting and feeding ecology. *Acta Chiropterologica,* 11(1), 61-69.
LUČAN , R. K. & HANÁK, V. 2011. Population ecology of *Myotis daubentonii (Mammalia: Chiroptera)* in South Bohemia: summary of two long-term studies: 1968-1984 and 1999-2009. *Acta Societatis Zoologicae Bohemicae,* 75(1), 67-85.
LUČAN, R. K. & RADIL, J. 2010. Variability of foraging and roosting activities in adult females of Daubenton's bat (*Myotis daubentonii*) in different seasons. *Biologia,* 65(6), 1072-1080.
LUNDY, M., MONTGOMERY, I. & RUSS, J. 2010. Climate change-linked range expansion of Nathusius' pipistrelle bat, *Pipistrellus nathusii* (Keyserling & Blasius, 1839). *Journal of Biogeography,* 37(12), 2232-2242.
LURZ, P. W. W., GARSON, P. J. & OGILVIE, J. F. 1998. Conifer species mixtures, cone crops and red squirrel conservation. *Forestry,* 71(1), 67-71.
LURZ, P. W. W., GARSON, P. J. & WAUTERS, L. A. 2000. Effects of temporal and spatial variations in food supply on the space and habitat use of red squirrels (*Sciurus vulgaris* L.). *Journal of Zoology,* 251(2), 167-178.
LUSH, L., WARD, A. I. & WHEELER, P. 2014. Opposing effects of agricultural intensification on two ecologically similar species. *Agriculture Ecosystems & Environment,* 192), 61-66.
MACDONALD, D., DONCASTER, C. P., NEWDICK, M., HOFER, H., MATHEWS, F. & JOHNSON, P. 2015. Foxes in the landscape: ecology and sociality. *Wildlife Conservation on Farmland: Conflict in the Countryside (Vol. 2).* Oxford: Oxford University Press.
MACDONALD, D., REYNOLDS, J., CARBONE, C., MATHEWS, F. & JOHNSON, P. 2003. The bioeconomics of fox control. *In:* TATTERSALL, F. & MANLEY, W. (eds.) *Conservation and Conflict: Mammals and Farming in Britain: Linnean Society Occasional Publications* Westbury Academic and Scientific Publishing.

MACDONALD, D. W., DANIELS, M. J., DRISCOLL, C., KITCHENER, A. & YAMAGUCHI, N. 2004a. The Scottish Wildcat: Analyses for Conservation and an Action Plan. *Wildlife Conservation Research Unit, University of Oxford.*

MACDONALD, D. W. & HARRINGTON, L. A. 2003. The American mink: the triumph and tragedy of adaptation out of context. *New Zealand Journal of Zoology,* 30)**,** 421-441.

MACDONALD, D. W., MITCHELMORE, F. & BACON, P. J. 1996. Predicting badger sett numbers: Evaluating methods in East Sussex. *Journal of Biogeography,* 23(5)**,** 649-655.

MACDONALD, D. W. & NEWMAN, C. 2002. Population dynamics of badgers (*Meles meles*) in Oxfordshire, UK: numbers, density and cohort life histories, and a possible role of climate change in population growth. *Journal of Zoology,* 256(1)**,** 121-138.

MACDONALD, D. W., NEWMAN, C., DEAN, J., BUESCHING, C. D. & JOHNSON, P. J. 2004b. The distribution of Eurasian badger, *Meles meles*, setts in a high-density area: field observations contradict the sett dispersion hypothesis. *Oikos,* 106(2)**,** 295-307.

MACDONALD, D. W., TATTERSALL, F. H., BROWN, E. D. & BALHARRY, D. 1995. Reintroducing the European beaver to Britain: Nostalgic meddling or restoring biodiversity? *Mammal Review,* 25(4)**,** 161-200.

MACKENZIE, G. A. & OXFORD, G. S. 1995. Prey of the noctule bat (*Nyctalus noctula*) in East Yorkshire. *Journal of Zoology,* 236(2)**,** 322-327.

MACKIE, I. J. 2002. *Aspects of the conservation biology of the noctule bat (Nyctalus noctula).* PhD, University of Aberdeen.

MACKIE, I. J. & RACEY, P. A. 2007. Habitat use varies with reproductive state in noctule bats (*Nyctalus noctula*): Implications for conservation. *Biological Conservation,* 140(1–2)**,** 70-77.

MACKINTOSH, M. 2016. *Bats and licensing: a report on the success of maternity roost compensation measure,* Edinburgh, Scottish Natural Heritage.

MACPHERSON, J., CROOSE, E., BAVIN, D., O'MAHONY, D., SOMPER, J. & BUTTRISS, N. 2014. *Feasibility Assessment for Reinforcing Pine Marten Numbers in England and Wales*. Ledbury: The Vincent Wildlife Trust.

MACPHERSON, J. L. & BRIGHT, P. W. 2010. A preliminary investigation into whether grazing marsh is an effective refuge for water voles from predation. *Lutra,* 53(1)**,** 21-28.

MACPHERSON, J. L. & BRIGHT, P. W. 2011. Metapopulation dynamics and a landscape approach to conservation of lowland water voles (*Arvicola amphibius*). *Landscape Ecology,* 26(10)**,** 1395-1404.

MAGURRAN, A. E., BAILLIE, S. R., BUCKLAND, S. T., DICK, J. M., ELSTON, D. A., SCOTT, E. M., SMITH, R. I., SOMERFIELD, P. J. & WATT, A. D. 2010. Long-term datasets in biodiversity research and monitoring: assessing change in ecological communities through time. *Trends in Ecology and Evolution,* 25(10)**,** 574-82.

MALO, A. F., GODSALL, B., PREBBLE, C., GRANGE, Z., MCCANDLESS, S., TAYLOR, A. & COULSON, T. 2012. Positive effects of an invasive shrub on aggregation and abundance of a native small rodent. *Behavioral Ecology,* 24(3)**,** 759-767.

MANN, J. & PUTMAN, R. 1989. Patterns of habitat use and activity in British populations of sika deer of contrasting environments. *Acta Theriologica,* 34(5)**,** 83-96.

MARQUES, F. F. C., BUCKLAND, S. T., GOFFIN, D., DIXON, C. E., BORCHERS, D. L., MAYLE, B. A. & PEACE, A. J. 2001. Estimating deer abundance from line transect surveys of dung: sika deer in southern Scotland. *Journal of Applied Ecology,* 38(2)**,** 349-363.

MARSH, A. C. W. & HARRIS, S. 2000. Partitioning of woodland habitat resources by two sympatric species of *Apodemus*: lessons for the conservation of the yellow-necked mouse (*A-flavicollis*) in Britain. *Biological Conservation,* 92(3)**,** 275-283.

MARSH, A. C. W., POULTON, S. & HARRIS, S. 2001. The Yellow-necked Mouse *Apodemus flavicollis* in Britain: status and analysis of factors affecting distribution. *Mammal Review,* 31(3)**,** 203-227.

MARTÍNKOVÁ, N., BARNETT, R., CUCCHI, T., STRUCHEN, R., PASCAL, M., PASCAL, M., FISCHER, M. C., HIGHAM, T., BRACE, S., HO, S. Y. W., QUÉRÉ, J.-P., O'HIGGINS,

P., EXCOFFIER, L., HECKEL, G., RUS HOELZEL, A., DOBNEY, K. M. & SEARLE, J. B. 2013. Divergent evolutionary processes associated with colonization of offshore islands. *Molecular Ecology,* 22(20), 5205-5220.

MATHEWS, F., RICHARDSON, S. M. & HOSKEN, D. J. 2016. *Understand the Risks to Bat Populations Posed By Wind Turbines - Phase 2 - WC0753*. London: Defra.

MATHEWS, F., ROCHE, N., AUGHNEY, T., JONES, N., DAY, J., BAKER, J. & LANGTON, S. 2015. Barriers and benefits: implications of artificial night-lighting for the distribution of common bats in Britain and Ireland. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 370(1667), 20140124.

MAYLE, B. A. 1996. Progress in predictive management of deer populations in British woodlands. *Forest Ecology and Management,* 88(1-2), 187-198.

MAYLE, B. A. & BROOME, A. C. 2013. Changes in the impact and control of an invasive alien: the grey squirrel (*Sciurus carolinensis*) in Great Britain, as determined from regional surveys. *Pest Management Science,* 69(3), 323-333.

MCANEY, C., SHIEL, C., SULLIVAN, C. & FAIRLEY, J. 1991. *The Analysis of Bat Droppings,* London, Mammal Society.

MCCRACKEN, D. I. 1993. The potential for avermectins to affect wildlife. *Veterinary Parasitology,* 48(1-4), 273-80.

MCDONALD, R. A., HUTCHINGS, M. R. & KEELING, J. G. 1997. The status of ship rats *Rattus rattus* on the Shiant Islands, Outer Hebrides, Scotland. *Biological Conservation,* 82(1), 113-117.

MCDONALD, R. A., O'HARA, K. & MORRISH, D. J. 2007. Decline of invasive alien mink (*Mustela vison*) is concurrent with recovery of native otters (*Lutra lutra*). *Diversity and Distributions,* 13(1), 92-98.

MCDONALD, R. A., WEBBON, C. & HARRIS, S. 2000. The diet of stoats (*Mustela erminea*) and weasels (*Mustela nivalis*) in Great Britain. *Journal of Zoology,* 252(3), 363-371.

MCGUIRE, C., WHITFIELD, D., PERKINS, H. & OWEN, C. 2014. *National Water Vole Database and Mapping Project: Guide to the Use of Project Outputs to End of 2012*, The Wildlife Trusts.

MERGEY, M., HELDER, R. & ROEDER, J. J. 2011. Effect of forest fragmentation on space-use patterns in the European pine marten (*Martes martes*). *Journal of Mammalogy,* 92(2), 328-335.

MICOL, T., DONCASTER, C. P. & MACKINLAY, L. A. 1994. Correlates of local variation in the abundance of hedgehogs *Erinaceus europaeus*. *Journal of Animal Ecology,* 63(4), 851-860.

MILLER, H. 2011. *Bechstein's bat survey: final report,* London, The Bat Conservation Trust.

MOLL, R. J., KILSHAW, K., MONTGOMERY, R. A., ABADE, L., CAMPBELL, R. D., HARRINGTON, L. A., MILLSPAUGH, J. J., BIRKS, J. D. S. & MACDONALD, D. W. 2016. Clarifying habitat niche width using broad-scale, hierarchical occupancy models: a case study with a recovering mesocarnivore. *Journal of Zoology,* 300(3), 177-185.

MOLLER, H. 1983. Foods and foraging behaviour of Red (*Sciurus vulgaris*) and Grey (*Sciurus carolinensis*) squirrels. *Mammal Review,* 13(2-4), 81-98.

MONTGOMERY, W. I. 1978. Studies on the distributions of *Apodemus sylvaticus* (L.) and *A. Flavicollis* (Melchior) in Britain. *Mammal Review,* 8(4), 177-184.

MONTGOMERY, W. I. 1980. Population structure and dynamics of sympatric *Apodemus* species (*Rodentia: Muridae*). *Journal of Zoology,* 192(3), 351-377.

MONTGOMERY, W. I. 1989. Population regulation in the wood mouse, *Apodemus sylvaticus*. I. density dependence in the annual cycle of abundance. *Journal of Animal Ecology,* 58(2), 465-475.

MONTGOMERY, W. I. & DOWIE, M. 1993. The distribution of the wood mouse *Apodemus sylvaticus* and the house mouse *Mus domesticus* on farmland in north-east Ireland. *Irish Naturalists Journal,* 24(5), 199-203.

MOORE, N. P., ASKEW, N. & BISHOP, J. D. 2003. Small mammals in new farm woodlands. *Mammal Review,* 33(1), 101-104.

MORO, D. & GADAL, S. 2007. Benefits of habitat restoration to small mammal diversity and abundance in a pastoral agricultural landscape in mid-Wales. *Biodiversity and Conservation,* 16(12), 3543-3557.

MORRIS, P. A. 1993. *A Red Data Book for British Mammals* London, The Mammal Society.

MORRIS, P. A. & HOODLESS, A. 1992. Movements and hibernaculum site in the fat dormouse (*Glis glis*). *Journal of Zoology,* 228(4), 685-687.

MORRIS, P. A. & MORRIS, M. J. 2010. A 13-year population study of the edible dormouse *Glis glis* in Britain. *Acta Theriologica,* 55(3), 279-288.

MORRIS, P. A. & TEMPLE, R. K. 1998. 'Nest tubes' — a potential new method for controlling numbers of the edible dormouse (*Glis glis*) in plantations. *Quarterly Journal of Forestry,* 92(-), 201–205.

MORTIMER, G. 2006. *Foraging, roosting and survival of Natterer's bats, Myotis nattereri, in a commercial coniferous plantation.* PhD, University of St Andrews.

MORTON, D., ROWLAND, C., WOOD, C., MEEK, L., MARSTON, C., SMITH, G., WADSWORTH, R. & SIMPSON, I. C. 2011. Final Report for LCM2007 - the new UK Land Cover Map. *CS Technical Report No 11/07.* Centre for Ecology & Hydrology (Natural Environment Research Council).

MOUSSY, C. 2011. Selection of Old Stone Buildings as Summer Day Roost by the Brown Long-Eared Bat *Plecotus auritus*. *Acta Chiropterologica,* 13(1), 101-111.

MOUSSY, C., ATTERBY, H., GRIFFITHS, A. G. F., ALLNUTT, T. R., MATHEWS, F., SMITH, G. C., AEGERTER, J. N., BEARHOP, S. & HOSKEN, D. J. 2015. Population genetic structure of serotine bats (*Eptesicus serotinus*) across Europe and implications for the potential spread of bat rabies (European bat lyssavirus EBLV-1). *Heredity,* 115(1), 83-92.

MOUSSY, C., HOSKEN, D. J., MATHEWS, F., SMITH, G. C., AEGERTER, J. N. & BEARHOP, S. 2013. Migration and dispersal patterns of bats and their influence on genetic structure. *Mammal Review,* 43(3), 183-195.

MURPHY, S. E., GREENAWAY, F. & HILL, D. A. 2012. Patterns of habitat use by female brown long-eared bats presage negative impacts of woodland conservation management. *Journal of Zoology,* 288(3), 177-183.

MUTCH, F. & SCOTTISH NATURAL HERITAGE 2000. The influence of dredging on water vole populations in small ditches. *Scottish Natural Heritage Commissioned Report F99LF13.*

NAIM, D. M., TELFER, S., SANDERSON, S., KEMP, S. J. & WATTS, P. C. 2011. Prevalence of multiple mating by female common dormice, *Muscardinus avellanarius*. *Conservation Genetics,* 12(4), 971-979.

NEWEY, S., BELL, M., ENTHOVEN, S. & THIRGOOD, S. 2003. Can distance sampling and dung plots be used to assess the density of mountain hares *Lepus timidus*? *Wildlife Biology,* 9(3), 185-192.

NEWEY, S., IASON, G. & RAYNOR, R. 2008. The conservation status and management of mountain hares. *Scottish Natural Heritage Commissioned Report No. 287 (ROAME No. F05AC316).*

NEWEY, S., WILLEBRAND, T., HAYDON, D. T., DAHL, F., AEBISCHER, N. J., SMITH, A. A. & THIRGOOD, S. J. 2007. Do mountain hare populations cycle? *Oikos,* 116(9), 1547-1557.

NEWSOME, T. M., CROWTHER, M. S. & DICKMAN, C. R. 2014. Rapid recolonisation by the European red fox: how effective are uncoordinated and isolated control programs? *European Journal of Wildlife Research,* 60(5), 749-757.

NEWTON-CROSS, G., WHITE, P. C. & HARRIS, S. 2007. Modelling the distribution of badgers *Meles meles*: comparing predictions from field-based and remotely derived habitat data. *Mammal Review,* 37(1), 54-70.

NGAMPRASERTWONG, T., MACKIE, I. J., RACEY, P. A. & PIERTNEY, S. B. 2008. Spatial distribution of mitochondrial and microsatellite DNA variation in Daubenton's bat within Scotland. *Molecular Ecology,* 17(14), 3243-58.

NICHOLLS, B. & A. RACEY, P. 2006. Habitat selection as a mechanism of resource partitioning in two cryptic bat species *Pipistrellus pipistrellus* and *Pipistrellus pygmaeus*. *Ecography,* 29(5)**,** 697-708.

NORBERG, U. M. 1976a. Aerodynamics of hovering flight in the long-eared bat *Plecotus auritus*. *Journal of Experimental Biology,* 65(2)**,** 459-70.

NORBERG, U. M. 1976b. Aerodynamics, kinematics, and energetics of horizontal flapping flight in the long-eared bat *Plecotus auritus*. *Journal of Experimental Biology,* 65(1)**,** 179-212.

NORBERG, U. M. & RAYNER, J. M. 1987a. Ecological morphology and flight in bats (Mammalia; Chiroptera): wing adaptations, flight performance, foraging strategy and echolocation. *Philosophical Transactions of the Royal Society B: Biological Sciences,* 316(1179)**,** 335-427.

NORBERG, U. M. & RAYNER, J. M. V. 1987b. Ecological Morphology and Flight in Bats (*Mammalia; Chiroptera*): Wing Adaptations, Flight Performance, Foraging Strategy and Echolocation. *Philos Trans R Soc Lond B Biol Sci,* 316(1179)**,** 335-427.

NORRDAHL, K. & KORPIMÄKI, E. 1995. Effects of predator removal on vertebrate prey populations: birds of prey and small mammals. *Oecologia,* 103(2)**,** 241-248.

O'KEEFFE, D. A. & FAIRLEY, J. S. 1981. Two population studies of Irish Pygmy shrews. *Irish Naturalists' Journal,* 20(July)**,** 269-275.

O'NEILL, D., TURNER, P. D., O'MEARA, D. B., CHADWICK, E. A., COFFEY, L. & O'REILLY, C. 2013. Development of novel real-time TaqMan® PCR assays for the species and sex identification of otter (*Lutra lutra*) and their application to noninvasive genetic monitoring. *Molecular Ecology Resources,* 13(5)**,** 877-883.

OAKLEY, S. F. & JONES, G. 1998. Habitat around maternity roosts of the 55 kHz phonic type of pipistrelle bats (*Pipistrellus pipistrellus*). *Journal of Zoology,* 245(2)**,** 222-228.

OXFORD, W. U. O. 2004. *Water vole surveys in Fife and Central Cairngorms,* Edinburgh, Scottish Natural Heritage.

OZGUL, A., TULJAPURKAR, S., BENTON, T. G., PEMBERTON, J. M., CLUTTON-BROCK, T. H. & COULSON, T. 2009. The Dynamics of Phenotypic Change and the Shrinking Sheep of St. Kilda. *Science,* 325(5939)**,** 464-467.

PALMER, E., PIMLEY, E., SUTTON, G. & BIRKS, J. 2013. *A study on the population size, foraging range and roosting ecology of Bechstein's bats at Grafton Wood SSSI Worcestershire: Report for the People's Trust for Endangered Species and Worcestershire Wildlife Trust* [Online]. Available: http://citeseerx.ist.psu.edu/viewdoc/download?doi=10.1.1.729.6707&rep=rep1&type=pdf [Accessed 05/03/2017].

PALMER, S. C. F., HESTER, A. J., ELSTON, D. A., GORDON, I. J. & HARTLEY, S. E. 2003. The perils of having tasty neighbors: Grazing impacts of large herbivores at vegetation boundaries. *Ecology,* 84(11)**,** 2877-2890.

PARK, K., MASTERS, E. & ALTRINGHAM, J. 1998. Social structure of three sympatric bat species (Vespertilionidae). *Journal of Zoology,* 244(3)**,** 379-389.

PARROTT, D., ETHERINGTON, T. R. & DENDY, J. 2014. A geographically extensive survey of hedgehogs (*Erinaceus europaeus*) in England. *European Journal of Wildlife Research,* 60(2)**,** 399-403.

PARROTT, D., PRICKETT, A., PIETRAVALLE, S., ETHERINGTON, T. R. & FLETCHER, M. 2012. Estimates of regional population densities of badger *Meles meles*, fox *Vulpes vulpes* and hare *Lepus europaeus* using walked distance sampling. *European Journal of Wildlife Research,* 58(1)**,** 23-33.

PARSONS, K. N. & JONES, G. 2003. Dispersion and habitat use by *Myotis daubentonii* and *Myotis nattereri* during the swarming season: implications for conservation. *Animal Conservation,* 6(4)**,** 283-290.

PARSONS, K. N., JONES, G., DAVIDSON-WATTS, I. & GREENAWAY, F. 2003a. Swarming of bats at underground sites in Britain—implications for conservation. *Biological Conservation,* 111(1)**,** 63-70.

PARSONS, K. N., JONES, G. & GREENAWAY, F. 2003b. Swarming activity of temperate zone microchiropteran bats: effects of season, time of night and weather conditions. *Journal of Zoology,* 261(3), 257-264.

PATEIRO-LÓPEZ, B. & RODRıGUEZ-CASAL, A. 2010. Generalizing the convex hull of a sample: the R package alphahull. *Journal of Statistical Software,* 34(5), 1-28.

PATTON, V., EWALD, J. A., SMITH, A. A., NEWEY, S., IASON, G. R., THIRGOOD, S. J. & RAYNOR, R. 2010. Distribution of mountain hares *Lepus timidus* in Scotland: results from a questionnaire. *Mammal Review,* 40(4), 313-326.

PEARCE, J. & VENIER, L. 2005. Small mammals as bioindicators of sustainable boreal forest management. *Forest Ecology and Management,* 208(1–3), 153-175.

PEREBOOM, V., MERGEY, M., VILLERETTE, N., HELDER, R., GERARD, J. F. & LODE, T. 2008. Movement patterns, habitat selection, and corridor use of a typical woodland-dweller species, the European pine marten (*Martes martes*), in fragmented landscape. *Canadian Journal of Zoology,* 86(9), 983-991.

PEREIRA, H. M., BELNAP, J., BRUMMITT, N., COLLEN, B., DING, H., GONZALEZ-ESPINOSA, M., GREGORY, R. D., HONRADO, J., JONGMAN, R. H. G., JULLIARD, R., MCRAE, L., PROENÇA, V., RODRIGUES, P., OPIGE, M., RODRIGUEZ, J. P., SCHMELLER, D. S., VAN SWAAY, C. & VIEIRA, C. 2010. Global biodiversity monitoring. *Frontiers in Ecology and the Environment,* 8(9), 459-460.

PEREZ-ESPONA, S., PEREZ-BARBERIA, F. J., JIGGINS, C. D., GORDON, I. J. & PEMBERTON, J. M. 2010. Variable extent of sex-biased dispersal in a strongly polygynous mammal. *Molecular Ecology,* 19(15), 3101-3113.

PERNETTA, J. C. 1977. Population ecology of British shrews in grassland. *Acta Theriologica,* 22(20-2), 279-296.

PERROW, M. R. & JOWITT, A. J. D. 1995. What future for the Harvest Mouse? *British Wildlife,* 6(-), 356-365.

PETIT, E. & MAYER, F. 2000. A population genetic analysis of migration: the case of the noctule bat (*Nyctalus noctula*). *Molecular Ecology,* 9(6), 683-690.

PETROVAN, S. O., BARRIO, I. C., WARD, A. I. & WHEELER, P. M. 2011a. Farming for pests? Local and landscape-scale effects of grassland management on rabbit densities. *European Journal of Wildlife Research,* 57(1), 27-34.

PETROVAN, S. O., WARD, A. I. & WHEELER, P. 2011b. Detectability Counts when Assessing Populations for Biodiversity Targets. *PLOS ONE,* 6(9), 8.

PETTETT, C. E., MOORHOUSE, T. P., JOHNSON, P. J. & MACDONALD, D. W. 2017. Factors affecting hedgehog (*Erinaceus europaeus*) attraction to rural villages in arable landscapes. *European Journal of Wildlife Research,* 63(3), 54.

PHILLIPS, W. W. A. & BLACKMORE, M. 1970. Mouse-eared bats *Myotis myotis* in Sussex. *Journal of Zoology,* 162), 520-521.

PICKVANCE, T. J. & CHARD, J. S. R. 1960. Midland mammals survey 1. Feral muntjac deer (*Muntiacus* spp.) in the West Midlands, with special reference to Warwickshire. *Proceedings of the Birmingham Natural History and Philosophical Society,* 19(-), 1-8.

PILASTRO, A., TAVECCHIA, G. & MARIN, G. 2003. Long living and reproduction skipping in the fat dormouse. *Ecology,* 84(7), 1784-1792.

PILĀTS, V., PILĀTE, D. & DZALBA, I. 2009. The use of nest boxes to survey marginally distributed fat dormouse *Glis glis* in Latvia. *Acta Universitatis Latviensis seria Biology,* 753(-), 7-18.

PLATT, F. B. W. & ROWE, J. J. 1964. Damage by the edible dormouse (*Glis glis* L.) at Wendover Forest (Chilterns). *Quarterly Journal of Forestry,* 58(-), 228-233.

PLUMMER, K. E., HALE, J. D., O'CALLAGHAN, M. J., SADLER, J. P. & SIRIWARDENA, G. M. 2016. Investigating the impact of street lighting changes on garden moth communities. *Journal of Urban Ecology,* 2(1), juw004-juw004.

POCOCK, M. J. O., SEARLE, J. B. & WHITE, P. C. L. 2004. Adaptations of animals to commensal habitats: population dynamics of house mice *Mus musculus* domesticus on farms. *Journal of Animal Ecology,* 73(5), 878-888.

POOLE, D. W., COWAN, D. P. & SMITH, G. C. 2003. Developing a census method based on sight counts to estimate rabbit (*Oryctolagus cuniculus*) numbers. *Wildlife Research,* 30(5), 487-493.

POWELL, R. A. 1979. Mustelid spacing patterns: variations on a theme by Mustela. *Ethology,* 50(2), 153-165.

PUTMAN, R., WATSON, P. & LANGBEIN, J. 2011. Assessing deer densities and impacts at the appropriate level for management: a review of methodologies for use beyond the site scale. *Mammal Review,* 41(3), 197-219.

PUTMAN, R. J. & STAINES, B. W. 2004. Supplementary winter feeding of wild red deer *Cervus elaphus* in Europe and North America: justifications, feeding practice and effectiveness. *Mammal Review,* 34(4), 285-306.

QUY, R. J., WATKINS, R. W., LAMBERT, M. S. & COWAN, D. P. 2009. Validating census methods to measure changes in house mouse populations. *Pest Management Science,* 65(3), 275-282.

R CORE TEAM 2015. R: a language and environment for statistical computing, version 3.2. 2. R Foundation for Statistical Computing, Vienna, Austria.

RACEY, P. A. 1969. Diagnosis of pregnancy and experimental extension of gestation in the pipistrelle bat, *Pipistrellus pipistrellus*. *Journal of Reproduction and Fertility,* 19(3), 465-74.

RACEY, P. R., SWIFT, S. M., RYDELL, J. & BRODIE, L. 1998. Bats and insects over two Scottish rivers with contrasting nitrate status. *Animal Conservation,* 1(3), 195-202.

RACKHAM, J. 1979. *Rattus rattus*: the introduction of the black rat into Britain. *Antiquity,* 53(208), 112-120.

RAINEY, E., BUTLER, A., BIERMAN, S. & ROBERTS, A. M. I. 2009. *Scottish Badger Distribution Survey 2006 - 2009: estimating the distribution and density of badger main setts in Scotland,* Innerleithen, Scottish Badgers.

RANSOME, R. D. 1989. Population changes of Greater horseshoe bats studied near Bristol over the past twenty-six years. *Biological Journal of the Linnean Society,* 38(1), 71-82.

RANSOME, R. D. 1996. *The management of feeding areas for greater horseshoe bats: English Nature Research Report No. 174,* Peterborough, English Nature.

RANSOME, R. D. 1998. *The impact of maternity roost conditions on populations of greater horseshoe bats: English Nature Research Report No. 292,* Peterborough, English Nature.

RAZGOUR, O., CLARE, E. L., ZEALE, M. R., HANMER, J., SCHNELL, I. B., RASMUSSEN, M., GILBERT, T. P. & JONES, G. 2011a. High-throughput sequencing offers insight into mechanisms of resource partitioning in cryptic bat species. *Ecology and Evolution,* 1(4), 556-570.

RAZGOUR, O., HANMER, J. & JONES, G. 2011b. Using multi-scale modelling to predict habitat suitability for species of conservation concern: the grey long-eared bat as a case study. *Biological Conservation,* 144(12), 2922-2930.

RAZGOUR, O., JUSTE, J., IBÁÑEZ, C., KIEFER, A., REBELO, H., PUECHMAILLE, S. J., ARLETTAZ, R., BURKE, T., DAWSON, D. A., BEAUMONT, M. & JONES, G. 2013. The shaping of genetic variation in edge-of-range populations under past and future climate change. *Ecology Letters,* 16(10), 1258-1266.

RAZGOUR, O., REBELO, H., PUECHMAILLE, S. J., JUSTE, J., IBÁÑEZ, C., KIEFER, A., BURKE, T., DAWSON, D. A. & JONES, G. 2014. Scale-dependent effects of landscape variables on gene flow and population structure in bats. *Diversity and Distributions,* 20(10), 1173-1185.

RAZGOUR, O. N. 2012. *From genes to landscapes: conservation biology of the grey long-eared bat, Plecotus austriacus, across spatio-temporal scales.* PhD Thesis, University of Bristol.

REBELO, H., TARROSO, P. & JONES, G. 2010. Predicted impact of climate change on European bats in relation to their biogeographic patterns. *Global Change Biology,* 16(2), 561-576.

REGAN, C. E., PILKINGTON, J. G., PEMBERTON, J. M. & CRAWLEY, M. J. 2016. Sex differences in relationships between habitat use and reproductive performance in Soay sheep (Ovis aries). *Ecology Letters,* 19(2), 171-179.

RENNISON, B. O. & DRUMMOND, D. C. 1984. Monitoring and improving rodent control progress in non-agricultural premises in England and Wales. *Environmental Health,* 92(11), 287-297.

REYNOLDS, P. 1992. *The impact of changes in land-use in Orkney, on the vole Microtus arvalis orcadensis and its avian predators.* PhD, University of Aberdeen.

REYNOLDS, P. & TELFER, S. 2000. *Survey of Water Voles within the River Tyne Catchment: Scottish Natural Heritage Commissioned Report F00LJ07,* Edinburgh, Scottish Natural Heritage.

RICHARDS, C. G. J. 1989. The pest status of rodents in the United Kingdom. *In:* PUTMAN, R. J. (ed.) *Mammals as Pests.* London: The Mammal Society.

RICHARDSON, P. W. 2000. *Distribution Atlas of Bats in Britain and Ireland, 1980-1999,* London, The Bat Conservation Trust.

RINDLE, U. & ZAHN, A. 1997. Untersuchungen zum Nahrungsspektrum der Kleinen Bartfledermaus (*Myotis mystacinus*). *Nyctalus,* 6(-), 304-308.

RISELY, K., BAILLIE, S. R., EATON, M. A., JOYS, A. C., MUSGROVE, A. J., NOBLE, D. G., RENWICK, A. R. & WRIGHT, L. J. 2010. *The Breeding Bird Survey 2009,* Thetford, British Trust for Ornithology.

RITCHIE, J. 2015. *The influence of man on animal life in Scotland: a study in faunal evolution,* Cambridge, Cambridge University Press.

RIVERS, N. M., BUTLIN, R. K. & ALTRINGHAM, J. D. 2005. Genetic population structure of Natterer's bats explained by mating at swarming sites and philopatry. *Molecular Ecology,* 14(14), 4299-312.

ROBERTSON, P. A., ADRIAENS, T., LAMBIN, X., MILL, A., ROY, S., SHUTTLEWORTH, C. M. & SUTTON-CROFT, M. 2017. The large-scale removal of mammalian invasive alien species in Northern Europe. *Pest Management Science,* 73(2), 273-279.

ROBINSON, M. F. & STEBBINGS, R. E. 1993. Food of the serotine bat, *Eptesicus serotinus*—is faecal analysis a valid qualitative and quantitative technique? *Journal of Zoology,* 231(2), 239-248.

ROBINSON, M. F. & STEBBINGS, R. E. 1997. Home range and habitat use by the serotine bat, *Eptesicus serotinus*, in England. *Journal of Zoology,* 243(1), 117-136.

ROELEKE, M., BLOHM, T., KRAMER-SCHADT, S., YOVEL, Y. & VOIGT, C. C. 2016. Habitat use of bats in relation to wind turbines revealed by GPS tracking. *Scientific Reports,* 6(-), 28961.

ROGERS, L. M., CHEESEMAN, C. L., MALLINSON, P. J. & CLIFTON-HADLEY, R. 1997. The demography of a high-density badger (*Meles meles*) population in the west of England. *Journal of Zoology,* 242(4), 705-728.

ROOS, S., JOHNSTON, A. & NOBLE, D. 2012. *UK Hedgehog Datasets and their Potential for Long-Term Monitoring. BTO Research Report No. 598,* Thetford, British Trust for Ornithology.

ROSSITER, S. J., JONES, G., RANSOME, R. D. & BARRATT, E. M. 2000. Parentage, reproductive success and breeding behaviour in the greater horseshoe bat (*Rhinolophus ferrumequinum*). *Proceedings of the Royal Society of London. Series B: Biological Sciences,* 267(1443), 545-551.

ROSSITER, S. J., JONES, G., RANSOME, R. D. & BARRATT, E. M. 2001. Outbreeding increases offspring survival in wild greater horseshoe bats (*Rhinolophus ferrumequinum*). *Proceedings of the Royal Society B: Biological Sciences,* 268(1471), 1055-61.

ROSSITER, S. J., RANSOME, R. D., FAULKES, C. G., LE COMBER, S. C. & JONES, G. 2005. Mate fidelity and intra-lineage polygyny in greater horseshoe bats. *Nature,* 437(7057), 408-411.

ROSSOLIMO, O. L., POTAPOVA, E. G., PAVLINOV, I. Y., KRUSKOP, S. V. & VOLTZIT, O. V. 2001. *Dormice (Myoxidae) of the world*, Izdatel'stvoMoskovskogo Universiteta, Moskva. (in Russian).
ROTHSCHILD, M. & MARSH, H. 1956. Increase of hares (*Lepus europaeus* Pallas) at Ashton Wold, with a note on the reduction in numbers of the brown rat (*Rattus norvegicus* Berkenhout). *Proceedings of the Zoological Society, London,* 127(3)**,** 441–445.
ROWCLIFFE, J. M., FIELD, J., TURVEY, S. T. & CARBONE, C. 2008. Estimating animal density using camera traps without the need for individual recognition. *Journal of Applied Ecology,* 45(4)**,** 1228-1236.
ROWE, F. P., SWINNEY, T. & QUY, R. J. 1983. Reproduction of the house mouse (*Mus musculus*) in farm buildings. *Journal of Zoology,* 199(2)**,** 259-269.
RUDOLPH, B.-U., KERTH, K., SCHLAPP, G. & WOLZ, I. 2004. Bechsteinfledermaus *Myotis bechsteinii* (Kuhl, 1817). *Fledermaüse in Bayern (eds Meschede A, Rudolph B-U).* Stuttgart: Ulmer Verlag.
RUDOLPH, B. U., LIEGL, A. & HELVERSEN, O. V. 2009. Habitat selection and activity patterns in the greater mouse-eared bat *Myotis myotis*. *Acta Chiropterologica,* 11(2)**,** 351-361.
RUEDI, M. & MAYER, F. 2001. Molecular systematics of bats of the genus *Myotis (Vespertilionidae)* suggests deterministic ecomorphological convergences. *Molecular Phylogenetics and Evolution,* 21(3)**,** 436-48.
RUF, T., FIETZ, J., SCHLUND, W. & BIEBER, C. 2006. High survival in poor years: life history tactics adapted to mast seeding in the edible dormouse. *Ecology,* 87(2)**,** 372-381.
RUSHTON, S. P., BARRETO, G. W., CORMACK, R. M., MACDONALD, D. W. & FULLER, R. 2000. Modelling the effects of mink and habitat fragmentation on the water vole. *Journal of Applied Ecology,* 37(3)**,** 475-490.
RUSHTON, S. P., SHIRLEY, M. D. F., MACDONALD, D. W. & REYNOLDS, J. C. 2006. Effects of culling fox populations at the landscape scale: A spatially explicit population modeling approach. *Journal of Wildlife Management,* 70(4)**,** 1102-1110.
RUSS, J. 2012. *British bat calls: a guide to species identification,* Exeter, Pelagic Publishing.
RUSS, J. M. 2014. *Nathusius' pipistrelle in Great Britain & Ireland* [Online]. Available at: http://www.nathusius.org.uk. [Accessed accessed 10th September 2016].
RUSS, J. M. & MONTGOMERY, W. I. 2002. Habitat associations of bats in Northern Ireland: implications for conservation. *Biological Conservation,* 108(1)**,** 49-58.
RUSSO, D. 2002. Elevation affects the distribution of the two sexes in Daubenton's bats *Myotis daubentonii (Chiroptera: Vespertilionidae)* from Italy. *Mammalia,* 66(4)**,** 543-52.
RUSSO, D., CISTRONE, L. & JONES, G. 2005. Spatial and temporal patterns of roost use by tree-dwelling barbastelle bats *Barbastella barbastellus*. *Ecography,* 28(6)**,** 769-776.
RUSSO, D., CISTRONE, L., JONES, G. & MAZZOLENI, S. 2004. Roost selection by barbastelle bats (*Barbastella barbastellus, Chiroptera: Vespertilionidae*) in beech woodlands of central Italy: consequences for conservation. *Biological Conservation,* 117(1)**,** 73-81.
RYDELL, J. & BOGDANOWICZ, W. 1997. *Barbastella barbastellus*. *Mammalian Species,* 557(-)**,** 1-8.
RYDELL, J., MILLER, L. A. & JENSEN, M. E. 1999. Echolocation constraints of Daubenton's Bat foraging over water. *Functional Ecology,* 13(2)**,** 247-255.
SCHLEY, L., SCHAUL, M. & ROPER, T. 2004. Distribution and population density of badgers Meles meles in Luxembourg. *Mammal Review,* 34(3)**,** 233-240.
SCHMIDT, N. M., OLSEN, H., BILDSØE, M., SLUYDTS, V. & LEIRS, H. 2005. Effects of grazing intensity on small mammal population ecology in wet meadows. *Basic and Applied Ecology,* 6(1)**,** 57-66.
SCHOBER, W. 2004. *Barbastella barbastellus (Schreber, 1774) - Mopsfledermaus. Handbuch der Saugetiere Europas, Band 4: Fledertiere, Teil 2: Chiroptera 2: Vespertilionidae 2, Molossidae, Nycteridae: 1071-1091.*
SCHOBER, W. & GRIMMBERGER, E. 1989. *A Guide to Bats of Britain and Europe,* London, Hamlyn.

SCHOFIELD, H. & MORRIS, C. 2000. Ranging behaviour and habitat preferences of female Bechsteins's bat, *Myotis bechsteinii* (Kuhl, 1818), in summer.: Vincent Wildlife Trust.

SCHOFIELD, H. W. 1996. *The ecology and conservation biology of Rhinolophus hipposideros, the lesser horseshoe bat.* PhD, University of Aberdeen.

SCHOFIELD, M. R. & BARKER, R. J. 2008. A unified capture-recapture framework. *Journal of Agricultural, Biological, and Environmental Statistics,* 13(4), 458-477.

SCOTT, D. M., BERG, M. J., TOLHURST, B. A., CHAUVENET, A. L., SMITH, G. C., NEAVES, K., LOCHHEAD, J. & BAKER, P. J. 2014. Changes in the distribution of red foxes (*Vulpes vulpes*) in urban areas in Great Britain: findings and limitations of a media-driven nationwide survey. *PLOS ONE,* 9(6), e99059.

SCOTT, W. A. 2007. *CS Technical Report No.4/07 Statistical Report,* Lancaster, NERC Centre for Ecology and Hydrology.

SCOTTISH NATURAL HERITAGE. 2010. *Expanding Scotland's woods and forests* [Online]. [Accessed 16 November 2016].

SCOTTISH NATURAL HERITAGE 2014. *Scotland's Wild Deer: A National Approach Including 2015-2020 priorities,* Edinburgh, Scottish Natural Heritage.

SCOTTISH NATURAL HERITAGE 2016. *Deer Management in Scotland: Report to the Scottish Government from Scottish Natural Heritage 2016,* Edinburgh, Scottish Natural Heritage.

SEARLE, J. B., JONES, C. S., GUNDUZ, I., SCASCITELLI, M., JONES, E. P., HERMAN, J. S., RAMBAU, R. V., NOBLE, L. R., BERRY, R. J., GIMENEZ, M. D. & JOHANNESDOTTIR, F. 2009. Of mice and (Viking?) men: phylogeography of British and Irish house mice. *Proceedings of the Royal Society B: Biological Sciences,* 276(1655), 201-207.

SENIOR, P., BUTLIN, R. K. & ALTRINGHAM, J. D. 2005. Sex and segregation in temperate bats. *Proceedings of the Royal Society B: Biological Sciences,* 272(1580), 2467-2473.

SENN, H. V. & PEMBERTON, J. M. 2009. Variable extent of hybridization between invasive sika (*Cervus nippon*) and native red deer (*C. elaphus*) in a small geographical area. *Molecular Ecology,* 18(5), 862-876.

SHEDDEN, C. B. 1993. Roe deer in Scotland. *Shooting & Conservation,* (Spring), 43-44.

SHEEHY, E. 2013. *The role of the pine marten in Irish squirrel population dynamics.* PhD, National University of Ireland, Galway.

SHIEL, C. B., DUVERGÉ, P. L., SMIDDY, P. & FAIRLEY, J. S. 1998. Analysis of the diet of Leisler's bat (*Nyctalus leisleri*) in Ireland with some comparative analyses from England and Germany. *Journal of Zoology,* 246(4), 417-425.

SHIEL, C. B. & FAIRLEY, J. S. 1999. Evening emergence of two nursery colonies of Leisler's bat (*Nyctalus leisleri*) in Ireland. *Journal of Zoology,* 247(4), 439-447.

SHIEL, C. B., MCANEY, C. M. & FAIRLEY, J. S. 1991. Analysis of the diet of Natterer's bat *Myotis nattereri* and the common long-eared bat *Plecotus auritus* in the West of Ireland. *Journal of Zoology,* 223(2), 299-305.

SHIRLEY, M. D. F., ARMITAGE, V. L., BARDEN, T. L., GOUGH, M., LURZ, P. W. W., OATWAY, D. E., SOUTH, A. B. & RUSHTON, S. P. 2001. Assessing the impact of a music festival on the emergence behaviour of a breeding colony of Daubenton's bats (*Myotis daubentonii*). *Journal of Zoology,* 254(3), 367-373.

SHORE, R. F., BIRKS, J. D. S., AFSAR, A., WIENBURG, C. L. & KITCHENER, A. C. 2003. Spatial and temporal analysis of second-generation anticoagulant rodenticide residues in polecats (*Mustela putorius*) from throughout their range in Britain, 1992-1999. *Environmental Pollution,* 122(2), 183-193.

SHORE, R. F. & MACKENZIE, S. 1993. The effects of catchment liming on shrews *Sorex spp. Biological Conservation,* 64(2), 101-111.

SHORE, R. F., MEEK, W. R., SPARKS, T. H., PYWELL, R. F. & NOWAKOWSKI, M. 2005. Will Environmental Stewardship enhance small mammal abundance on intensively managed farmland? *Mammal Review,* 35(3-4), 277-284.

SHUTTLEWORTH, C. M., LURZ, P. W. W., GEDDES, N. & BROWNE, J. 2012. Integrating red squirrel (*Sciurus vulgaris*) habitat requirements with the management of

pathogenic tree disease in commercial forests in the UK. *Forest Ecology and Management,* 279(-), 167-175.
SIEMERS, B. M. & SCHNITZLER, H.-U. 2000. Natterer's Bat (*Myotis nattereri* Kuhl, 1818) Hawks for Prey Close to Vegetation Using Echolocation Signals of Very Broad Bandwidth. *Behavioral Ecology and Sociobiology,* 47(6), 400-412.
SIEMERS, B. M. & SCHNITZLER, H.-U. 2004. Echolocation signals reflect niche differentiation in five sympatric congeneric bat species. *Nature,* 429(6992), 657-661.
SIEMERS, B. M. & SWIFT, S. M. 2006. Differences in sensory ecology contribute to resource partitioning in the bats *Myotis bechsteinii* and *Myotis nattereri (Chiroptera: Vespertilionidae)*. *Behavioral Ecology and Sociobiology,* 59(3), 373-380.
SIERRO, A. & ARLETTAZ, R. 1997. Barbastelle bats (*Barbastella* spp.) specialize in the predation of moths: implications for foraging tactics and conservation. *Acta Oecologica,* 18(2), 91-106.
SILVA, A. P., ROSALINO, L. M., JOHNSON, P. J., MACDONALD, D. W., ANDERSON, N. & KILSHAW, K. 2013. Local-level determinants of wildcat occupancy in Northeast Scotland. *European Journal of Wildlife Research,* 59(3), 449-453.
SLUITER, J. W. & VAN HEERDT, P. F. 1966. Seasonal habits of the noctule bat (*Nycalus noctula*). *Archives Neérlandaises de Zoologic,* 16(4), 423-439.
SMITH, D. 1999. *Grey squirrel, Sciurus carolinensis, population dynamics and feeding biology in conifer forest.* PhD, University of London.
SMITH, G. C., AEGERTER, J. N., ALLNUTT, T. R., MACNICOLL, A. D., LEARMOUNT, J., HUTSON, A. M. & ATTERBY, H. 2011. Bat population genetics and Lyssavirus presence in Great Britain. *Epidemiology and Infection,* 139(10), 1463-9.
SMITH, P. G. 2001. *Habitat preference, range use and roosting ecology of Natterer's bats (Myotis nattereri) in a grassland-woodland landscape.* PhD, University of Aberdeen.
SMITH, P. G. & RACEY, P. A. 2005. The itinerant Natterer: physical and thermal characteristics of summer roosts of *Myotis nattereri (Mammalia: Chiroptera)*. *Journal of Zoology,* 266(2), 171-180.
SMITH, P. G. & RACEY, P. A. 2008. Natterer's bats prefer foraging in broad-leaved woodlands and river corridors. *Journal of Zoology,* 275(3), 314-322.
SMITH, R. K., JENNINGS, N. V. & HARRIS, S. 2005. A quantitative analysis of the abundance and demography of European hares *Lepus europaeus* in relation to habitat type, intensity of agriculture and climate. *Mammal Review,* 35(1), 1-24.
SMITH, R. K., JENNINGS, N. V., ROBINSON, A. & HARRIS, S. 2004. Conservation of European hares *Lepus europaeus* in Britain: is increasing habitat heterogeneity in farmland the answer? *Journal of Applied Ecology,* 41(6), 1092-1102.
SOULSBURY, C. D., IOSSA, G., BAKER, P. J., COLE, N. C., FUNK, S. M. & HARRIS, S. 2007. The impact of sarcoptic mange Sarcoptes scabiei on the British fox Vulpes vulpes population. *Mammal Review,* 37(4), 278-296.
SPEAKMAN, J. R., RACEY, P. A., CATTO, C. M. C., WEBB, P. I., SWIFT, S. M. & BURNETT, A. M. 1991. Minimum summer populations and densities of bats in N.E. Scotland, near the northern borders of their distributions. *Journal of Zoology,* 225(2), 327-345.
SPITZ, F. & JANEAU, G. 1990. Spatial strategies: an attempt to classify daily movements of wild boar. *Acta Theriologica,* 35(1-2), 129-149.
STAHL, P. & LEGER, F. 1992. *Le chat sauvage d'Europe (Felis silvestris Schreber, 1777)*, SFEPM, Nort s/Edre (in French).
STAINES, B. W. & RATCLIFFE, P. R. 1987. Estimating the abundance of red deer (*Cervus elaphus* L.) and roe deer (*Capreolus capreolus* L.) and their current status in Great Britain. *Symposia of the Zoological Society of London,* 58(-), 131-152.
STANBURY, A., BROWN, A., EATON, M. A., AEBISCHER, N. J., GILLINGS, S., HEARN, R., NOBLE, D., STROUD, D. & GREGORY, R. D. 2017. The risk of extinction for birds in Britain. *British Birds,* 110), 15.
STAPP, P. 2002. Stable isotopes reveal evidence of predation by ship rats on seabirds on the Shiant Islands, Scotland. *Journal of Applied Ecology,* 39(5), 831-840.

STEVENSON-HOLT, C. D., WATTS, K., BELLAMY, C. C., NEVIN, O. T. & RAMSEY, A. D. 2014. Defining Landscape Resistance Values in Least-Cost Connectivity Models for the Invasive Grey Squirrel: A Comparison of Approaches Using Expert-Opinion and Habitat Suitability Modelling. *PLOS ONE,* 9(11)**,** 11.

STEWART, R. A., CLARK, T. J., SHELTON, J., STRINGFELLOW, M., SCOTT, C., WHITE, S. A. & MCCAFFERTY, D. J. 2017. Urban grasslands support threatened water voles. *Journal of Urban Ecology,* 3(1)**,** 1-7.

STONE, D. R. & GORMAN, M. L. 1985. Social organization of the European mole (*Talpa europaea*) and the Pyrenean desman (*Galemys pyrenaicus*). *Mammal Review,* 15(1)**,** 35-42.

STONE, E., ZEALE, M. R. K., NEWSON, S. E., BROWNE, W. J., HARRIS, S. & JONES, G. 2015. Managing Conflict between Bats and Humans: The Response of Soprano Pipistrelles (*Pipistrellus pygmaeus*) to Exclusion from Roosts in Houses. *PLOS ONE,* 10(8)**,** e0131825.

STONE, E. L., JONES, G. & HARRIS, S. 2009. Street Lighting Disturbs Commuting Bats. *Current Biology,* 19(13)**,** 1123-1127.

STRACHAN, C. & JEFFERIES, D. J. 1993. *The water vole Arvicola Terrestris in Britain 1989-1990: its distribution and changing status.,* Ledbury, The Vincent Wildlife Trust.

STRACHAN, C., STRACHAN, R. & JEFFERIES, D. J. 2000. *Preliminary report on the changes in the water vole population of Britain as shown by the national surveys of the 1989-1990 and 1996-1998,* Ledbury, The Vincent Wildlife Trust.

STRACHAN, R. 2007. *National survey of otter Lutra lutra distribution in Scotland 2003-04: Scottish Natural Heritage Commissioned Report No. 211,* Edinburgh, Scottish Natural Heritage.

STRACHAN, R. 2015. *Otter survey of Wales 2009-10*, Natural Resources Wales.

STRACHAN, R. & MOORHOUSE, T. 2006. *Water vole conservation handbook*, Wildlife Conservation Research Unit, University of Oxford.

SULLIVAN, C. M., SHIEL, C. B., MCANEY, C. M. & FAIRLEY, J. S. 1993. Analysis of the diets of Leisler's *Nyctalus leisleri,* Daubenton's *Myotis daubentoni* and pipistrelle *Pipistrellus pipistrellus* bats in Ireland. *Journal of Zoology,* 231(4)**,** 656-663.

SUMPTION, K. & FLOWERDEW, J. 1985. The ecological effects of the decline in rabbits (*Oryctolagus cuniculus L.*) due to myxomatosis. *Mammal Review,* 15(4)**,** 151-186.

SUTTON, R. T. & DONG, B. 2012. Atlantic Ocean influence on a shift in European climate in the 1990s. *Nature Geoscience,* 5(11)**,** 788-792.

SWIFT, S. & RACEY, P. 2002. Gleaning as a foraging strategy in Natterer's bat *Myotis nattereri*. *Behavioral Ecology and Sociobiology,* 52(5)**,** 408-416.

SWIFT, S. M. 1997. Roosting and foraging behaviour of Natterer's bats (*Myotis nattereri*) close to the northern border of their distribution. *Journal of Zoology,* 242(2)**,** 375-384.

SWIFT, S. M. 1998. *Long-Eared Bats,* London, Poyser Ltd.

SWIFT, S. M. & RACEY, P. A. 1983. Resource partitioning in two species of vespertilionid bats (*Chiroptera*) occupying the same roost. *Journal of Zoology,* 200(2)**,** 249-259.

SYMES, R. & YALDEN, D. 2002. The former status of the black rat *(Rattus rattus*) in England and Wales. *Mammal Review,* 32(4)**,** 315-316.

TAAKE, K. H. 1984. Strukturelle Unterschiede zwischen den Sommerhabitaten von Kleiner und Großer Bartfledermaus (*Myotis mystacinus und M. brandti*) in Westfalen. *Nyctalus,* 2)**,** 16-32.

TAPPER, S. & PARSONS, N. 1984. The changing status of the Brown hare (*Lepus capensis* L.) in Britain. *Mammal Review,* 14(2)**,** 57-70.

TAPPER, S. C. & BARNES, R. F. W. 1986. Influence of Farming Practice on the Ecology of the Brown Hare (*Lepus europaeus*). *Journal of Applied Ecology,* 23(1)**,** 39-52.

TATTERSALL, F. H., AVUNDO, A. E., MANLEY, W. J., HART, B. J. & MACDONALD, D. W. 2000. Managing set-aside for field voles (*Microtus agrestis*). *Biological Conservation,* 96(1)**,** 123-128.

TATTERSALL, F. H., MACDONALD, D. W., HART, B. J., JOHNSON, P., MANLEY, W. & FEBER, R. 2002. Is habitat linearity important for small mammal communities on farmland? *Journal of Applied Ecology,* 39(4)**,** 643-652.
TATTERSALL, F. H., MACDONALD, D. W., HART, B. J., MANLEY, W. J. & FEBER, R. E. 2001. Habitat use by wood mice (*Apodemus sylvaticus*) in a changeable arable landscape. *Journal of Zoology,* 255(4)**,** 487-494.
TATTERSALL, F. H., SMITH, R. H. & NOWELL, F. 1997. Experimental colonisation of contrasting habitats by house mice. *Zeitschrift für Säugetierkunde,* 62(6)**,** 350-358.
TAYSIDE BEAVER STUDY GROUP. 2015. *Tayside Beaver Study Group Final Report* [Online]. Available: https://www.nature.scot/sites/default/files/2017-11/Tayside%20Beaver%20Study%20Group%20-%20%20Final%20Report%202015.pdf [Accessed 01/02/2017].
TELFER, S., DALLAS, J. F., AARS, J., PIERTNEY, S. B., STEWART, W. & LAMBIN, X. 2003. Demographic and genetic structure of fossorial water voles (*Arvicola terrestris*) on Scottish islands. *Journal of Zoology,* 259(1)**,** 23-29.
TELFER, S., HOLT, A., DONALDSON, R. & LAMBIN, X. 2001. Metapopulation processes and persistence in remnant water vole populations. *Oikos,* 95(1)**,** 31-42.
TEMPLE, R., CLARK, S. & HARRIS, S. 2000. The National Hare Survey. PTES: University of Bristol.
TEMPLE, R. & MORRIS, P. 1997. The lesser white-toothed shrew on the Isles of Scilly. *British Wildlife,* 9(-)**,** 94-99.
TEW, T. E. & MACDONALD, D. W. 1993. The effects of harvest on arable wod mice *Apodemus sylvaticus*. *Biological Conservation,* 65(3)**,** 279-283.
THIRGOOD, S. J. 1996. Ecological factors influencing sexual segregation and group size in fallow deer (*Dama dama*). *Journal of Zoology,* 239(4)**,** 783-797.
THIRGOOD, S. J. & HEWSON, R. 1987. Shelter characteristics of mountain hare resting sites. *Holartic Ecology,* 10(4)**,** 294-298.
THORBURN, A. 1920. *British Mammals,* London, Longmans, Green & Co.
THULIN, C. G., TEGELSTROM, H. & FREDGA, K. 2003. Haplotype-diversity of mountain hare mtDNA among native mountain-hares and introduced brown hares in Scandinavia. *Annales Zoologici Fennici,* 40(1)**,** 45-52.
THURFJELL, H., BALL, J. P., ÅHLÉN, P.-A., KORNACHER, P., DETTKI, H. & SJÖBERG, K. 2009. Habitat use and spatial patterns of wild boar *Sus scrofa* (L.): agricultural fields and edges. *European Journal of Wildlife Research,* 55(5)**,** 517-523.
TINK, M., BURNSIDE, N. G. & WAITE, S. 2014. A Spatial Analysis of Serotine Bat (*Eptesicus serotinus*) Roost Location and Landscape Structure: A Case Study in Sussex, UK. *International Journal of Biodiversity,* 2014(-)**,** 9.
TRENKEL, V. M., PARTRIDGE, L. W., GORDON, I. J., BUCKLAND, S. T., ELSTON, D. A. & MCLEAN, C. 1998. The management of red deer on Scottish open hills: Results of a survey conducted in 1995. *Scottish Geographical Magazine,* 114(1)**,** 57-62.
TREWBY, I. D., YOUNG, R., MCDONALD, R. A., WILSON, G. J., DAVISON, J., WALKER, N., ROBERTSON, A., DONCASTER, C. P. & DELAHAY, R. J. 2014. Impacts of Removing Badgers on Localised Counts of Hedgehogs. *PLOS ONE,* 9(4)**,** e95477.
TROUT, R., BROOKS, S. & RUDLIN, P. 2012. Hazel dormice in British conifer forests and their ecology in a pine plantation during restoration to broadleaf. *Peckiana,* 8(-)**,** 31-39.
TROUT, R. C., LANGTON, S., SMITH, G. C. & HAINES-YOUNG, R. H. 2000. Factors affecting the abundance of rabbits (*Oryctolagus cuniculus*) in England and Wales. *Journal of Zoology,* 252(2)**,** 227-238.
UK NATIONAL ECOSYSTEM ASSESSMENT 2011. *The UK National Ecosystem Assessment Technical Report*. Cambridge: UNEP-WCMC.
URBANCZYK, Z. 1990. Northern Europe's most important bat hibernation site. *Oryx,* 24(1)**,** 30-4.
UZAL FERNANDEZ, A. 2010. *The interaction of Sika deer (Cervus nippon Temminck 1838) with lowland heath mosaics.* PhD, Bournemouth University.
VAUGHAN, N. 1997. The diets of British bats (*Chiroptera*). *Mammal Review,* 27(2)**,** 77-94.

VAUGHAN, N., JONES, G. & HARRIS, S. 1996. Effects of sewage effluent on the activity of bats (*Chiroptera: Vespertilionidae*) foraging along rivers. *Biological Conservation,* 78(3)**,** 337-343.

VINCENT WILDLIFE TRUST 2014. *Polecats and Ferrets. How to tell them apart.,* Ledbury, Vincent Wildlife Trust.

VON HELVERSEN, O., HELLER, K. G., MAYER, F., NEMETH, A., VOLLETH, M. & GOMBKÖTÖ, P. 2001. Cryptic mammalian species: a new species of whiskered bat (*Myotis alcathoe n. sp.*) in Europe. *Naturwissenschaften,* 88(5)**,** 217-223.

WABER, K. & DOLMAN, P. M. 2015. Deer abundance estimation at landscape-scales in heterogeneous forests. *Basic and Applied Ecology,* 16(7)**,** 610-620.

WAITS, L. P. & PAETKAU, D. 2005. Noninvasive genetic sampling tools for wildlife biologists: a review of application and recommendations for accurate data collection. *Journal of Wildlife Management,* 69(4)**,** 1419-1433.

WALSH, J. & HALL, C. 2005. *The status of water voles in two upland catchments in Wales. Contract No: FC 73-01-537.* Countryside Council for Wales.

WARD, A. I. 2005. Expanding ranges of wild and feral deer in Great Britain. *Mammal Review,* 35(2)**,** 165-173.

WARD, A. I., ETHERINGTON, T. & EWALD, J. 2008. Five years of change. *Deer,* 14(8)**,** 17-20.

WARD, A. I., WHITE, P. C. L. & CRITCHLEY, C. H. 2004. Roe deer *Capreolus capreolus* behaviour affects density estimates from distance sampling surveys. *Mammal Review,* 34(4)**,** 315-319.

WARING, S. D., ESSAH, E., GUNNELL, K. & BONSER, R. 2013. Double jeopardy: the potential for problems when bats interact with breathable roofing membranes in the United Kingdom. *Architecture and Environment,* 1(-)**,** 1-3.

WARREN, R. D., WATERS, D. A., ALTRINGHAM, J. D. & BULLOCK, D. J. 2000. The distribution of Daubenton's bats (*Myotis daubentonii*) and pipistrelle bats (*Pipistrellus pipistrellus*) (*Vespertilionidae*) in relation to small-scale variation in riverine habitat. *Biological Conservation,* 92(1)**,** 85-91.

WATERS, D., JONES, G. & FURLONG, M. 1999. Foraging ecology of Leisler's bat (*Nyctalus leisleri*) at two sites in southern Britain. *Journal of Zoology,* 249(2)**,** 173-180.

WAUTERS, L. A., LURZ, P. W. W. & GURNELL, J. 2000. Interspecific effects of grey squirrels (*Sciurus carolinensis*) on the space use and population demography of red squirrels (*Sciurus vulgaris*) in conifer plantations. *Ecological Research,* 15(3)**,** 271-284.

WEBBON, C. C., BAKER, P. J. & HARRIS, S. 2004. Faecal density counts for monitoring changes in red fox numbers in rural Britain. *Journal of Applied Ecology,* 41(4)**,** 768-779.

WEIR, A., MCLEOD, J. & ADAMS, C. E. 1995. The winter diet and parasitic fauna of a population of Rednecked Wallabies *Macropus rufogriseus* recently introduced to Scotland. *Mammal Review,* 25(3)**,** 111-116.

WELCH, D., STAINES, B. W., CATT, D. C. & SCOTT, D. 1990. Habitat usage by red (*Cervus elaphus*) and roe (*Capreolus capreolus*) deer in a Scottish Sitka spruce plantation. *Journal of Zoology,* 221(3)**,** 453-476.

WEMBRIDGE, D., AL-FULAIJ, N. & LANGTON, S. 2016a. *The state of Britain's dormice 2016*. London: People's Trust for Endangered Species.

WEMBRIDGE, D., NEWMAN, M. R., BRIGHT, P. W. & MORRIS, P. A. 2016b. An estimate of the annual number of hedgehog (*Erinaceus europaeus*) road casualties in Great Britain. *Mammal communications,* 2(-)**,** 8-14.

WHITE, T. A. & SEARLE, J. B. 2007. Genetic diversity and population size: island populations of the common shrew, *Sorex araneus*. *Molecular Ecology,* 16(10)**,** 2005-2016.

WILLIAMS, C. 2001. *The winter ecology of Rhinolophus hipposideros, the lesser horseshoe bat.* PhD, Open University.

WILSON, C. J. Wild boar in south west England. *In:* C., T. & N., K., eds. Proceedings of the International Union of Game Biologists XXIVth Congress, 1999 Thessaloniki, Greece. 490-492.

WILSON, C. J. 2003. Distribution and status of feral wild boar *Sus scrofa* in Dorset, southern England. *Mammal Review,* 33(3-4), 302-307.
WILSON, G., HARRIS, S. & MCLAREN, G. 1997. *Changes in the British Badger Population 1988–1997,* London, People's Trust for Endangered Species.
WOHLGEMUTH, R., DEVRIENT, I., GARCÍA, A. & HUTTERER, R. 2004. Long distance flight of a Lesser noctule (*Nyctalus leisleri*) after rehabilitation. *Myotis,* 41-42(-), 69-73.
WOLTZ, I. 1992. *Zur Ökologie der Bechsteinfledermaus Myotis bechsteinii (Kuhl, 1818) (Mammalia: Chiroptera).* PhD, University of Erlangen (in German).
WOODS, M., MCDONALD, R. A. & HARRIS, S. 2003. Predation of wildlife by domestic cats *Felis catus* in Great Britain. *Mammal Review,* 33(2), 174-188.
WRIGHT, A., FIELDING, A. H. & WHEATER, C. P. 2000. Predicting the distribution of Eurasian badger (*Meles meles*) setts over an urbanized landscape: a GIS approach. *Photogrammetric Engineering and Remote Sensing,* 66(4), 423-428.
WRIGHT, L. J., NEWSON, S. E. & NOBLE, D. G. 2014. The value of a random sampling design for annual monitoring of national populations of larger British terrestrial mammals. *European Journal of Wildlife Research,* 60(2), 213-221.
WRIGHT, P. G., HAMILTON, P. B., SCHOFIELD, H., GLOVER, A., DAMANT, C., DAVIDSON-WATTS, I. & MATHEWS, F. 2018. Genetic structure and diversity of a rare woodland bat, *Myotis bechsteinii*: comparison of continental Europe and Britain. *Conservation Genetics,* (13th Feb), 1-11.
YALDEN, D. 1999. *The history of British mammals,* London, Poyser.
YALDEN, D. W. 1992. Changing distribution and status of small mammals in Britain. *Mammal Review,* 22(2), 97-106.
YAMAGUCHI, N., RUSHTON, S. & MACDONALD, D. W. 2003. Habitat preferences of feral American mink in the Upper Thames. *Journal of Mammalogy,* 84(4), 1356-1373.
YOUNG, R. P., DAVISON, J., TREWBY, I. D., WILSON, G. J., DELAHAY, R. J. & DONCASTER, C. P. 2006. Abundance of hedgehogs (*Erinaceus europaeus*) in relation to the density and distribution of badgers (*Meles meles*). *Journal of Zoology,* 269(3), 349-356.
ZAHN, A., ROTTENWALLNER, A. & GÜTTINGER, R. 2006. Population density of the greater mouse-eared bat (*Myotis myotis*), local diet composition and availability of foraging habitats. *Journal of Zoology,* 269(4), 486-493.
ZALEWSKI, A. & JEDRZEJEWSKI, W. 2006. Spatial organisation and dynamics of the pine marten *Martes martes* population in Bialowieza Forest (E Poland) compared with other European woodlands. . *Ecography* 29(1), 31-43.
ZARZOSO-LACOSTE, D., JAN, P. L., LEHNEN, L., GIRARD, T., BESNARD, A. L., PUECHMAILLE, S. J. & PETIT, E. J. 2017. Combining noninvasive genetics and a new mammalian sex-linked marker provides new tools to investigate population size, structure and individual behaviour: an application to bats. *Molecular Ecology Resources,* 12(2), 217-228.
ZEALE, M. R., BENNITT, E., NEWSON, S. E., PACKMAN, C., BROWNE, W. J., HARRIS, S., JONES, G. & STONE, E. 2016. Mitigating the Impact of Bats in Historic Churches: The Response of Natterer's Bats *Myotis nattereri* to Artificial Roosts and Deterrence. *PLOS ONE,* 11(1), e0146782.
ZEALE, M. R. K. 2011. *Conservation biology of the barbastelle (Barbastella barbastellus): applications of spatial modelling, ecology and molecular analysis of diet.* PhD, University of Bristol.
ZEALE, M. R. K., DAVIDSON-WATTS, I. & JONES, G. 2012. Home range use and habitat selection by barbastelle bats (*Barbastella barbastellus*): implications for conservation. *Journal of Mammalogy,* 93(4), 1110-1118.
ZYLSTRA, E. R., STEIDL, R. J. & SWANN, D. E. 2010. Evaluating Survey Methods for Monitoring a Rare Vertebrate, the Sonoran Desert Tortoise. *Journal of Wildlife Management,* 74(6), 1311-1318.