BTO Research Report 384

The London Bird Project

Authors

Dan Chamberlain, Su Gough, Howard Vaughan, Graham Appleton, Steve Freeman, Mike Toms, Juliet Vickery, David Noble

A report to The Bridge House Estates Trust

January 2005

British Trust for Ornithology, The Nunnery, Thetford, Norfolk IP24 2PU
Registered Charity No. 216652

British Trust for Ornithology

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Published in January 2005
by
the British Trust for Ornithology
The Nunnery, Thetford, Norfolk, IP24 2PU, UK

ISBN 1-904870-18-X

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Executive Summary

1. Relatively little is known about the ecology of birds in cities and suburbs of the UK. However, following recent declines in urban populations of several species, notably House Sparrow and Starling, the study of urban birds is now a major conservation issue. Parks and gardens are the main contributors to urban biodiversity, and an understanding of the factors that make these habitats more attractive to birds would both greatly enhance our ability to improve urban habitats for wildlife and add to our knowledge of this generally neglected area of ecology.

2. The distributions and population trends for several species are known reasonably well in Greater London and there are several studies that have considered the bird communities of London's green spaces. Whilst our knowledge of London's bird populations is good, we know relatively little about the factors that actually determine the presence and abundance of species within London's parks, private gardens and other green spaces. A thorough assessment of habitat associations is needed if we are to improve the bird diversity of these green spaces. This can be achieved through analysis of bird data in conjunction with spatially referenced habitat data, the ultimate aim being the development of management recommendations.

3. The British Trust for Ornithology's London Bird Project was largely funded by the Bridge House Trust, the grant giving arm of the Corporation of London, to assess the value of the city's green spaces (including private gardens) for birds and to suggest ways of managing them for the benefit of birds. The project had four main aims: (i) to determine the bird species richness and density of individual species in a large sample of public green spaces; (ii) to relate these measures to habitat variables in both summer and winter, in order to understand how habitat type, size and structure affect bird communities in urban green spaces; (iii) to identify the broad composition of urban bird communities and to determine how the breeding distributions of individual species relate to large-scale distribution of habitat throughout the London area; (iv) to detect patterns in the usage of gardens by birds throughout the year with respect to garden habitat.

4. The aims were achieved using three data sources, one from a novel survey and two from existing long-term monitoring schemes run by the BTO. A survey of public green spaces was carried out from summer 2002 to winter 2003/04 which counted every individual bird located within each site. Habitat data were also collected so bird density could be related to habitat availability. Data for a wider range of habitats is collected in the BTO/ JNCC/RSPB Breeding Bird Survey (BBS). BBS data were analysed with respect to broad-scale habitat data within Greater London. The BTO/CJ Wildbird Foods Garden BirdWatch (GBW) is a weekly volunteer survey that records presence/absence of selected species in gardens throughout the year. Trends over time and habitat associations of gardens birds within Greater London were investigated using these data.

Green Spaces Survey

5. Volunteer ornithologists within Greater London were recruited to survey public access green spaces with a maximum area of 80ha. Site selection was made by the volunteers from a predefined list that excluded farmland, designated nature reserves and sites that did not have free public access.

6. Each site received six visits, three in spring/summer (April-July) and three in the winter (October-February) of each of two years. At each visit, observers were asked to record numbers of all bird species seen and heard on the site, taking note of the habitat type in which they were first detected. For each site, a simple habitat description was carried out. Surveyors also counted squirrels on each visit.

7. Species richness, individual species density and individual species probability of occurrence were analysed in relation to habitat. Habitat associations were considered both at the site level and at the habitat patch level. The former considered effects of habitat extent or presence on species richness, at the whole site level, including both internal and external habitat features. The analytical goal was to identify those features that determine species richness, and the density and probability of occurrence of individual species per site. The patch level analysis aimed to identify those habitat patches that held the most species or highest density of individual species within each site.

8. A total of 301 sites were covered. Most sites were classified as parks (n =142). Other sites types were: cemeteries (n = 57), public gardens (n = 29), recreation grounds (n = 11), open space (n = 12), woods (n = 22), playing fields (n = 9), squares (n = 7), commons (n = 4), meadows (n = 3), scrub =(n = 2), allotment and marsh (n = 1 each) and 1 unclassified. Sites were defined according to location in inner or outer boroughs. Sites in outer boroughs were significantly larger and comprised a greater proportion of cemeteries than inner boroughs. The latter comprised a greater proportion of squares and public gardens and tended to be subject to higher traffic levels and human disturbance than outer borough sites.

9. A total of 90 target species were recorded in the two years of the survey, 85 in the summer and 75 in the winter (most wildfowl species were not included in the survey). Overall, outer boroughs had significantly more species than inner boroughs, but this reflected the fact that sites tended to be larger in outer boroughs. In terms of site type, woods and parks had the highest richness and public gardens the lowest overall. There was a highly significant positive effect of site area on species richness and individual species abundance. When corrected for area, species richness was higher in inner than outer boroughs. This result demonstrates that it should not be assumed that inner city green spaces are of less value than those in outer London.

10. At the site level some general patterns emerged. Species richness and the density of several individual species were positively associated with buildings (Feral Pigeon, Woodpigeon, Wren, Robin, Blackbird, Blue Tit, Long-tailed Tit and Magpie), deciduous bushes (Feral Pigeon, Wren, Blackbird, Blue Tit, Starling, Carrion Crow and Magpie) and negatively associated with coniferous trees and/or coniferous bushes (Blackbird, Starling, Magpie, Greenfinch, House Sparrow). The positive association with buildings is likely to represent inter-correlated factors rather than a causal effect. For example, deciduous bushes, deciduous trees and mown grass covered a greater proportion of site area on sites with buildings.

11. At the patch level many species were found at their highest density in deciduous trees (Woodpigeon, Greenfinch, Jay, Blue Tit, Long-tailed Tit, Great Tit and Song Thrush) and deciduous bushes (Wren, Dunnock, Robin, Mistle Thrush, Blackcap, House Sparrow and Chaffinch).

12. Deciduous bushes in particular seemed important for a number of species. When site-level comparisons were made between sites with trees, mown grass and bushes and sites with trees and mown grass but without bushes, there were significant differences for several species (Feral Pigeon, Woodpigeon, Blackbird, Redwing, Blue Tit, Starling, House Sparrow and Greenfinch). Clearly for some species, bushes represent an additional resource for birds and their presence can increase densities quite considerably (many by a factor of 2 or more).

13. Habitat diversity was determined per site using the Shannon index. There were very few significant associations between species richness/density and habitat diversity. The few significant associations were non-linear in form. This result suggests that promoting habitat diversity *per se* may not necessarily enhance bird species diversity. Rather than maximise habitats, it is probably better to concentrate on management strategies that promote a smaller number of the better ‘bird’ habitats to increase bird abundance/diversity.

14. Squirrel numbers were positively correlated with the proportion of adjacent gardens making up the site boundary, possibly indicating an association with food (especially supplementary bird food) available in gardens. Squirrels were also significantly correlated with the cover of coniferous trees, but not deciduous trees.

Core Breeding Survey

15. A sub-sample of the green spaces survey sites (n = 81) was surveyed at a higher level of intensity in the breeding season, by recording activity codes of birds seen. Noting behaviour such as singing, feeding young or carrying nesting material enabled a more accurate assessment of whether a given species was likely to be attempting to breed at a given site. Density of individual potential breeders was analysed in the same way as in the main survey.

16. On this sub-sample of sites, there were 43 breeding species in 2002 and 55 in 2003. A total of 64 species were recorded, 58 of these showing breeding evidence. There were few species where density was correlated with habitat variables at the site level. Habitat variables significant in more than one species included pavement (Robin, Wren and Blackcap all showed significantly higher density where pavement was present at the boundary) and buildings (Woodpigeon, Starling and House Sparrow density was higher where buildings were present). Most associations were weakly significant ($0.05 < P < 0.01$).

17. At the patch level, there was no significant difference in breeding density between different habitat types for most species. There was greater variation for occurrence rates. Generally, mixed and deciduous trees had high occurrence rates for many species (Woodpigeon, Green Woodpecker, Wren, Dunnock, Robin, Blackbird, Blackcap, Chiffchaff, Blue Tit, Great Tit, Long-tailed Tit, Jay, Chaffinch and Greenfinch). There were also some species that showed high occurrence rates on mown grass (Collared Dove, Mistle Thrush, Magpie, House Sparrow).

18. There were relatively few habitats in which there was evidence of breeding detected. This in part may be due to bias in detecting behavioural evidence of breeding (e.g. singing birds, which comprised most of the records, may often select specific locations such as song posts that aren't necessarily reflective of wider ecological requirements).

Trends and Habitat Associations in the Wider Countryside

19. The BBS is the main annual monitoring scheme for relatively common terrestrial breeding birds in the UK (Raven *et al*. 2003). It is carried out annually on a stratified random sample of 1-km squares. For each square, two parallel 1-km long transects are identified. All birds detected within 100m of each transect are recorded along with the main habitat type on every 200m section of these transects.

20. There is a sufficient sample of BBS squares in Greater London to enable trends to be estimated for 16 species. The most recent trends published suggest that London's birds are going through a period of major change. There were significant increases between 1994 and 2003 in Woodpigeon, Collared Dove, Wren, Robin, Blue Tit, Great Tit, Magpie, Carrion Crow and Greenfinch. However, there were significant decreases in Blackbird, Song Thrush, Starling and House Sparrow.

21. BBS data were used to describe the wider bird community of Greater London, to assess bird-habitat associations and also to consider bird population trends over time in relation to habitat. There were between 35 and 64 1-km squares covered in the BBS between 1994 and 2002 and a total of 83 different squares were covered. In total there were 103 species (and 2 hybrids) recorded during BBS in the period 1994-2002 in Greater London.

22. Habitat associations were considered by using data collected during the course of BBS and also using remotely-sensed land cover data provided by the Centre for Ecology and Hydrology. Species richness and the abundance and occurrence of several individual species were highest in suburban habitats, illustrating the value of suburban gardens and parks compared to urban and rural habitats (farmland, grassland and woodland). Urban cover and farmland cover were both negatively associated with species richness. Heavily built environments held the lowest richness and individual species abundance/occurrence with two exceptions, Starling and House Sparrow, which occurred at higher densities in urban and suburban habitats than in rural habitats.

23. Habitat-specific population trends were considered between 1994 and 2002. Annual trends differed between habitat types. Increasing species often showed the greatest gains in urbanized or built-up habitats (e.g. Feral Pigeon, Woodpigeon, Blue Tit). Habitat-specific declines may prove to be very important for declining species. House Sparrow showed declines in parkland and farmland. Furthermore, Starling in woodland and Blackbird in urban parks both showed clear steady declines relative to other habitats.

Trends and Habitat Associations in Private Gardens

24. Temporal trends of bird occurrence (derived from presence/absence data) in 953 gardens in Greater London were analysed using data from GBW. Although the populations of several species, including Wood Pigeon, Collared Dove, Blackbird, Blue Tit, Carrion Crow, Magpie and Greenfinch, appeared stable between 1995 and 2002, there were some species that showed clear temporal trends over this time period. In particular, Starling and House Sparrow showed significant declines and Feral Pigeon showed a significant increase.

25. The habitat associations found were complicated and varied from year to year for most species. There were relatively few cases where habitats had consistent effects on species occurrence in all years considered. For many species the number of large coniferous trees and the cover of hedges in the garden boundary were consistent determinants of species presence (Sparrowhawk, Wren, Blackcap, Goldcrest, Long-tailed Tit, Chaffinch and Goldfinch). More species were found in larger gardens and gardens with a greater cover of lawn, although the latter is probably merely reflective of the former associations.

26. The surrounding landscape is also important in determining species presence in a given green space. Many species were, for example, more closely associated with gardens that were in more rural settings (close to arable farmland, grassland and woodland). Species richness was also greater in rural and suburban gardens compared to gardens in urban landscapes.

Statistical Techniques for Analysing Categorical Count Data

27. For the ten most commonly encountered species in GBW, additional, more detailed information on the counts of birds were available, where counts are expressed as categories (e.g. 1-4 birds, 5-8 birds etc.). Novel analytical techniques (ordinal regression and interval-censored Poisson regression) were used to determine bird-habitat associations of garden birds. Specific examples were analysed for four species to illustrate the more detailed habitat associations that may be derived. The technique allows an estimation of mean abundance in different habitat categories if an underlying Poisson distribution is assumed. These statistical techniques have great potential in examining specific hypotheses about effects of particular habitat types on abundance in the commoner species recorded in GBW.

28. Trends over time were derived from these count species from GBW and compared with BBS trends to assess whether temporal trends in gardens are equivalent to those in the wider countryside. The matching between BBS trends and GBW trends was fairly close suggesting

that GBW count data are probably generally good monitors of population trends for the ten species analysed. The fit was especially good for House Sparrow.

Comparison of Surveys

29. Occurrence rates were compared between three surveys: London Bird Project (LBP), Breeding Bird Survey (BBS) and Garden BirdWatch (GBW). Compared to LBP, in most cases, occurrence was higher in BBS. Occurrence rates were generally more similar between LBP and GBW.

30. A comparison of densities found that LBP density far exceeded that for BBS in virtually every species. However, when the decline in detectability at increasing distance from BBS transect lines was modelled using Distance software (Buckland *et al*. 1994) and only BBS parkland habitat data were used, densities were broadly similar in most species. It is suggested that any future comparison of surveys should use Distance software when possible.

Overview

31. This project has covered a range of public green spaces, many of which will not have been surveyed for birds previously. We now have knowledge of the range of species that occur in London's green spaces in winter and summer, including those species that are likely to be breeding. Parallel analyses of pre-existing data sets have enabled green spaces to be placed in a wider context of a range of habitats (BBS) and compared to a specific habitat, gardens (GBW). Specific management recommendations have arisen from this project which are given in a separate section (see below).

32. This project has identified a number of key research areas that would further enhance our knowledge of the biodiversity value of green spaces and of the urban environment in general. These are: An assessment of the effects of the structural features of bushes that are of most importance to birds for nesting habitat, foraging habitat and general cover; A study of the placement of bushes (adjacent habitats, value as understorey or without trees); An assessment of the resources available to birds in native versus non-native trees and bushes; Intensive studies of reproductive performance, feeding ecology and movements within and between urban greens spaces. Furthermore, it should be noted that this study considers only birds and draws no conclusions about wider biodiversity. It would be extremely valuable to survey green spaces for a wide range of other taxa in order to identify potential conflicts or similarities in habitat requirements and management approaches.

33. The BTO London Bird Project has gathered a valuable ecological resource. The information collected for this project will be made available to interested parties, who will hopefully use it to enhance or reinforce their current management strategies. The survey should also act as a baseline from which to compare future trends and assess the effects of any habitat or wider environmental change. Finally, the project has also recruited a valuable human resource. Over 225 volunteers took part in the survey and GBW membership increased from about 400 to 1031 in Greater London during the project. It is hoped that the London Bird Project has increased awareness of bird conservation issues, provided a better understanding of ecology, and enhanced appreciation of the value of local green spaces and the threats they may face.

Management Recommendations

34. Deciduous trees were clearly very important habitats for a range of species. Planting of deciduous trees will certainly increase bird abundance and species richness in urban green spaces. It seems likely that native species will accrue wider biodiversity benefits and they should be preferred where possible.

35. Deciduous bushes were particularly important habitats for certain smaller passerines. Most notable amongst the individual species was House Sparrow, a red-listed species of conservation concern. Planting deciduous bushes is likely to increase the abundance of several species. For many species, both deciduous bushes and trees were important habitats so planting bushes as understorey may be of benefit, especially for Wren (breeding season), Robin, Blackbird, tits, Starling, Chaffinch and Greenfinch. Other species showed clearly the highest densities in deciduous bushes but were not apparently closely associated with deciduous trees. For Dunnock, Blackcap, Whitethroat and House Sparrow bushes alone are likely to be beneficial. In common with deciduous trees, native species are probably desirable, bearing in mind wider biodiversity considerations.

36. Areas of dense cover may be perceived as a potentially threatening habitat to some members of the general public (due to provision of cover for potential attackers). The practice of bush clearance could have major repercussions for London's birds, not least the House Sparrow. It is suggested that effects of bush removal could be mitigated through better park design, where bushes are planted and encouraged away from paths and perhaps at site edges or in areas that are less well used by the general public.

37. In comparison with deciduous plants, coniferous trees and bushes had little value for birds. Only the Goldcrest and Coal Tit had particular associations with coniferous trees. Planting conifers therefore appears to confer little benefit to many species and deciduous trees and bushes should usually be preferred unless these species are being targeted. There are possible benefits to conifer belts as roost sites in winter for certain species. An assessment of use of conifers in winter evenings (which was outside the scope of this study) should be carried out before any planned removal of conifers.

38. Mown grass was a well used foraging habitat for a number of species. Maintenance of mown areas should be part of any management plan for birds. There was little evidence that rough grass/nettles were used by any species to any great extent. However, it would be unwise to exclude these habitats from urban green space as there is evidence from other BTO studies that they may act as a reservoir for foliage invertebrates. Patches of rough grass/nettles adjacent to mown areas may provide a good combination of prey abundance and prey accessibility.

39. Buildings are likely to provide nesting habitat for Starlings and House Sparrows, especially where there are plenty of cavities.

40. The London Bird Project has shown the wider value of both public green spaces and private gardens to the Greater London bird community. These habitats, or the elements that make them such great places for birds, need to be protected and enhanced.

CHAPTER 1 BIRDS AND HABITAT IN LONDON'S PUBLIC GREEN SPACES: RESULTS OF THE LONDON BIRD PROJECT

Dan Chamberlain, Su Gough, Howard Vaughan, Juliet Vickery, Graham Appleton

1. INTRODUCTION

Birds are the most frequent of urban wildlife encountered by most people, yet despite their popularity (an estimated 75% of households provide food for birds in their gardens – Cowie & Hinsley 1988), relatively little is known about the ecology of birds in cities and suburbs of the UK. Recent population declines detected in a number of common species (Raven *et al*. 2003) have made the study of urban birds a major conservation issue. Parks and gardens are the main contributors to urban biodiversity, and an understanding of the factors that make these habitats more attractive to birds would greatly enhance our ability to improve urban habitats for wildlife and would add to our knowledge of this generally neglected area of ecology.

Urban parks can often hold a relatively rich bird community compared to other urban habitats (Fernandez-Juricic & Jokimaki 2001) and in exceptional cases can be indistinguishable from communities in surrounding natural habitat (Gavareski 1976). Even small parks in highly urbanized areas can support exceptionally high densities of certain species (Tomialojc 1978). For example, small urban parks in Wrocław (Poland) had breeding bird densities twice as high on average as in the richest nearby forest (Tomialojc & Profus 1977). Jokimäki (1999) considered the habitat features within parks and the surrounding landscape that were correlated with species richness and the probability of species occurrence. Park area was the most important determinant of species richness but was less influential on the occurrence of individual species where habitat had strong influence in a number of species. For example, occurrence of Pied and Spotted Flycatcher[1] were related to tree diversity, Fieldfare was related to tree height and Willow Warbler and Magpie were related to park management (where occurrence was higher in unmanaged sites). Furthermore, the landscape surrounding the park was also important. For example, larger built areas surrounding the study park negatively correlated with Willow Warbler, Hooded Crow, Pied and Spotted Flycatcher occurrence. This study shows that both features within and surrounding a given park can be important determinants of its bird community.

Parks are probably the most studied urban green space in terms of bird communities. Less research has been carried out on other types of public green space such as squares, cemeteries and playing fields. In London, Baker (1988) presented results of a survey of small open spaces carried out by volunteers from London Natural History Society. A total of 20 sites were surveyed throughout the year, all being <3ha in area, yet there were 37 species recorded. Even within this small range, area had a clear positive effect on species richness. Sites with the greatest number of breeding species tended to be those where there were high areas of cover. 'Wild' areas and untended flower beds were also of apparent benefit. A more comprehensive survey of the City of Westminster was carried out by Hewlett *et al*. (1995) who provided very detailed information on biodiversity on most green spaces within the borough, including a large number of town squares.

Hewlett (2002) suggests that parkland is essentially akin to open woodland and woodland edge and supports a similar bird community. There have been few detrimental changes to London's parks over the past three decades (e.g. conversion of playing fields to artificial all-weather surfaces) and these may have been off-set by an increasing awareness of biodiversity issues and provision of wild areas and the planting of native trees and bushes. Nevertheless, Hewlett (2002) documents some major changes in London's bird populations in parks, including increases in Green and Great Spotted Woodpeckers, Collared Dove, Long-tailed Tit and Magpie. However there have also been decreases, most notably Spotted Flycatcher and House Sparrow.

[1] Scientific names of all species mentioned in the report are given in Appendix I

The distributions and population trends for several species are known reasonably well in Greater London and there are several studies which have considered the bird communities of London's green spaces. However, in the latter case, these tend to be borough-specific. Also, whilst our knowledge of London's bird populations is good, we know relatively very little about the factors that actually dictate the presence and abundance of species within London's green spaces. A thorough assessment of habitat associations is needed if we are to improve the biodiversity value of these green spaces. This can be achieved through analysis of bird data in conjunction with spatially referenced habitat data, which ultimately should lead to management recommendations.

2. AIMS

The aims of this chapter are: (i) to describe summer and winter bird communities in the different types of public green space in Greater London; (ii) to assess how seasonal influences dictate use of green spaces; (iii) to assess which habitat features are of most importance in determining species richness and individual species density and occurrence in public green spaces in Greater London.

2.1 Site Selection

The sites used in the survey were all public green spaces within the Greater London area. Larger parks were not included, partly because they are relatively well known but also because relatively small discrete areas were preferred for the survey (and it would have been difficult if not impossible to cover all of some of the larger parks). Therefore, an upper limit of 80ha was put on the site selection. An initial list of over 1000 green spaces in Greater London was made with reference to *The Master Atlas of Greater London* (Anon 2001) and *London: The Photographic Atlas* (Anon 2000).

Volunteer recruitment was through several channels including the BTO's Regional Representative, BTO News, publicity at conferences and events (e.g. The London Birdwatching Fair at Lee Valley Park, The London Birdwatching Conference at Barn Elms nature reserve), in newspapers and also through talks at bird clubs and other organizations. Volunteers were allowed to select a site from the initial list. In a number of cases, after an initial visit, the surveyor found that the site was either not suitable (e.g. not a true free-access green space such as a nature reserve, farmland etc.) or that access was not possible. In such cases a replacement site was provided, but there were some volunteers who opted to survey some atypical sites (see below).

3. METHODS

There were two levels of survey intensity. The main survey comprised a general site description and total bird count for each site. The core survey involved a more complex mapping approach where activity of the birds was recorded, following standard codes (Bibby *et al*. 2000) so a more accurate determination of likely breeding birds was possible. However, any species detected was recorded (whether showing breeding behaviour or not) and habitat was recorded in the same way as the main survey, making the two methodologies compatible. In this chapter, both main and core survey data (all species recorded) are analysed together. Analysis of birds showing evidence of breeding in the core survey is presented in Chapter 2.

Each site had six visits per year, three in spring/summer (April-July) and three in the winter (October-February). At each visit, observers were asked to record numbers of any species seen and heard on the site, taking note of the habitat type in which they were first detected. All species using the site were included with the exception of most wildfowl species, many of which are exotic species commonly kept in a semi-feral state. There were three exceptions: Mute Swan, Canada Goose and Mandarin. A bird was considered not using a site if it was in flight and obviously not interacting with the site (so birds hunting over the site such as Kestrels or hirundines would be included). Observers were asked to try to walk a survey route so they came within at least 50m of every part of the site, taking precautions to minimise the risk of double-counting.

For each site, a simple habitat description was carried out. Surveyors were also encouraged to record squirrels (count), nest boxes, ant hills and mole hills (presence/absence) at each visit (the former were not unfortunately counted on core sites due to an omission in the instructions). Furthermore, observers were asked to make simple assessments of human disturbance and traffic noise levels adjacent to the site (ranked as high, medium or low).

3.1 Analysis

Data were analysed with respect to differing levels of location and habitat. At the simplest level, comparisons were made between different regions of Greater London defined according to inner and outer boroughs, referred to henceforward as 'location' (Fig. 1.1) and between different site types (defined as cemetery, parks, playing fields, public gardens, woods, squares, recreation grounds and miscellaneous). These comparisons were made at the whole site level.

There were 38 habitat types recorded (Table 1.1). However, in many cases, these categories were scarce. When analysing bird data in relation to habitat, the scarcest variables were either omitted or amalgamated with others. Many variables were initially recorded as % cover of each site, but the majority covered only a small area (<10%) and these were converted to presence/absence. The continuous variables deciduous tree cover, coniferous tree cover and deciduous bush cover were log(x+1) transformed to normalize the data distribution. Mown grass cover and proportion of adjacent garden were not transformed. Finally, there was a high degree of inter-correlation in the data. Variables that were very highly correlated were either combined when ecologically similar (e.g. rough grass and weeds/nettles) or not included (the least common in terms of occurrence over the sample of sites was always deleted in pair-wise habitat correlations). This left a total of 17 individual variables that were analysed in relation to the bird data. In addition, composite variables were derived. First ordination was carried out on all 38 variables listed in Table 1.1 (this was on the raw cover values rather than data converted to presence/absence). Second, Shannon diversity index (Magurran 1988) was determined per site for habitat cover variables and site boundary variables separately (Table 1.1).

Analysis for species richness, individual species density and individual species probability of occurrence followed similar modelling procedures. Species richness was determined for each site over the three visits separately for each season (summer and winter). Log-transformed species richness per site was analysed with respect to location and site type using normal regression where log(site area) was used as an offset. Model fit was described using the ratio of deviance divided by degrees of freedom, denoted as *D*. These normally distributed models gave a better fit to the data than Poisson models which typically were highly over-dispersed (D>10). The dispersion parameter was adjusted according to the scaled deviance and significance was assessed using F-tests which are more appropriate than the standard χ^2 for over-dispersed models. Differences between summer and winter were analysed using paired t-tests where a given site was surveyed in both seasons. Habitat was considered both at the site level and at the patch level. The former considered effects of habitat extent or presence on species richness at the whole site level where the analytical goal was to identify those features that determine the species richness per site, including both internal and external habitat features.

The patch level analysis considered differences between habitat patches where species richness was determined at the patch level using the reduced set of habitat variables (17 variables in total). The goal of this analysis was to identify those habitat types within public green spaces which had the greatest number of species. The area of each patch was used as an offset in the model, where species richness was analysed with respect to the class variable 'habitat' which had 17 possible levels (1 for each reduced habitat category in Table 1.1). In this case, the model had a repeated measures design to adjust for spatial auto-correlation of patches within the same site and significance was tested using χ^2.

Within-season changes in species richness and density were made by comparing the three visit periods per season as categorical variables. As multiple visits were made to each site, this also used a repeated measures modelling framework with site as the repeated subject factor.

A similar modelling approach was used for species density and species presence. The former used a normal error structure and density was log-transformed prior to analysis. The latter used a logistic regression approach where presence was expressed as the proportion of times a species was recorded out of the total number of visits in a given season/year. Area (log-transformed) was used as an offset in this model which necessitated the use of a complementary log-log link function. Individual species density and presence were analysed with the exception of gulls. It was suspected that some observers misidentified (or mis-named) Black-Headed Gull as Common Gull. Furthermore, several observers just entered 'Gull' or 'Gull sp'. Therefore, all gull species were combined in the analysis.

4. RESULTS

A total of 301 sites were surveyed, 219 from the main survey and 82 from the core survey. There were 219 sites surveyed in spring/summer 2002 and 210 sites surveyed in winter 2002/03. Respective numbers for 2003/04 were 208 and 198. 131 sites were surveyed in both years (with the exception of a single site, the two years were mutually exclusive in the core survey). There were several sites where some or all of the habitat data were missing so in many analyses, sample sizes are reduced.

4.1 Habitat

Most sites were classified as parks (n =142). Other sites types were: cemeteries (n = 57), public gardens (n = 29), recreation grounds (n = 11), open space (n = 12), woods (n = 22), playing fields (n = 9), squares (n = 7), commons (n = 4), meadows (n = 3), scrub (n = 2), allotment and marsh (n = 1 each). To a large extent these categories were observer-defined (on the habitat recording form, the choices were park, square, cemetery, botanic gardens and 'other (please specify)') and there will clearly be some overlap between how different observers classified their site. For the purposes of formal comparisons of site type, open space, commons, scrub, allotment, marsh and meadow were combined into a single 'miscellaneous' category.

There was little difference in the proportion of site types between location (Fig. 1.2). Both had parks as roughly 50% of their sample. There were a few differences (e.g. higher proportion of cemeteries in outer boroughs, higher proportion of public gardens in inner boroughs) but sample sizes were often small. The distribution of site area was skewed towards smaller sites. Subsequent analyses involving site area used log-transformed data. Site area differed significantly between different types (ANOVA $F_{7,271} = 7.29$, $P < 0.001$) where playing fields and woods had the largest mean area and squares and public gardens the lowest. Site area also varied according to location, with outer boroughs having a significantly larger area than inner boroughs ($F_{7,271} = 12.18$, $P < 0.001$). Furthermore, sites with the highest rank of road traffic level were significantly smaller ($F_{2,226} = 6.35$, $P < 0.01$; low traffic = 11.24 ± 14.46 ha (114), medium traffic = 10.67 ± 14.85 ha (100), high traffic = 6.35 ± 10.36 ha (55)).

Traffic level differed significantly between location, where inner boroughs had higher traffic levels than outer boroughs (Fig. 1.3). Similarly, human disturbance was higher in inner than outer boroughs (Fig. 1.3). Human disturbance and traffic levels were associated (Fig. 1.4). Nest boxes occurred in only 15% of the 297 sites where this information was recorded. There was no association with site area or with location. Of the 45 sites with boxes, 50% were in parks and a further 34% were in cemeteries.

Presence/absence of habitat types (using cover variables converted to binomials as in Table 1.1) was compared between site types. For categorical variables, a χ^2 test was used, but in order to avoid low expected values, three habitat types were not included in the analysis: squares, woods and playing fields. There were four variables where there was a significant difference in proportion of occurrence between site types (Fig. 1.5). Flower beds, hedgerows and sports surfaces were all by far the most common in parks. Statues/memorials were not surprisingly most common in cemeteries, but there was also a relatively high proportion in parks (Fig. 1.5). Presence of moles hills and ant hills was also recorded but these were rare occurring in 12 and 28 sites respectively (4.0% and 9.4% of the sample).

Continuous cover variables were compared between habitat using analysis of variance. Means are shown for the variables that were significant in Table 1.2. Coniferous tree cover differed significantly between site types ($F_{7,275} = 5.21$, $P < 0.001$) where cemeteries and playing fields had the highest cover and miscellaneous and woods had the lowest. Deciduous tree cover was highest in woods, but also in squares and public gardens and lowest in recreation grounds ($F_{7,275} = 2.26$, $P < 0.03$). Deciduous bush cover was highest in woods and lowest in recreation grounds and squares ($F_{7,275} = 2.22$, $P < 0.033$). Mown grass cover was highest in playing fields and recreation grounds and lowest in woods and miscellaneous site types ($F_{7,275} = 14.20$, $P < 0.001$).

Continuous cover variables were correlated with site area. There was a significant negative correlation between proportion of mown grass and site area ($r = -0.20$, df = 274, $P < 0.001$) and a positive correlation between proportion of coniferous tree cover and site area ($r = 0.18$, df = 274, $P < 0.003$).

The percentage cover of 38 habitat types was analysed using ordination for 216 sites surveyed over the two years (3 sites did not have any habitat data collected). DCA gave a length of gradient for axis 1 of 2.19 indicating that the distribution is linear and therefore that PCA would be a more appropriate ordination method (Jongman *et al*. 1995). Axis 1 and Axis 2 explained 15.0% and 10.2% of variation in the data respectively in a PCA. Bi-plots of axis scores are shown in Fig. 1.6. Interpretation was not straightforward. Axis 1 may have represented a gradient of sites with built structures (wall, adjacent wall and buildings all scores highly on this axis) to those without. The second axis was harder to interpret. This was repeated for inner and outer boroughs separately, but results were similar with a relatively low amount of variation explained on Axis 1 for both inner (16.1%) and outer (14.5%) boroughs and similar patterns being evident. Due to difficulties in interpretation, the derived axes were not considered in relation to the bird data.

Habitat diversity and boundary diversity were significantly correlated ($r = 0.23$, df = 275, $P < 0.001$). There was no relationship between site area and habitat diversity ($r = 0.06$, df = 275, NS) or boundary diversity ($r = 0.02$, df = 275, NS). There was no significant difference in habitat diversity between inner and outer boroughs ($T_{214} = 1.21$, NS). There was a significant difference in habitat diversity between site types ($F_{7,275} = 4.12$, $P < 0.004$) where playing fields surprisingly had the highest diversity and recreation grounds the lowest (Fig. 1.7). However, there were some small sample sizes, notably for playing fields and squares. When these site types were removed there were still significant differences. Very similar results were found with boundary diversity in terms of significance, although in this case the most diverse boundaries were on average in the miscellaneous site type and the lowest were in squares (Fig. 1.7).

4.2 Bird Data

A total of 90 species were recorded in the two years of the survey, 85 in the summer and 75 in the winter. A full list of species occurrence and mean density is given in Appendix II. The most commonly occurring species was Blackbird (present on almost all sites in the summer), with Woodpigeon, Blue Tit, Carrion Crow, Magpie and Robin all being present on over 90% of sites in at least one season. Feral Pigeon, Blackbird, Starling and Woodpigeon had the highest densities, the latter species showing a notably higher density in winter (and a similar though lower occurrence rate) indicating that there is some influx into green spaces in winter.

BAP species recorded included Song Thrush (present in 53-55% of sites depending on year/season), Linnet, Bullfinch, Skylark and Reed Bunting (all < 10%). Two declining urban species, Starling and House Sparrow, occurred in 83% and 50% of sites in the breeding season respectively and 75% and 36% of sites in the winter. Densities in both species were also lower in the winter. For House

Sparrow at least, this decrease in winter coincides with an increase in reporting rates in gardens in winter (Chapter 4; Cannon *et al*. in press, Chamberlain *et al*. in press) indicating an increase in exploitation of bird food at this time. There were also several unusual species recorded, including Black-necked Grebe, Cetti's Warbler, Grasshopper Warbler, Waxwing and Firecrest (all single records).

4.3 Species Richness

There was a large variation in species richness between sites. Stubbers Outdoor Pursuit Park, Havering had the highest summer richness with 45 species. In the winter, the greatest number of species was 40 at Harrow Lodge Park. At the other extreme there was one site, St Luke's Churchyard in Islington where only two species were recorded in both summer and winter.

Species richness was analysed with respect to year, season, location and site type using a Poisson regression modelling approach. Initially all terms and two-way interactions were entered into the model. Patterns were consistent across years and season, there being no significant interactions and also no significant difference in species richness between years ($F_{1,629} = 0.01$, NS) or between season ($F_{1,629} = 1.47$, NS; Table 1.3), where summer and winter species richness were virtually identical in the first year. There was a highly significant difference in species richness between location ($F_{7,629} = 15.67$, $P< 0.001$). Outer boroughs had more species than inner boroughs overall (Table 1.4), but sites tended to be larger in outer boroughs. There was also a significant difference between site types ($F_{5,629}= 9.52$, $P < 0.001$). Woods and parks had the highest richness and public gardens the lowest overall (Table 1.5).

There was a highly significant effect of site area on species richness, with more species detected in larger sites (Fig. 1.8). This pattern was consistent in each year/season. The effect this has on interpretation can be seen in Tables 1.4 and 1.5 where a simple measure of species richness per ha has been calculated as a comparison. When considered by unit area, species richness showed a more variable pattern with respect to location (Table 1.4) and was actually highest in the Inner SW location. Furthermore, inner boroughs held more species per hectare than outer boroughs. Similarly, the pattern with respect to site type was different when site area was taken into account. The greatest mean number of species per hectare was found in public gardens, squares and miscellaneous habitats (Table 1.5). Playing fields and cemeteries had the lowest number of species per hectare. There was no significant difference in species richness between different levels of traffic or human disturbance.

In subsequent analyses, species richness is corrected by site area. This was done in a two stage process. First, species richness per hectare was calculated. This itself may not be adequate to control for site area effects (D. Dawson pers. comm.) as the relationship between area and richness is non-linear (Fig. 1.8). Therefore, in addition, (log) area was used as an offset in subsequent models. These models used a normal error structure and the dependent variables were also log-transformed which improved the model fit (as measured by *D*).

Seasonal patterns in species richness were analysed using a repeated measures model (where there were three visits to each site per season). There were weakly significant differences in summer 2002 ($\chi^2_2 = 9.50$, $P < 0.025$) and stronger differences in summer 2003 ($\chi^2_2 = 19.93$, $P < 0.001$), where species richness declined over the season (Fig. 1.9). There were also significant differences in the winter 2003/04 ($\chi^2_2 = 12.00$, $P < 0.008$), where the late visit had significantly higher species richness than the earlier two (Fig. 1.9).

Seasonal patterns differed between inner and outer boroughs in summer 2003 ($\chi^2_2 = 6.43$, $P < 0.045$) where both locations showed a decline, but this was steeper in inner boroughs. There was no significant interaction between location and visit in the winter of either year. Furthermore, there were no significant interactions between site type and visit in any year or season.

Site level - Variables that were significantly associated with species richness adjusted for site area are shown in Table 1.6. Model fit was slightly over-dispersed (2.73-2.95) but was within acceptable limits. There were three variables that were significant in each of the four year/seasons. Richness was significantly lower in the presence of rough grass/nettles (RG), significantly higher when buildings were present (BLDG) and was significantly positively correlated with the cover of deciduous bushes. Other associations included negative correlations with coniferous tree cover and the proportion of adjacent gardens, lower species richness where mixed trees were present and where a railway was present within 100m of the site and positive correlations with mown grass cover.

Common habitats in parks include mown grass, deciduous trees and bushes (Table 1.2). The effects of bush presence were analysed by defining sites into groups: those with deciduous trees and mown grass and those with deciduous trees, mown grass and deciduous bushes (those without deciduous trees and mown grass were omitted). Species richness was compared between the two types using Poisson regression. Years differed significantly in species richness, but there were no seasonal differences or interactions so seasons were combined. There was a significantly higher species richness per hectare in sites with bushes in 2002/03 ($F_{2,157} = 9.94$, $P < 0.001$; with bushes = 4.37 ± 0.78 95%CL, n = 105; without bushes = 3.75 ± 0.86, n = 54). However, the significant result was reversed in 2003/04 ($F_{2,141} = 7.44$, $P < 0.001$; with bushes = 4.39 ± 0.81, n = 104; without bushes = 4.48 ± 1.38, n = 39). In both cases but especially 2003/04, differences, though significant, were small.

Patch level - Estimates of species richness per hectare of habitat, offset by site area, are shown in Fig. 1.10. The effect of habitat was highly significant in each year/season ($P < 0.0001$). Patterns were generally consistent across seasons and years with the greatest species richness found in vegetated habitats. Deciduous trees and deciduous bushes in particular had high species richness. Coniferous trees, evergreen bushes and evergreen trees also had relatively high richness. Of the non-vegetated habitat types, bare ground and walls had relatively high richness.

4.4 Species Density

Site level – In most cases, there were significant interactions with both year and season so models were run separately according to year and season. In many cases, the density in 2003/04 was higher than in 2002/03 although this was significant in only three species: Greenfinch (winter), Dunnock (summer) and Redwing (winter) (two-sample t-tests).

Within-season variation in density was considered by comparing visit periods. In summer the three visit periods were: mid-April to mid-May; mid-May to mid-June; and mid-June to mid-July. In total eight species showed a significant difference in density between these periods. Mean density per visit period for these species are shown in Fig. 1.11. A range of patterns was shown and there were relatively few species that showed the same significant difference in both summers. Magpie was the most obvious exception, with much higher densities in the first period in both years. It seems likely that conditions in individual years dictate within-season patterns (e.g. weather may cause earlier or later breeding which in turn could affect detectability). It should also be noted that the sample is only partially overlapping between years, so site differences may also influence year-to-year variation.

In winter the three visit periods were mid-November to mid-December, mid-December to mid-January and mid-January to mid-February. There were 22 species that showed significant differences between visit periods in at least one year. However, the patterns were even less consistent than in the summer. For example, only a single species, Blue Tit, showed very similar patterns in both winters (an increase through the winter). Furthermore, in many species, the significance was caused by a markedly high density in a single period and more or less equal density in the other two. These results may suggest that peaks are caused by influxes, possibly as a result of cold weather.

There was a significant interaction between visit and location in only three species, Blue Tit (summer 2002), Magpie (winter 2002/03) and Redwing (winter 2003/04) indicating that seasonal patterns do not differ between inner and outer boroughs for most species. Furthermore, the above three species showed only weakly significant differences (0.05 > P > 0.03 in each case). Fig. 1.12 shows means per visit for the three significant species. For Blue Tit and Redwing, increases over the winter were especially large in outer boroughs. Magpie declined through the winter and this was most marked in inner boroughs (Fig. 1.12).

There were three species that showed a significant interaction with site type when the scarcer sites (playing fields and squares where n < 10) were omitted: Blackbird (winter 2003/04), Carrion Crow (summer 2002) and Magpie (winter 2002/03). Blackbird showed the most marked decrease over the winter in public gardens, whereas there were increases in most other habitats (Fig. 1.13). The pattern in Carrion Crow was similar (but note that this was in summer rather than winter) with decreases through the summer in public gardens and miscellaneous site types and increases in parks and recreation grounds. Magpie showed peaks in density in winter 2002/03 in each site type except recreation grounds and woods where densities were stable through the winter (Fig. 1.13).

A summary of those habitat features that were significantly associated with bird density at the site level is shown in Table 1.7. As a rule of thumb, models are considered a good fit to the data if *D* is between 0.5 and 2.0. There were few species where the preferred range of *D* was met. For this reason, the more conservative F-tests are used to assess significance in Table 1.7, but highly under (D<0.20) or over-dispersed (D>10) models are not included. Where model fit was adequate, there were several significant associations (Table 1.7). One variable, the presence of buildings, stood out as having consistent effects across a range of species in all seasons and years. Density was significantly higher where buildings were present on a green space in 9 species in at least one year/season: Blue Tit, Blackbird, Feral Pigeon, Woodpigeon (these four species significant in each year/season), Wren, Magpie, Carrion Crow, Long-tailed Tit and Robin. Mown grass cover was also commonly correlated with species density, although the direction of the relationship varied from species to species. Density increased with mown grass cover in Feral Pigeon, Starling, Carrion Crow and House Sparrow and decreased in Blue Tit, Great Tit, Robin and Wren. Deciduous bush cover was positively correlated with density of Blackbird, House Sparrow, Starling, Magpie and Greenfinch. However, density was significantly lower where coniferous trees or bushes were present in Blue Tit, Blackbird, Feral Pigeon, Wren, Starling, Carrion Crow and Magpie. There were several other variables that had significant associations with bird density, but their effects were less consistent across species.

The presence of buildings seems unlikely to represent causal effects, especially as the species showing significant associations are not closely associated with buildings for nesting. It seems likely that buildings were present on sites that had some co-related factor present. A comparison of cover between sites with and without buildings found that mown grass, deciduous trees and deciduous bushes covered a significantly greater proportion of area on sites with buildings. These sites were also smaller than sites without buildings (see below). Sites with buildings may therefore represent sites with a range of beneficial habitats present.

The above models were repeated but site area was added. Although site area was taken into account in the analysis by using density as the dependent variable, it is possible that density itself could be affected by the area of a site due to edge effects (for example, Skylark density in farmland increases as field area increases- Chamberlain *et al*. 1999). There were a total of 13 species that showed a significant negative effect of site area (density decreased as area increased) in at least one year/season: Blue Tit, Blackbird, Chaffinch, Carrion Crow, Feral Pigeon, Great Tit, Greenfinch, House Sparrow, Long-tailed Tit, Magpie, Starling, Woodpigeon and Wren. To some extent this could have been caused by an inflation of density estimates in very small sites (<1 ha). When these sites were removed, results were similar although fewer species showed a significant association (Blackbird, Carrion Crow, Feral Pigeon, House Sparrow, Starling and Woodpigeon).

Species density was compared between sites with and without nest boxes present for selected box-nesting species in the summer: Great Tit, Blue Tit, Coal Tit, Marsh/Willow Tit, Robin, House Sparrow, Nuthatch and Treecreeper. There were significant differences in a number of species (Table 1.8). However, densities were significantly higher where boxes were absent in most cases. There were relatively few sites with nest boxes present (15%) and most of these were in cemeteries or parks so these differences may have been caused by co-related factors.

Species density was considered in relation to habitat diversity (expressed using the Shannon index). Few species were significantly correlated with habitat diversity (Table 1.9) and for those that were the relationship was non-linear in form where density peaked at intermediate levels of habitat diversity. This was the case for Blackbird, Woodpigeon and Blue Tit. An example of correlation between density and diversity is shown in Fig. 1.14. Such a non-linear relationship suggests that sites with many habitat types or evenly distributed habitat types are not necessarily the best ones. This may also suggest that a small number of key individual habitats are important rather than habitat diversity *per se*. These issues are returned to in the discussion.

Analyses of individual habitat variables indicated that mown grass and deciduous bushes often were closely correlated with individual species abundance (Table 1.7). Rather than consider individual (and often correlated) variables, an analysis of three major habitat types, deciduous bushes, deciduous trees and mown grass, was undertaken in order to assess the effects of the presence of deciduous bushes. The density of species was compared between those sites with deciduous trees and mown grass and those with deciduous trees, mown grass and deciduous bushes. There were several species that showed significant differences between these two categories (Table 1.10). There were five species where the density was significantly higher where bushes were present in each season/year: Blue Tit, Blackbird, Starling, Feral Pigeon and Woodpigeon. Other species showing this result (but not in each year/season) were House Sparrow, Redwing and Greenfinch. Long-tailed Tit showed a significantly higher density in sites without bushes in winter 2003/04. Carrion Crow apparently showed seasonal interactions, as density was significantly higher on sites with bushes in one winter but higher on sites without bushes in each summer.

Patch level – In common with the site level analyses, many species showed significant interactions with year and season so models were run separately by year and season.

Estimates of (log) density of birds per habitat in 2002/03 determined at the patch level are shown in Fig. 1.15 for species that occurred in at least 10% of sites, where differences between habitats were significant and where model fit was not too over dispersed (D>10) or underdispersed (D<0.20). Note that these estimates are adjusted for autocorrelation using General Estimating Equations in a repeated measures design. Generally, densities were higher in winter than in summer probably because local numbers are augmented by young birds from the previous breeding season, short-distance migrants moving in from rural areas and for some species (thrushes, gulls) long distance migrants from other countries. There was significantly higher density (paired t-test) in summer for Starling and Wren and also for Blackcap and Whitethroat (where virtually all records came from the summer). In winter, densities were significantly higher for Moorhen, Woodpigeon, Pied Wagtail, Blue Tit, Great Tit and Chaffinch. Gulls and Redwing were mostly confined to the winter and were not considered in summer. As numbers were often low and relatively constant in the summer, models for many species had low dispersion indicating that assumptions about the data distribution were not met. In cases where dispersion was very low, binomial models may be more appropriate (see below).

For several species, the highest densities were found in deciduous trees in 2002/03 (Fig. 1.15). There were clear peaks in density (particularly in winter) for the following species: Woodpigeon, Greenfinch, Jay, Blue Tit, Long-tailed Tit, Great Tit and Song Thrush. For many of these species deciduous bushes and mixed bushes and trees also held high densities. In contrast there were relatively few species showing a clearly higher density in coniferous trees, the exception being Goldcrest. Several species had their highest recorded density in deciduous bushes: Wren, Dunnock, Robin, Mistle Thrush, Blackcap, House Sparrow and Chaffinch. However, other types of bush were not heavily used by any species. Of the other habitats, mown grass also held high densities, particularly in the winter for gulls, Feral Pigeon, Blackbird, Starling and Carrion Crow (Fig. 1.15). In 2003/04 results were very similar, with deciduous trees and bushes typically having the highest densities in the majority of species, although there were fewer species where density differed significantly between habitats (Fig. 1.16).

For selected species where model fits were highly under or over-dispersed, count data were reduced to presence/absence per visit. The ratio of the number of times a species was present on a site to the number of times a site was visited in a season was analysed with binomial logistic regression. In common with the above approach, this used area as an offset and adopted a repeated measures model framework. The species analysed were Green and Greater Spotted Woodpecker, Pied Wagtail, Fieldfare, Chiffchaff, Whitethroat, Coal Tit, Treecreeper, Nuthatch, Goldfinch and Bullfinch. Estimated probability of occurrence per hectare of species showing a significant effect of habitat are shown in Figs. 1.17 and 1.18.

Woodpeckers and Nuthatch showed the highest probability of occurrence, as expected, in deciduous trees, although Green Woodpecker also had relatively high occurrence in mown grass and rough grass, reflecting its preference for foraging on the ground. Chiffchaff and Goldfinch also occurred most commonly in deciduous trees. Deciduous bushes had relatively high probability of occurrence in Whitethroat, Chiffchaff and Goldfinch. Coal Tit was most likely to occur in coniferous trees. Pied Wagtail was more likely to occur on open areas, especially hard surfaces (pavement and sports surface). It should be noted that occurrence rates were very low and error bars often large in many cases. Although all models from which estimates were derived for Figs. 1.17 and 1.18 were within the acceptable range of deviance, they were still mostly underdispersed (D<0.50).

4.5 Squirrels

The distribution of squirrel counts was highly skewed and no transformation was identified that normalized the data, hence non-parametric statistics were used. Mean count per site was determined over all visits within each season and year. Squirrel count was significantly positively related to site area in each year and season so further analyses used squirrel density. There was no significant difference in median squirrel density between inner and outer boroughs in any year or season (Kruskal-Wallis tests). Furthermore, there was no significant difference in squirrel density between site types.

Squirrel density was analysed in relation to the four continuous habitat variables used in the bird density analysis (see Table 1.1) using Spearman rank correlation. There was no significant correlation with deciduous tree cover. There was a significant negative correlation with mown grass cover in winter 2002/03 only ($r_s = -0.17$, $P < 0.021$, n= 177). Both coniferous tree cover and the proportion of adjacent garden habitat were significantly ($P < 0.01$) positively correlated with squirrel density in each year and season. There was no significant correlation with deciduous tree cover.

For the binomial habitat variables, squirrel density was compared between sites where a given habitat was either present or absent using Wilcoxon tests. There were eight habitats where squirrel density differed significantly according to whether that habitat was present or absent at a site (Fig. 1.19). Squirrels had significantly higher density in sites where hedges, mixed trees, walls and rough grass were present than where they were absent in most years/seasons. Sites with sports surfaces and flower beds had significantly higher densities of squirrels in winter 2003/04 only. Squirrel density was significantly higher in sites where there were no buildings or paved areas.

5. DISCUSSION

A wide range of species was detected. In total, there were 90 species recorded. In the more complete atlas (Hewlett 2002), 141 species were recorded. Given that the green spaces survey didn't count wildfowl apart from swans, Canada Goose and Mandarin and that gulls were grouped together, a more comparable number from Hewlett (2002) would be 122 species (taking away 16 wildfowl species recorded and substituting 4 gull species for 1 generic group). The results of this survey therefore suggest that approximately 74% of the total London avifauna can be found in public green spaces.

Most species were more abundant in the winter. This may be due to greater detectability in the winter, a greater number of birds due to juveniles in the population or due to seasonal influxes from the surrounding countryside and, possibly for certain species, overseas. Species richness decreased throughout the summer. This may have been due to a lower detectability as breeding activity (especially singing) subsides. There was a slight increase in species richness through the winter which may represent increasing influxes of species into urban green spaces as resources are depleted in semi-natural habitats, as observed for certain granivorous species on farmland (Gillings & Beaven 2003). However, there were no consistent within-season patterns across individual species. Rather, most species did not show significant differences between visit periods and in those that did, patterns were species specific. There are strong seasonal patterns in the use of gardens in winter (Cannon *et al*. in press, Chamberlain *et al*. in press, Chapter 4), but similar patterns of increase were only evident in Blue Tit and Redwing (Fig. 1.12). Within season use of green spaces is therefore difficult to predict and may depend on short-term weather conditions.

Potential bias – There were several potential biases in the survey. First, site selection was not random. This may have biased against certain site types and may have led volunteers to select sites that were of known ornithological interest. A particular concern was that smaller sites and/or inner city sites would not be well covered. However, because of *a priori* concerns over site selection, a particular effort was made to encourage volunteers to cover small sites in addition to larger, potentially more bird-rich sites. Furthermore, when a request was made for a site that was already covered, an effort was made to allocate less favoured sites to the volunteer. Therefore, it is likely that biases towards larger sites of greater ornithological interest were minimised.

There was a geographical bias in the distribution of sites in Greater London. In order to ensure that enough sites in inner boroughs were covered, many core sites were allocated here. However, Fig. 1.1 shows that poor coverage was actually a problem in the outer boroughs: Havering, Bromley, Barnet, Harrow, Croydon, Richmond and Redbridge were outer boroughs that had only 1 site each covered whereas only 1 inner borough, Wandsworth, had a single site covered. It is unclear whether this could have led to any bias in the analysis of habitat associations, but it is certainly a bias in describing a complete London avifauna as large areas were not surveyed adequately.

Habitat associations – The area of a site was a major determinant of individual species abundance and species richness. This is not surprising – even a random settlement of individuals or species would result in a larger number in larger areas. Furthermore, larger sites would have been subject to longer survey times, so increasing the likelihood of detecting more species. Interpreting the value of a site in terms of species richness is not straightforward and different results can be obtained depending on whether area effects are taken into account. For example, sites in outer boroughs have a higher species richness than inner boroughs. However, species per hectare is actually significantly higher in sites in inner boroughs because these are significantly smaller. This is an admittedly very simple correction for habitat area, but it nonetheless demonstrates that it cannot be assumed that inner city green spaces are of less value than those in outer London. The reason they have more species per unit area may be due to habitat. Larger parks in outer boroughs may be more dominated by large patches of single habitat types (e.g. mown grass) that do not necessarily hold the highest species richness. Alternatively, there may be an effect where green spaces are relatively more attractive to birds when the immediate surroundings are less suitable.

At the site level there were some general patterns that emerged. Species richness and the density of several individual species were positively associated with buildings and deciduous bushes and negatively associated with rough grass/nettles, coniferous trees and bushes. For the latter category, many conifers will be non-native (e.g. *leylandii*) which may confer lower ecological value to birds. For example, native trees are known to have a higher richness of invertebrates than non-native trees (Southwood 1961) which could affect use by birds. Conversely, deciduous bushes may be more likely to be native species. Similar associations were seen at the patch level in that many species were found at their highest density in deciduous trees and deciduous bushes. It is likely that both coniferous and deciduous bushes and trees have the potential to provide cover in addition to food resources. Varying degrees of cover could also influence detectability of birds using these habitats. However, there was no information on tree/bush species composition or structure so these ideas cannot be examined further. Additional research is required into these questions.

In farmland, taller swards and grass managed at a low intensity tends to support more surface/foliage invertebrates than short, highly managed swards (Atkinson *et al*. in prep.). This may also be the case in urban green spaces. However, in pasture, accessibility appears to be a key determinant of habitat use and short swards are actually preferred by many species (Atkinson *et al*. in prep.), which may explain some of the negative associations with rough grass/nettles. This does not however mean that they are poor habitats. Rather, they may act as reservoirs of invertebrate prey which can only be accessed in more open areas. Several species showed positive associations with the area of mown grass. This may indicate the importance of accessibility. Earthworms in particular are likely to be important for several species such as gulls, Blackbird, Starling and Carrion Crow.

The positive association with buildings, which was one of the most consistent habitat associations, is hard to explain in ecological terms. It is more likely that sites where buildings were present also had some other attributes that were attractive to a range of species. For example, deciduous bushes, deciduous trees and mown grass covered a greater proportion of site area on sites with buildings. These habitats are positively associated with several species and deciduous bushes with species richness. This example illustrates the general problem of interpreting correlations where there is colinearity in the data. It is very difficult to tease apart effects of individual habitat factors when they are closely correlated with one another. This also extends to analyses of site type and location. Sites in inner boroughs were more likely to be small, to be squares or public gardens, to have higher traffic levels and higher human disturbance. Any one of these factors could influence bird communities. Interpretation should therefore bear this in mind. A consideration of general patterns rather than specific associations is probably more sensible in this case.

There were three habitat types that consistently arose as significant in models of density and richness: deciduous trees, deciduous bushes and mown grass. Deciduous bushes in particular seemed important for a number of species. When site-level comparisons were made between sites with trees, mown grass and bushes and sites with trees and grass but without bushes, there were significant differences in seven species, including the declining House Sparrow and Starling (Table 1.10). Clearly for some species, bushes represent an additional resource and their presence can increase densities quite considerably. However, the effects on species richness were not consistent and there were two species, Long-tailed Tit and Carrion Crow, where density was higher (at least some times) in the absence of bushes, so deciduous bushes are not necessarily universally of benefit.

Jokkimäki (1999) found that the number of nest boxes in a site was positively correlated with Pied Flycatcher abundance and species richness in Finnish parks. However, there was no evidence that nest boxes benefited hole-nesting species in London’s green spaces. Indeed, for several species, sites where nest boxes were present had significantly lower densities. It is possible that nest boxes are put up in sites that are poorer than average for birds (indeed a lack of birds may be a reason for their provision). Furthermore, the placement of a nest box (e.g. height, aspect, cover) is likely to be crucial to its likely occupation (Luniak 1992).

There were few significant correlations with habitat diversity and where this was significant, the relationship was non-linear (Fig. 1.14). There are several ways to express habitat diversity (Magurran 1988) and all diversity indices have particular biases. Use of only a single index in the analysis is therefore a valid criticism. However, the analysis illustrates that promoting habitat diversity *per se* may not be a particularly useful management strategy. Clearly, evidence from associations with individual habitats shows that some habitats are avoided by most species. Rather than maximise habitats, it is probably better to concentrate on a smaller number of the better habitats to increase bird abundance/diversity. However, it should be noted that this may not be the case if it is sought to increase diversity of other taxa. The issue of habitat diversity should therefore be given serious consideration in any management plan.

Species of conservation concern – The results of this study may be used to further our knowledge of species of conservation concern. Of the declining species in Greater London, the House Sparrow is probably of the greatest concern and its decline has been severe, even over relatively short periods (Raven *et al*. 2003). There was one factor that was consistently related to House Sparrow abundance in different analyses: deciduous bushes. These may be important due to cover as shelter from the elements and predators. However, unlike trees, smaller bushes offer proximity of dense cover with a view of potential predators (e.g. Sparrowhawk). Unless deciduous and coniferous bushes differ greatly in the cover provided, it seems unlikely that cover alone would be the reason for the association as House Sparrows were rarely recorded in coniferous bushes or indeed coniferous trees (Figs. 1.15 & 1.16). It may be that food abundance is also higher in deciduous bushes. Mown grass was also positively associated with House Sparrow density at the site level, reflecting this species' preference for ground foraging where seeds of grasses and dicots are taken.

Starling showed very similar habitat preferences to House Sparrow in that higher densities were associated with deciduous bushes and mown grass. However, food resources exploited in grass are likely to differ to those exploited by House Sparrow, Starlings commonly feeding on soil invertebrates such as leather jackets and earthworms. Starlings (along with many species) also had high densities in deciduous trees.

The Spotted Flycatcher had a widespread and relatively stable distribution in London, but there is evidence from several studies that numbers have been in decline for some time (Hewlett 2002) which reflects the national trend (Freeman & Crick 2003). Spotted Flycatchers were recorded on only three occasions in this survey and there was no evidence of breeding from the core survey (Chapter 2). This is a species that is clearly in a perilous state in Greater London. The causes of decline are unclear but appear to be acting on survival of juveniles (Freeman & Crick 2003). Whether this is due to factors on the breeding grounds or in wintering areas (or both) is a matter for urgent research.

Not all species of conservation interest are declining. The Ring-necked Parakeet is an introduced species which has caused some concern as a potential pest as it continues to increase (Pithon & Dytham 1999, Hewlett 2002). The species occurred on between 16-19% of sites. Habitat associations were difficult to analyse due to the data distribution as the species was fairly scarce but could be highly clumped with large roosts. Patch-level estimates suggested that deciduous trees and mown grass were the only habitats significantly associated with numbers of this species, but error bars were high due to the data distribution. However, these data may prove important as a baseline for future surveys into the spread of this species.

In addition to bird counts, one species of mammal, the Grey Squirrel, was surveyed. Habitat associations were not easy to analyse or interpret. Two results were of particular note. First, squirrel numbers were positively correlated with the proportion of the site boundary comprising adjacent gardens, possibly indicating an association with food (especially supplementary bird food) available in gardens. Second, squirrels were significantly correlated with the cover of coniferous trees, but not deciduous trees. It would be interesting to know if the coniferous trees in these cases are largely *leylandii* and other exotic species which may provide particularly good nesting habitat. Grey Squirrels are not native species. Whether their spread has been responsible for the demise of the native Red Squirrel is a matter for conjecture (Tompkins *et al*. 2002). Of more relevance to London is whether they actually do any harm to native species. There is evidence that Grey Squirrels could be major nest predators and possibly could have contributed to population declines in some bird species (Hewson *et al*. 2003). Whether it is desirable to take management steps to reduce squirrel numbers is open to question. Certainly squirrels are popular amongst many members of the public and the exceptionally high numbers in certain sites in this study were clearly due to food being provided specifically for squirrels.

Management recommendations – It cannot, of course, be assumed that any of the habitat associations presented represent causal effects. However, there were a number of statistical associations that were consistent across years/seasons and in different analyses, but more importantly some of these consistent effects were in accord with what is known about particular species' ecological requirements.

Deciduous trees were clearly very important habitats for a range of species and patches of deciduous trees held the highest species richness in 3 out of 4 years/seasons. This is perhaps not surprising as many species that are common in urban green spaces are birds whose ancestral habitat is woodland or woodland edge (Hewlett 2002). Planting of deciduous trees will certainly increase bird abundance and species richness in urban green spaces (but not to the extent of reducing all open space – see below). There is no evidence from this study that native deciduous trees are better than non-native deciduous trees, but it seems likely that native species will accrue wider biodiversity benefits and they should be preferred where possible.

Deciduous bushes were particularly important habitats for certain smaller passerines, and patches of deciduous bushes held the first or second highest number of species (depending on year/season). Most notable amongst the individual species was House Sparrow. Planting deciduous bushes is likely to increase the abundance of several species. For many species, both deciduous bushes and trees were important habitats so planting bushes as understorey may be of benefit, especially for Wren (breeding season), Robin, Blackbird, tits, Starling, Chaffinch and Greenfinch. Other species showed clearly the highest densities in deciduous bushes and were not apparently closely associated with deciduous trees. For Dunnock, Blackcap, Whitethroat and House Sparrow bushes alone are likely to be beneficial. In common with deciduous trees, native species are probably desirable for wider biodiversity considerations.

Provision of deciduous bushes in parks may benefit birds and regarding areas of dense cover in parks biodiversity in general. However, there is a health and safety issue in that they are perceived as potentially threatening to some members of the general public (due to provision of cover for potential attackers). Whether this perception has any basis in fact (i.e. are muggings more likely in greens spaces with bushes?) is a moot point. Nevertheless, bush clearance or height reduction is often carried out, particularly adjacent to paths (D. Dawson, J. Hewlett, J. Archer, pers. comm.). Clearly this practice could have major repercussions for London's birds, not least the House Sparrow. It is suggested that effects of bush removal could be mitigated through better park design, where bushes are planted and encouraged away from paths and perhaps at site edges or in areas that are less well used by the general public.

In comparison with deciduous plants, coniferous trees and bushes had little value for birds. Only the Goldcrest and Coal Tit had particular associations with coniferous trees. Planting conifers therefore appears to confer little benefit to many species and deciduous trees and bushes should usually be preferred. There is one potential use of conifers that would not have been detected in this study in that they may provide good roost sites for species such as thrushes, Pied Wagtails and corvids in the winter. Further research is required to assess the potential value of conifers, especially denser species such as *Leylandii*, as roost sites.

Mown grass was clearly a well-used foraging habitat for a number of species. Maintenance of mown areas should be part of any management plan for birds. There was little evidence that rough grass/nettles were used by any species to any great extent. However, it would be unwise to exclude these habitats from urban green space as there is evidence from other studies that they may act as a reservoir for foliage invertebrates (unpublished BTO data). Patches of rough grass/nettles adjacent to mown areas may provide a good combination of prey abundance and accessibility.

Code	Habitat	Proportion present	Level	Units	Modelled
BE	Bare earth and bare paths	0.48	P	%cover	
CB	Coniferous bushes	0.20			y
CT	Coniferous trees	0.52			y
DB	Broadleaved bushes	0.62			y
DT	Broadleaved trees	0.92			y
FF	Flowerbed (clean)	0.41			y
FO	Fountain	0.08			with ST
GV	Gravestones	0.22			with ST
MB	Mixed bushes	0.42			
MG	Mown/short turf	0.92			y
MT	Mixed trees	0.28			y
OX	Other misc. habitats	0.06			
PV	Paved area	0.74			
PY	Playground/play area	0.39			
RF	Flowerbed (weedy)	0.12			with FF
RG	Rough grass	0.45			y
SP	Sports surface	0.24			y
ST	Statues	0.14			y
VB	Evergreen bushes	0.46			
VT	Evergreen trees	0.45			
WD	Weeds/nettles	0.48			with RG
WN	Wall/building (no vegetation)	0.59			y
WT	Water body (pond, stream)	0.31			
WV	Wall/building (vegetation)	0.26			with WN
BLDG	Building	0.22	B	% perimeter	y
FENC	Fence	0.86			
HDGE	Hedge	0.49			y
NONE	No boundary	0.10			y
OTHR	Misc. boundary feature	0.15			
PAVE	Pavement/road	0.23			y
WALL	Wall	0.34			
WATR	Water body	0.12			
AGARD	Private gardens	0.69	A	% perimeter	y
AGREN	Public green space	0.31			
AOTHR	Misc. habitat	0.23			
ARAIL	Railway	0.13			y
AROAD	Road	0.89			
AWALL	Buildings	0.32			
AWATR	Water body	0.11			

Table 1.1 Habitat types identified in the survey and the proportion of sites where they were present. Level P = patch, B = boundary, A = adjacent habitat. Modelled indicates whether a variable was used to analyse bird distributions (y) or not (blank) or whether variables were combined.

Habitat	Site type	Mean	LCL	UCL	n
Coniferous trees	Cemetery	7.19	4.87	9.50	54
	Gardens	2.86	1.21	4.50	28
	Misc	1.65	0.93	2.37	20
	Park	3.18	2.50	3.85	130
	Playing	3.83	-0.33	8.00	6
	Rec	2.27	0.30	4.24	11
	Square	2.57	0.38	4.76	7
	Wood	2.15	1.04	3.26	20
Deciduous trees	Cemetery	17.72	13.04	24.07	54
	Gardens	22.89	14.76	35.50	28
	Misc	15.36	9.01	26.18	20
	Park	14.97	12.47	17.99	130
	Playing	14.45	5.42	38.56	6
	Rec	10.71	5.29	21.69	11
	Square	21.22	15.01	30.00	7
	Wood	36.97	17.55	77.88	20
Deciduous bushes	Cemetery	5.94	2.71	9.17	54
	Gardens	7.36	3.08	11.64	28
	Misc	12.85	7.36	18.34	20
	Park	7.35	5.61	9.09	130
	Playing	4.67	0.65	8.69	6
	Rec	4.00	0.00	8.00	11
	Square	3.43	1.17	5.69	7
	Wood	16.20	5.10	27.30	20
Mown grass	Cemetery	40.83	34.04	47.63	54
	Gardens	53.64	43.76	63.52	28
	Misc	35.60	19.21	51.99	20
	Park	63.04	58.53	67.55	130
	Playing	75.00	63.99	86.01	6
	Rec	70.73	49.72	91.73	11
	Square	52.29	25.35	79.22	7
	Wood	10.65	-0.35	21.65	20

Table 1.2 Mean (±95% CL) percentage habitat cover in different site types. Only variables showing significant differences between site types are given. Note that coniferous tree cover and deciduous bush cover were log-transformed prior to analysis but are presented as raw percentages here.

	2002/03		2003/04	
Summer	16.45 ± 0.56	(219)	17.22 ± 0.57	(208)
Winter	16.45 ± 0.56	(210)	18.75 ± 0.55	(192)

Table 1.3 Mean species richness ± SE (n) in different years and season.

Location	Species per site	n	Species per ha	n
Inner NE	13.35 ± 1.04	52	1.91 ± 0.14	48
Inner NW	12.74 ± 1.27	31	1.78 ± 0.18	30
Inner SE	15.55 ± 1.44	25	1.62 ± 0.15	22
Inner SW	13.40 ± 1.30	21	1.99 ± 0.24	19
Outer NE	20.02 ± 1.23	47	1.51 ± 0.13	42
Outer NW	17.13 ± 0.85	59	1.49 ± 0.10	56
Outer SE	19.39 ± 1.57	19	1.56 ± 0.25	18
Outer SW	20.34 ± 1.15	47	1.67 ± 0.12	46

Table 1.4 Mean species richness ± SE (n) in relation to location in Greater London. Means were calculated from site means over all years and seasons. Note that n differs between the two measures due to missing area data for some sites.

Site type	Species per site	n	Species per ha	n
Cemetery	16.70 ± 0.78	57	1.57 ± 0.10	54
Public gardens	11.60 ± 1.20	29	2.35 ± 0.16	28
Misc.	18.77 ± 1.86	23	1.93 ± 0.25	21
Park	17.12 ± 0.70	142	1.59 ± 0.07	132
Playing field	18.19 ± 2.39	9	1.31 ± 0.23	8
Recreation ground	17.95 ± 1.55	11	1.55 ± 0.27	11
Square	7.07 ± 0.72	7	1.68 ± 0.18	7
Wood	21.15 ± 1.31	22	1.18 ± 0.20	20

Table 1.5 Mean species richness ± SE (n) in relation to site type.

Year	Season	Significant variables	*D*
2002	Summer	AGARD-- RG- MT- DB++ BLDG++	2.73
2002/03	Winter	CT- AGARD-- RG- MT- DB+ BLDG++	2.76
2003	Summer	MG+ CT- RG--- DB+ BLDG+++ ARAIL-	2.80
2003/04	Winter	MG++ CT- RG--- DB+ BLDG+++ ARAIL-	2.92

Table 1.6 Habitat features significantly associated with species richness at the site level. Habitat codes are given in Table 1.1. P values derived from F-tests. Sign indicates direction of effect where a negative sign indicates absence of a given habitat has a greater density than presence of that habitat. +/- $P < 0.05$, ++/-- $P < 0.01$, +++/--- $P < 0.001$.

(a) Summer

Year	Species	Habitat	*D*
2002/03	Blue Tit	BLDG+ CT- MG--	0.45
	Blackbird	AGARD- BLDG+++ CT—DB++ HDGE-	0.66
	Feral Pigeon	AGARD- BLDG+++ CT- FF++ HDGE- MG+	1.55
	House Sparrow	DB+	0.38
	Woodpigeon	BLDG+++	0.48
	Starling	CB- DB++ MG+++ SP-	0.83
	Robin	BLDG+ MG—PAVE-	0.25
	Wren	BLDG+ CT- MG—RG++	0.22
2003/04	Blue Tit	ARAIL- BLDG++	0.42
	Blackbird	BLDG+++ CT- DB++ RG--	0.57
	Carrion Crow	CB- MG+	0.36
	Feral Pigeon	AGARD--- BLDG+++ CT- FF++ HDGE—MG++ RG---	1.21
	House Sparrow	MG+	0.34
	Magpie	DB+	0.27
	Starling	MG+++	0.76
	Woodpigeon	BLDG++	0.53
	Wren	ST-	0.28

(b) Winter

Year	Species	Habitat	*D*
2002/03	Blue Tit	BLDG++ CT- HDGE- MG-	0.50
	Blackbird	BLDG+++ CT—DB++ HDGE—NONE-	0.65
	Chaffinch	AGARD++	0.21
	Feral Pigeon	AGARD--- BLDG+++ DT+ FF- HDGE—MG++	1.45
	Great Tit	BLDG+ DT++ MG-	0.35
	Greenfinch	ARAIL- DB+	0.22
	House Sparrow	DB+ MG++ ST-	0.25
	Long-tailed Tit	AGARD+ RG+	0.31
	Magpie	BLDG+ CB- DB++	0.32
	Robin	BLDG+++ HDGE- MG- PAVE-	0.25
	Redwing	AGARD+ ARAIL- RG+	0.35
	Starling	DB+ MG++	0.77
	Woodpigeon	BLDG+++ DB++ HDGE--	0.68
2003/04	Blue Tit	BLDG++ DT+	0.49
	Blackbird	BLDG++ CT- DB+ RG-	0.59
	Chaffinch	AGARD+ HDGE-	0.22
	Feral Pigeon	AGARD—BLDG+++ FF++ HDGE- MG+ RG---	1.27
	Carrion Crow	BLDG+ CB—MG++ PAVE-	0.40
	Greenfinch	MG+	0.33
	Great Tit	BLDG+	0.33
	House Sparrow	MG++ SP-	0.33
	Magpie	BLDG+	0.28
	Long-tailed Tit	BLDG+	0.39
	Woodpigeon	BLDG++	0.77
	Redwing	AGARD+	0.70
	Robin	BLDG++	0.28
	Starling	CB—MG+	0.82

Table 1.7 Habitat features significantly associated with bird density at the site level. Habitat codes are given in Table 1.1. P values derived from F-tests. Sign indicates direction of effect where a negative sign indicates absence of a given habitat has a greater density than presence of that habitat. +/- $P < 0.05$, ++/-- $P < 0.01$, +++/--- $P < 0.001$.

Species	**Year**	**Box present**			**Box absent**			***D***
		Mean	**LCL**	**UCL**	**Mean**	**LCL**	**UCL**	
Blue Tit	1	0.71	0.49	0.94	0.78	0.67	0.90	0.51
Great Tit	1	0.32	0.17	0.47	0.40	0.32	0.49	0.28
House Sparrow	1	0.29	0.04	0.54	0.34	0.24	0.44	0.38
Robin	1	0.57	0.36	0.79	0.55	0.47	0.64	0.28
Starling	1	0.92	0.61	1.22	1.20	1.04	1.36	0.96
Blue Tit	2	0.71	0.51	0.91	0.83	0.72	0.93	0.45
Great Tit	2	0.46	0.29	0.62	0.42	0.32	0.51	0.32
House Sparrow	2	0.19	0.03	0.35	0.38	0.29	0.48	0.32
Robin	2	0.45	0.32	0.58	0.56	0.47	0.64	0.27
Starling	2	0.87	0.61	1.13	1.12	0.97	1.26	0.80

Table 1.8 Density of selected species in sites with and without nest boxes in the breeding season. Only significant differences ($P < 0.05$ from F-tests) and models where $0.2 > D > 10.0$ are shown. Sample sizes 2002: sites with nest boxes = 26, sites without = 154; Sample sizes 2003: sites with nest boxes = 31, sites without = 159.

Species	**Year**	**Season**	**Model**	***D***
Blackbird	2	s	H+++ H^2--	0.68
Woodpigeon	2	s	H++ H^2--	0.55
Blue Tit	2	w	H^2-	0.53
Blackbird	2	w	H^2-	0.66

Table 1.9 Associations between site habitat diversity (H) with species density. + indicates positive association, - indicates negative association, superscript 2 = quadratic term. +/- $P < 0.05$, ++/-- $P < 0.01$, +++/--- $P < 0.001$.

(a) Summer 2002

Spp	Bush	LCL	UCL	No Bush	LCL	UCL	*D*
	105			**54**			
Blackbird	2.54	1.91	3.16	1.75	1.14	2.35	0.41
Blue Tit	1.30	0.95	1.64	1.06	0.77	1.35	0.26
Carrion Crow	0.87	0.66	1.07	1.21	0.61	1.80	0.21
Feral Pigeon	5.50	2.88	8.12	3.45	1.55	5.36	1.27
House Sparrow	0.65	0.37	0.93	0.30	0.10	0.50	0.20
Starling	4.10	2.82	5.39	2.64	1.65	3.64	0.76
Woodpigeon	1.86	1.38	2.35	1.33	0.74	1.93	0.32

(b) Winter 2002/03

Spp	Bush	LCL	UCL	No Bush	LCL	UCL	*D*
	102			**52**			
Blackbird	2.26	1.71	2.82	1.30	0.83	1.76	0.36
Blue Tit	1.85	1.38	2.32	1.03	0.70	1.36	0.31
Feral Pigeon	6.55	3.20	9.91	4.32	1.53	7.11	1.40
Redwing	1.00	0.37	1.63	0.61	-0.18	1.39	0.35
Starling	2.84	1.78	3.91	0.89	0.53	1.25	0.60
Woodpigeon	3.21	1.85	4.57	1.61	1.07	2.14	0.48

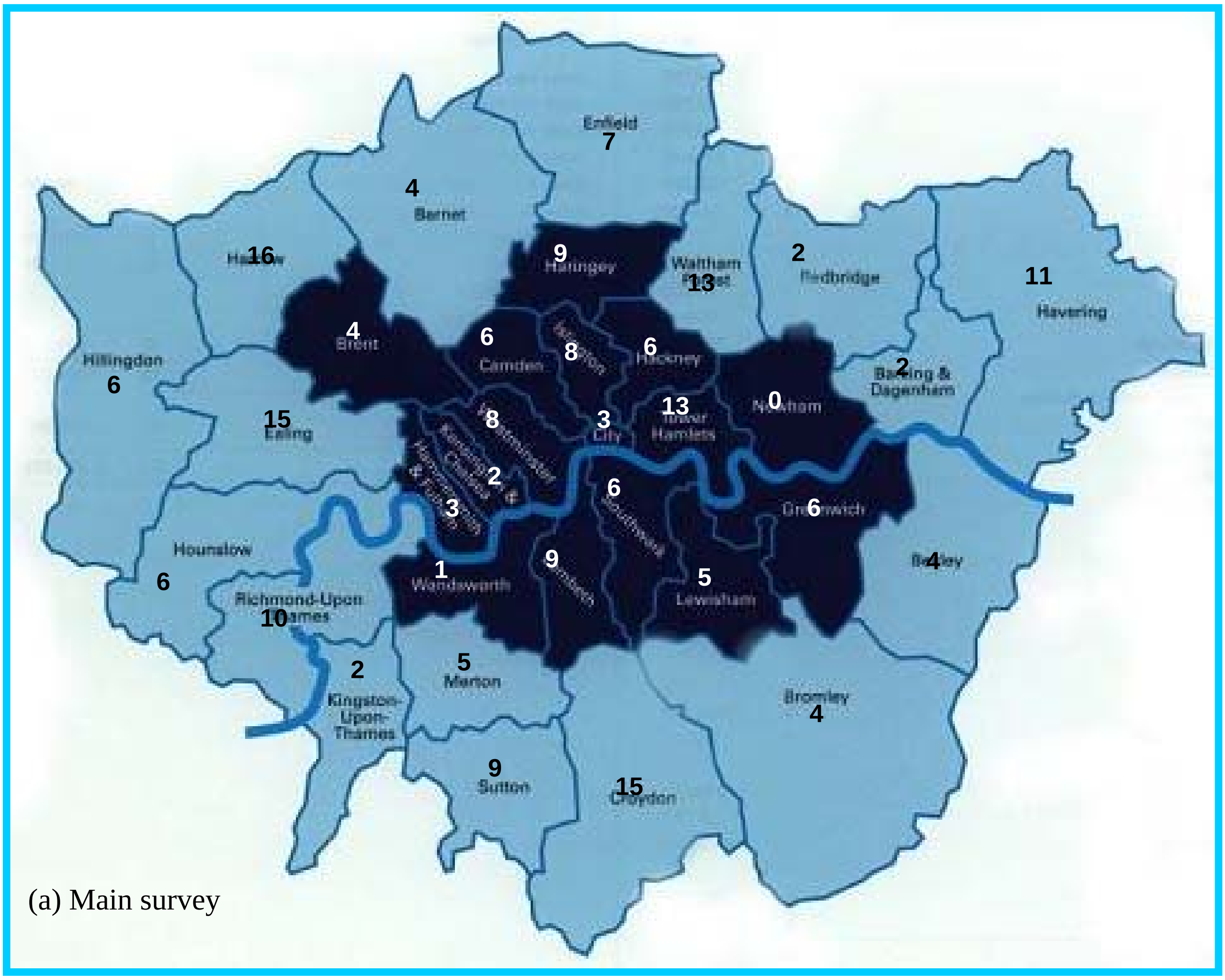
(c) Summer 2003

Spp	Bush	LCL	UCL	No Bush	LCL	UCL	*D*
	104			**39**			
Blackbird	2.55	1.99	3.11	1.94	1.24	2.65	0.37
Blue Tit	1.33	0.98	1.67	1.02	0.56	1.48	0.27
Carrion Crow	1.00	0.79	1.21	1.32	0.70	1.94	0.21
Feral Pigeon	4.90	2.64	7.16	3.45	0.65	6.26	1.23
House Sparrow	0.73	0.44	1.01	0.66	-0.12	1.43	0.26
Starling	3.64	2.42	4.85	2.62	1.39	3.85	0.64
Woodpigeon	2.02	1.44	2.60	1.50	1.03	1.96	0.33

(d) Winter 2003/04

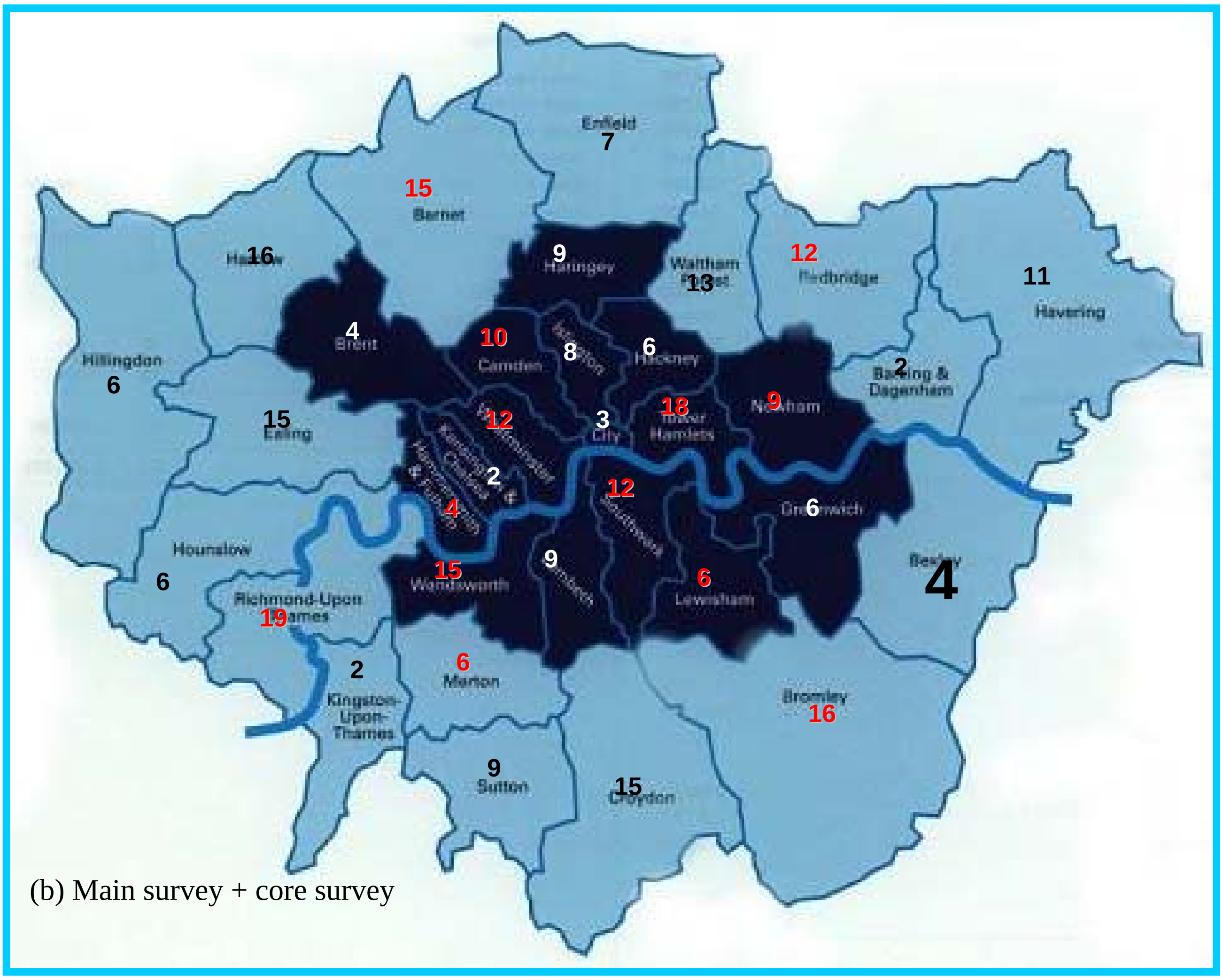
Spp	Bush	LCL	UCL	No Bush	LCL	UCL	*D*
	101			**36**			
Blackbird	1.85	1.33	2.36	1.25	0.71	1.80	0.32
Blue Tit	1.60	1.22	1.98	1.16	0.65	1.67	0.29
Carrion Crow	1.38	1.03	1.73	1.35	0.66	2.05	0.26
Feral Pigeon	6.29	3.04	9.54	2.41	0.37	4.44	1.35
Greenfinch	0.83	0.44	1.21	0.45	0.18	0.71	0.24
House Sparrow	0.71	0.43	0.99	0.36	0.10	0.62	0.25
Long-tailed Tit	0.65	0.35	0.96	0.79	0.16	1.42	0.26
Redwing	2.33	0.61	4.05	1.40	0.53	2.27	0.61
Starling	2.87	1.58	4.16	2.07	0.88	3.27	0.75
Woodpigeon	3.42	2.21	4.63	3.34	1.96	4.71	0.55

Table 1.10 A comparison of species density (birds/ha) between those sites with deciduous trees and mown grass and those with deciduous trees, mown grass and bushes. Significant ($P < 0.05$) differences only are shown.



(a) Main survey

Figure 1.1 The distribution of sites covered according to borough (a) in the main survey only and (b) in the main and core survey combined. Inner boroughs are shaded black, outer boroughs are shaded grey.



(b) Main survey + core survey

Figure 1.1 Continued.

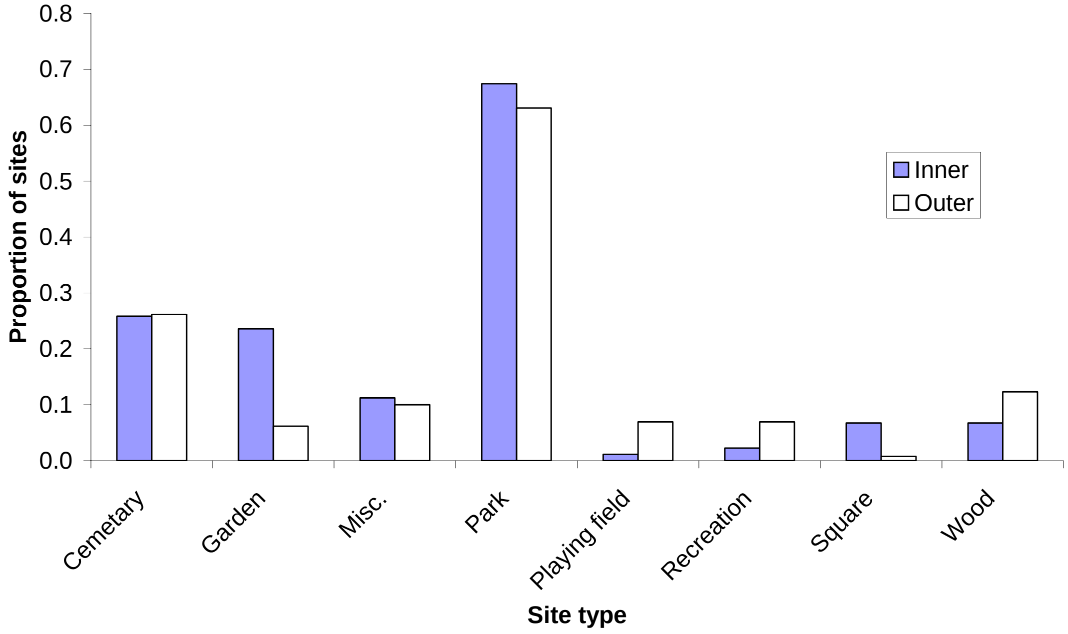


Figure 1.2 The proportion of sites of different type in Inner and Out boroughs of London.

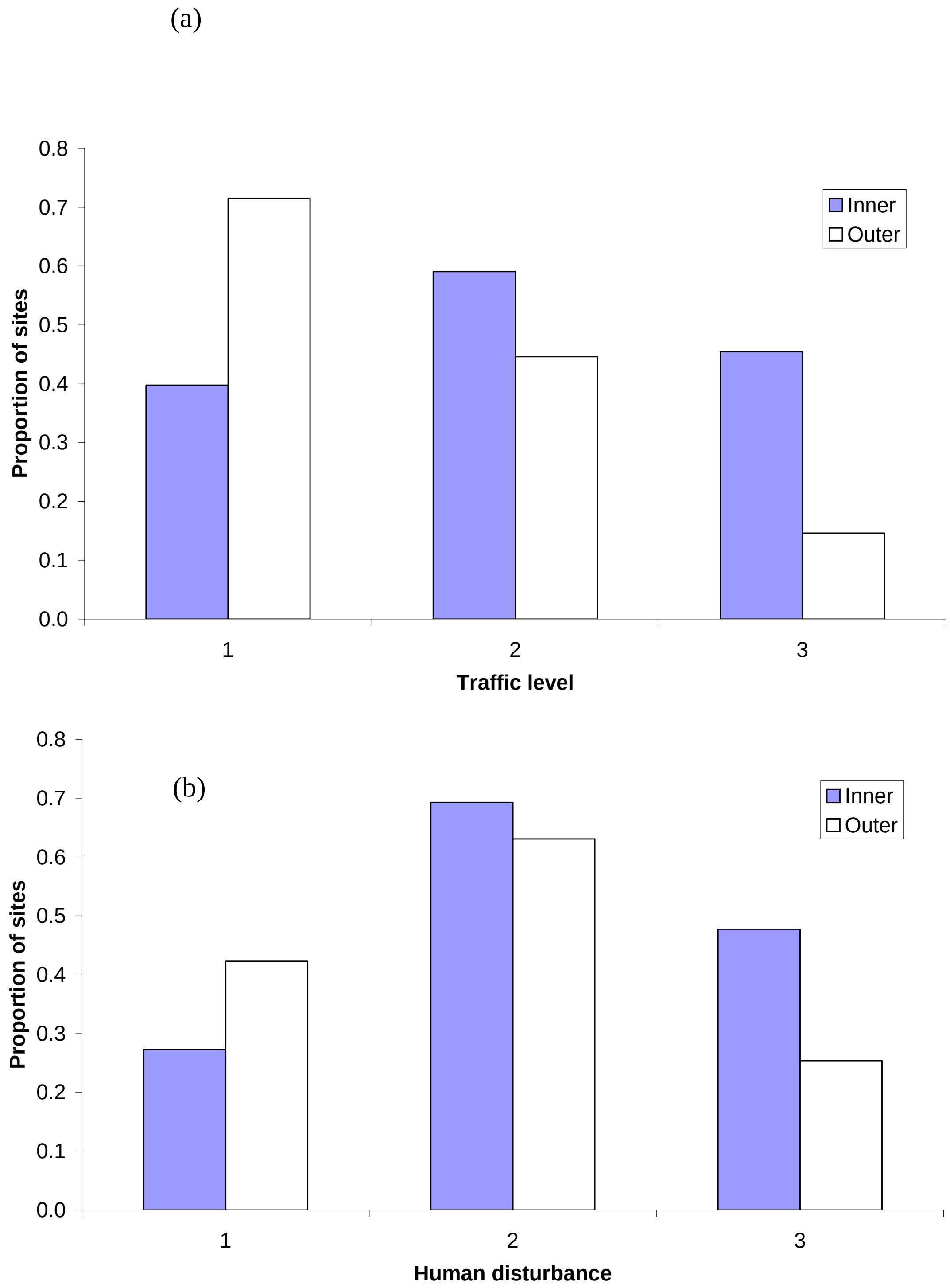


Figure 1.3 Traffic levels (a) and human disturbance levels (b) in green spaces in Inner and Outer boroughs of London.

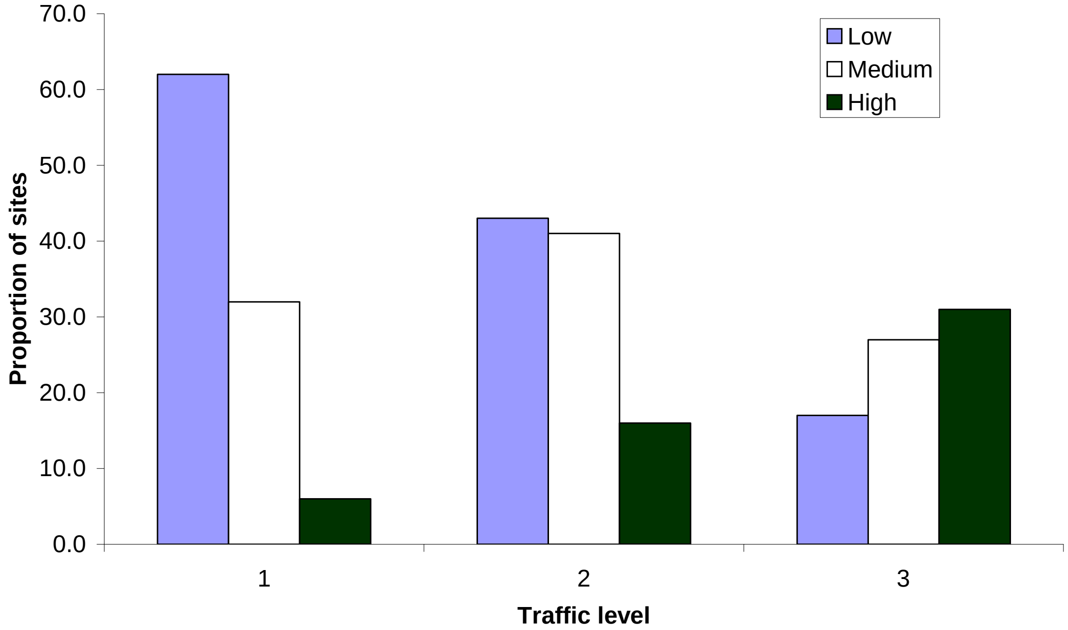


Figure 1.4 The proportion of sites of low, medium or high human disturbance within each traffic level (1,2 or 3).

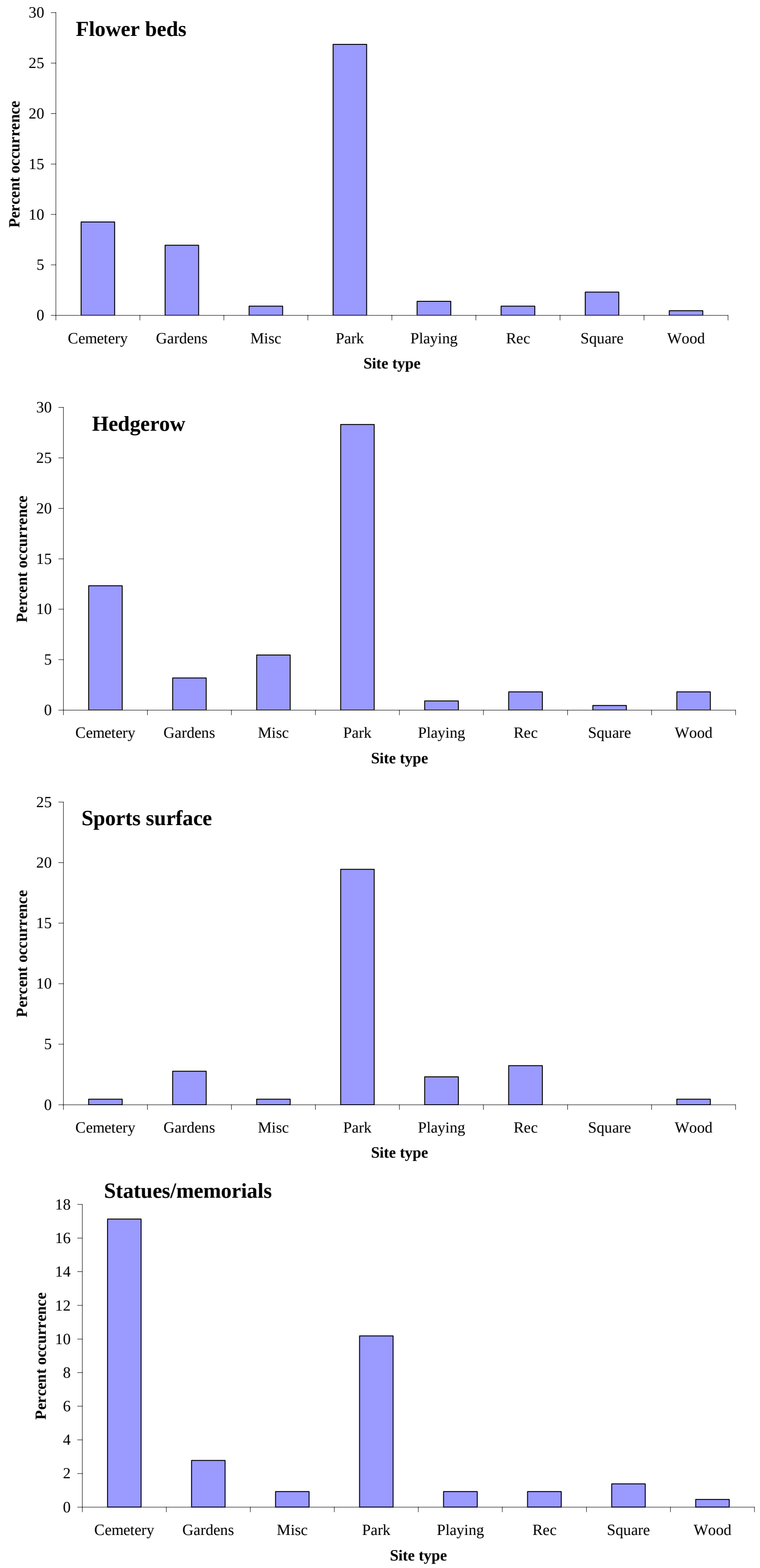


Figure 1.5 The proportion of sites in which a habitat was present according to site type. Sample sizes are given in Table 1.2. The proportion varied significantly (χ^2 tests) in each case, when the site types with lower sample sizes (squares, woods, playing fields) were omitted.

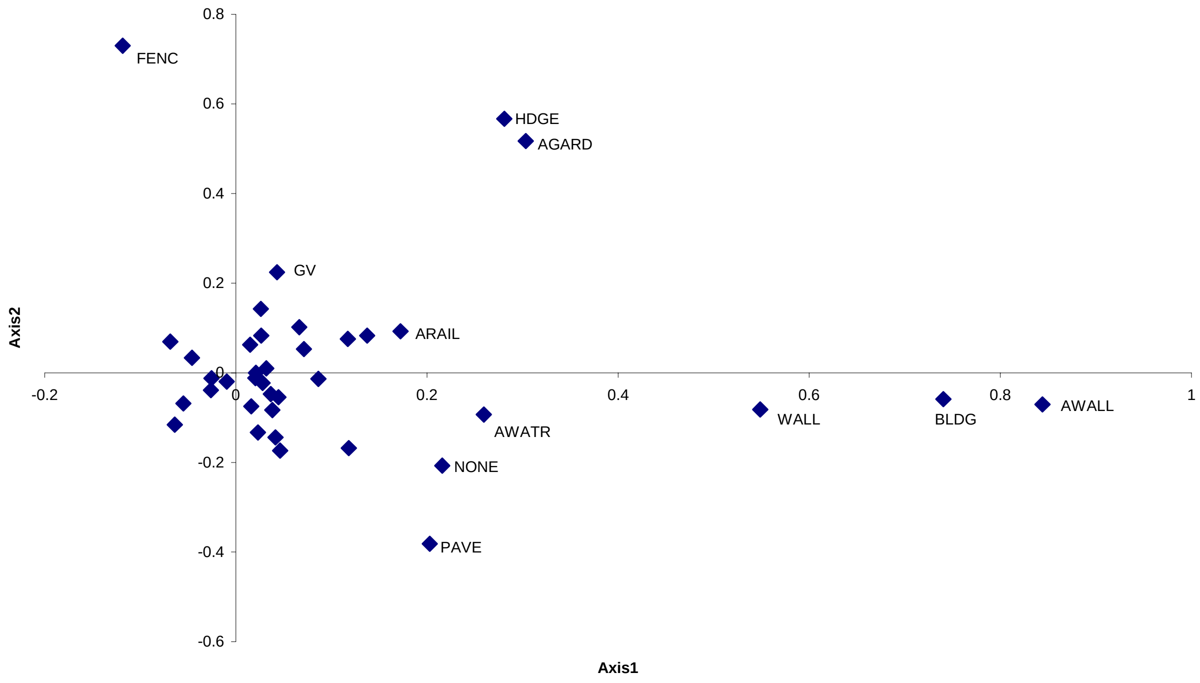


Figure 1.6 Bi-plot of PCA scores for the first two axes. Only variables with the most extreme scores on the axes are labelled.

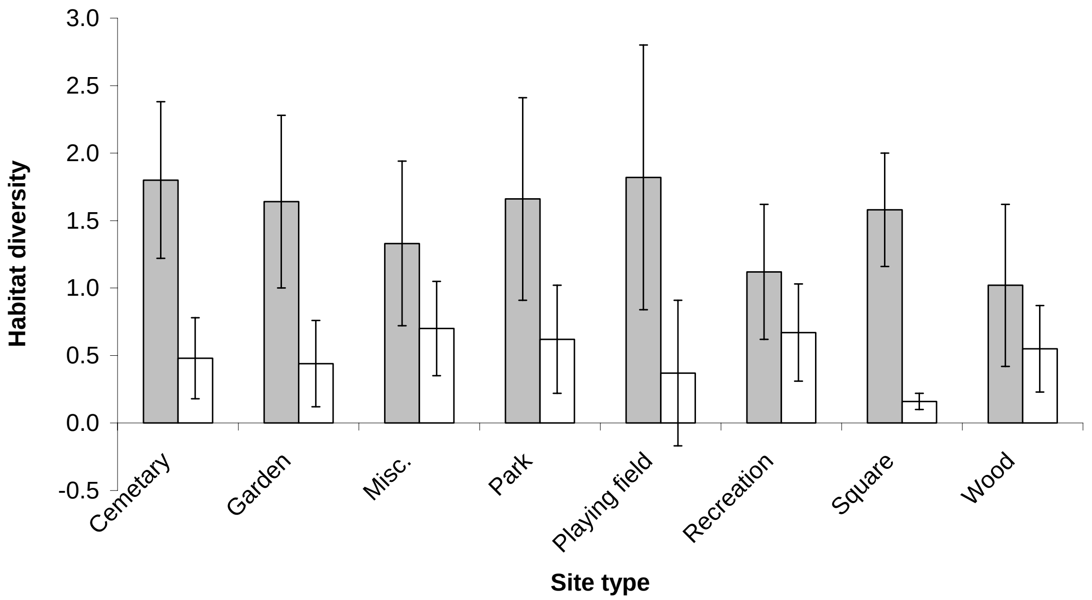


Figure 1.7 Mean (±SD) habitat diversity (shaded bars) and site boundary diversity (white bars) in different site types.

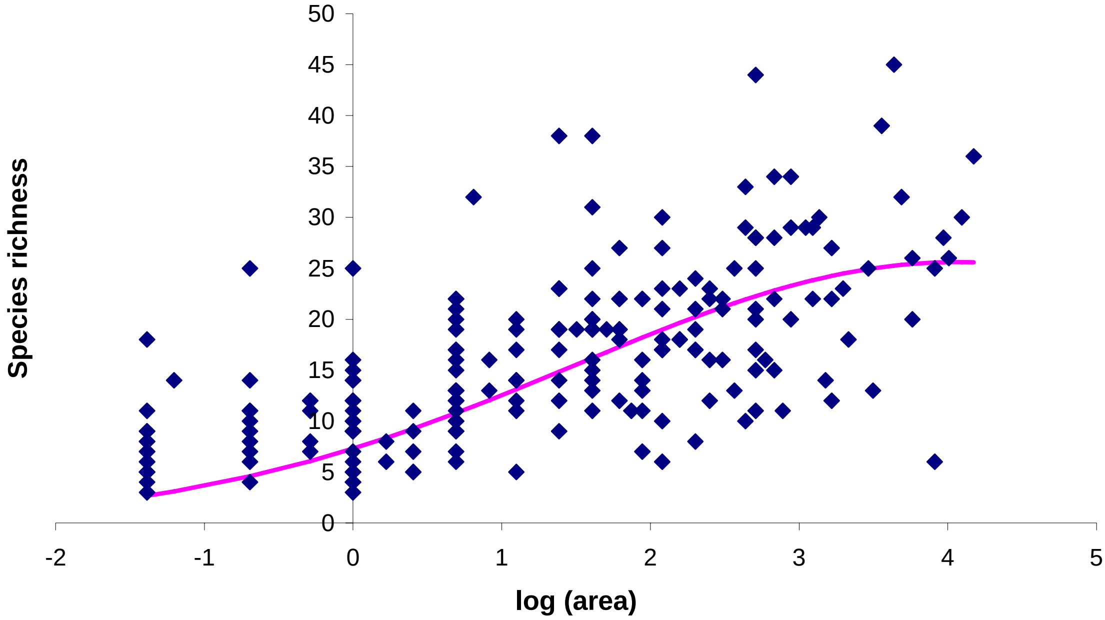


Figure 1.8 Variation in species richness with site area in summer 2003. The curve was fitted using Poisson regression. This pattern was consistent in other seasons/years.

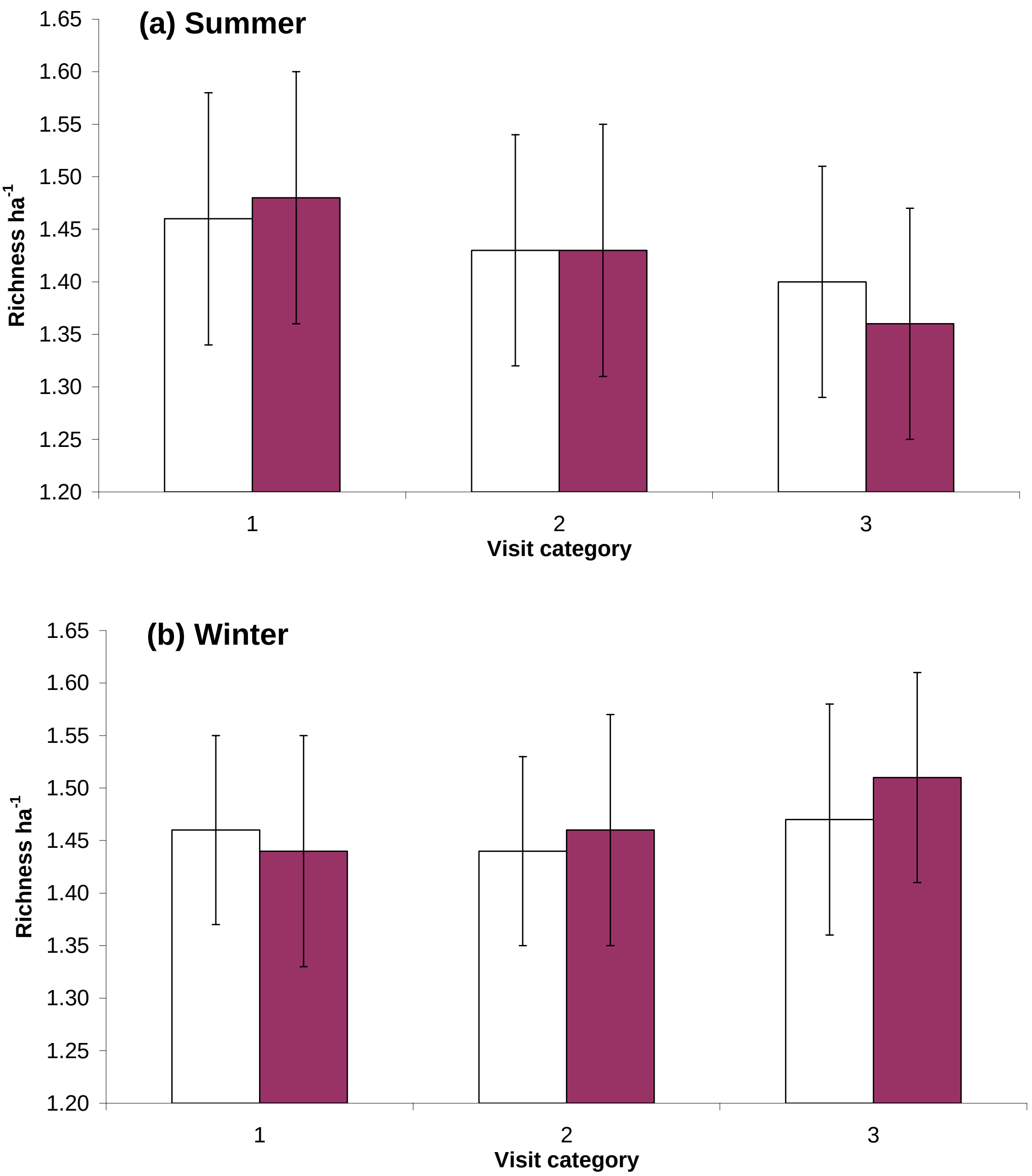


Figure 1.9 Mean (±95% CL) species richness per hectare in different sampling periods. White bars = 2002/03, shaded bars = 2003/04. Differences were significant between visit categories with the exception of winter 2003/04. Sample sizes were between 185 and 204.

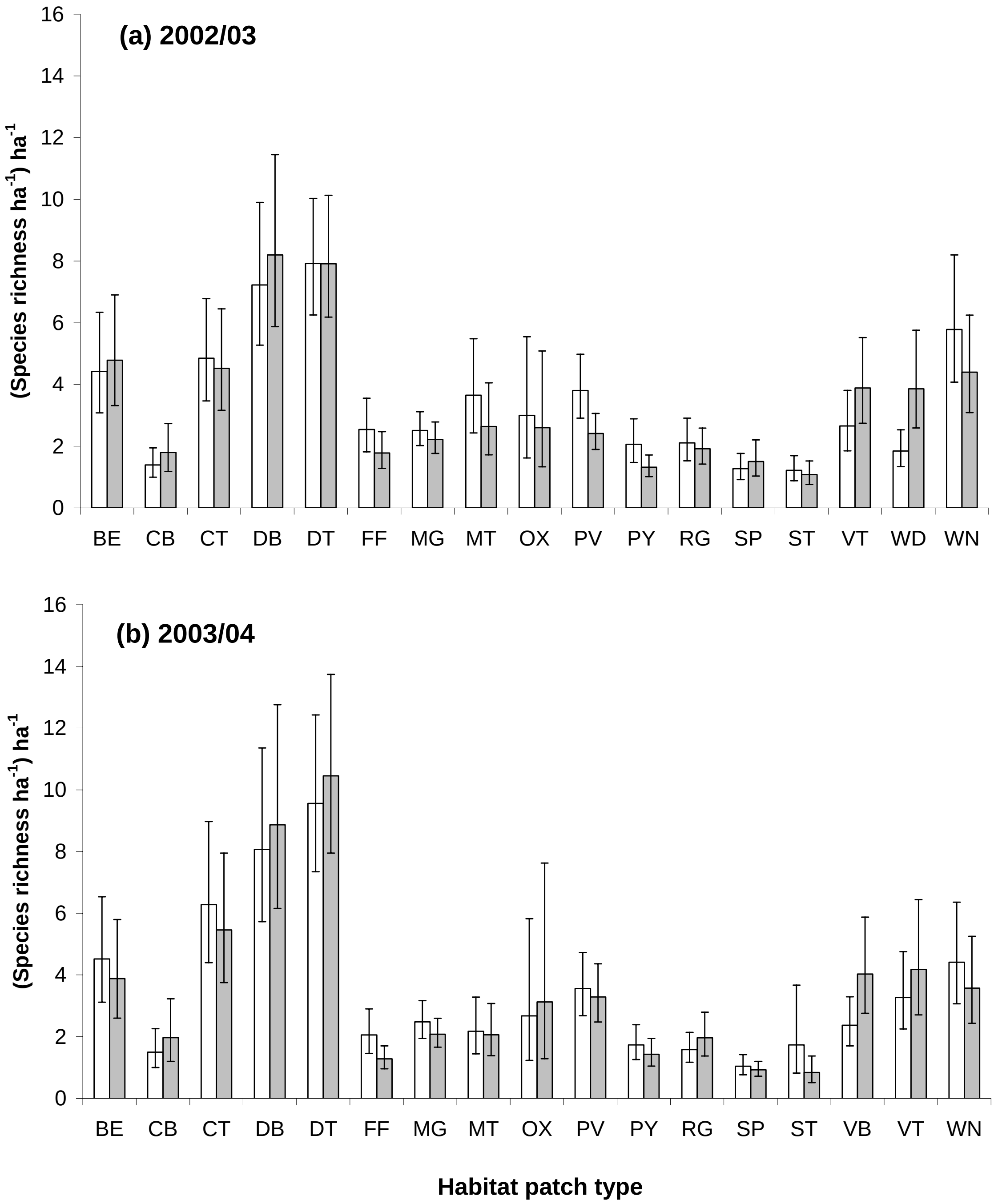


Figure 1.10 Species richness per hectare in different habitat types (codes given in Table 1.1). Error bars represent 95% confidence limits. Summer = white bars, winter=grey bars. Means are derived from regression models (GEE estimates) with site area as an offset.

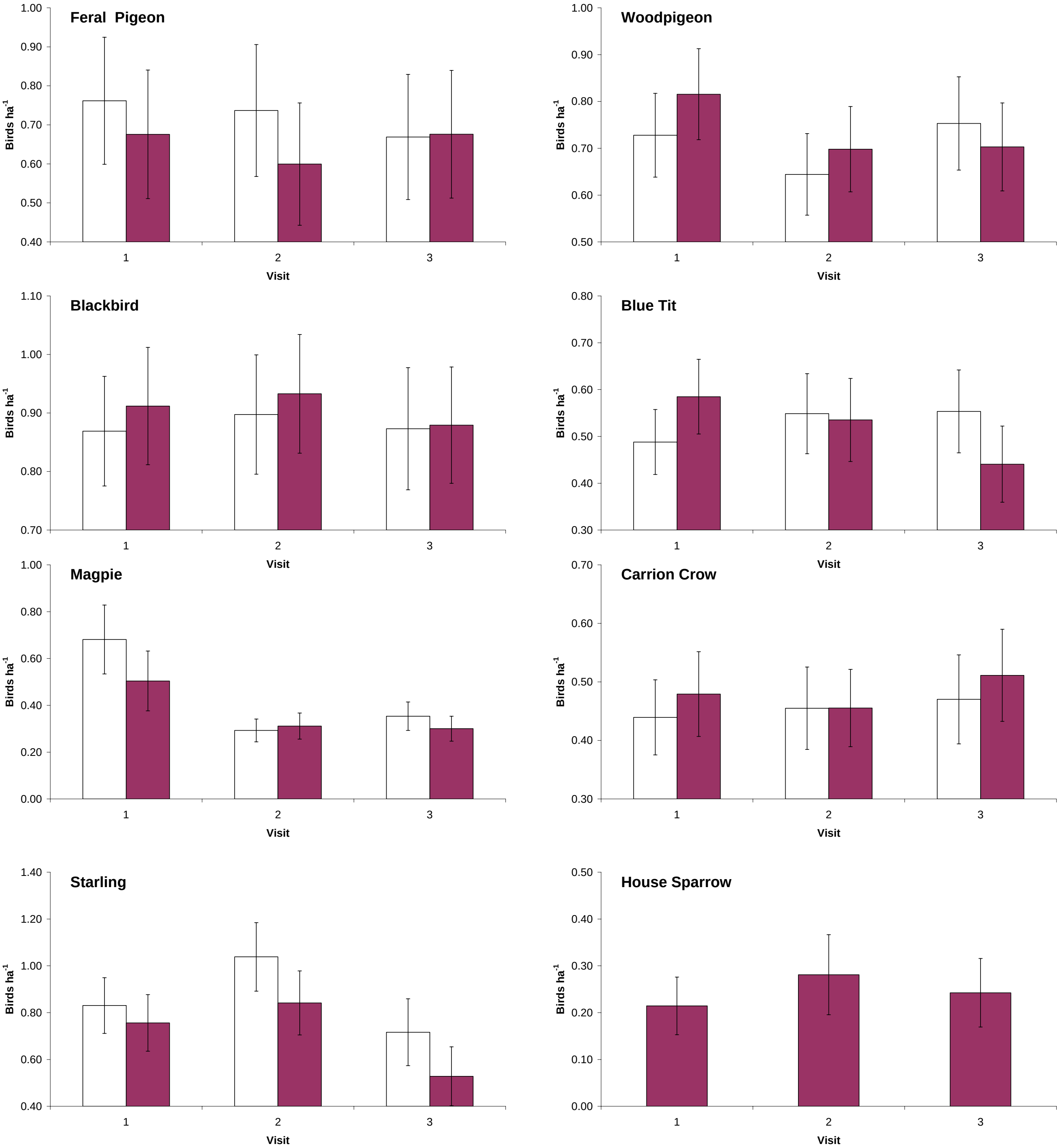


Figure 1.11 Mean density of birds in London green spaces in different visit periods in summer 2002 (light bars) and 2003 (shaded bars). Only seasons where the difference was significant between visit periods are shown. s1 = mid-April to mid-May; s2 = mid-May to mid-June; s3 = mid-June to mid-July. Sample sizes were between 187 and 204.

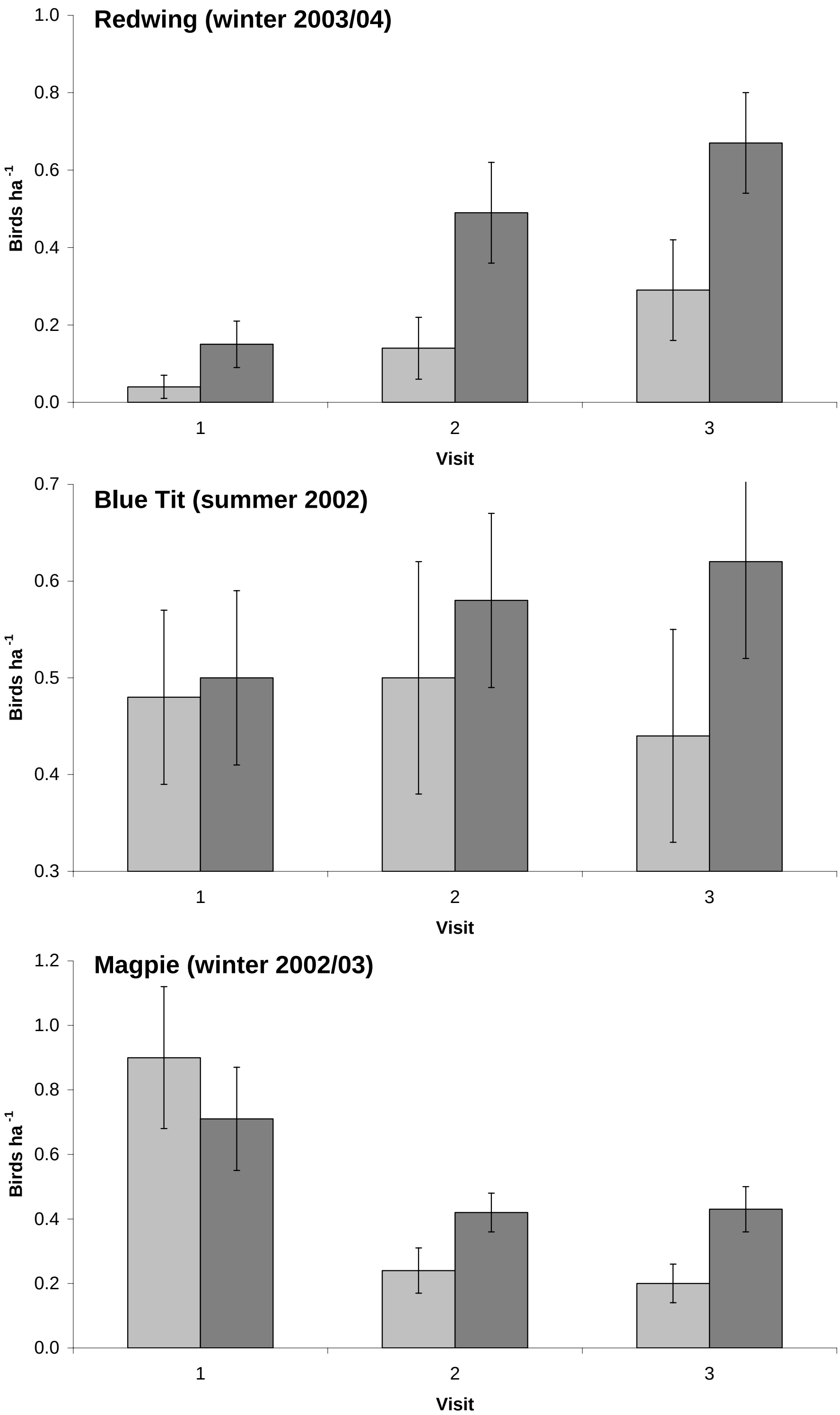


Figure 1.12 Mean density (± 95% CI) in different visits in inner (light shading) and outer (dark shading) boroughs. Interactions between visit and location were significant in each case.

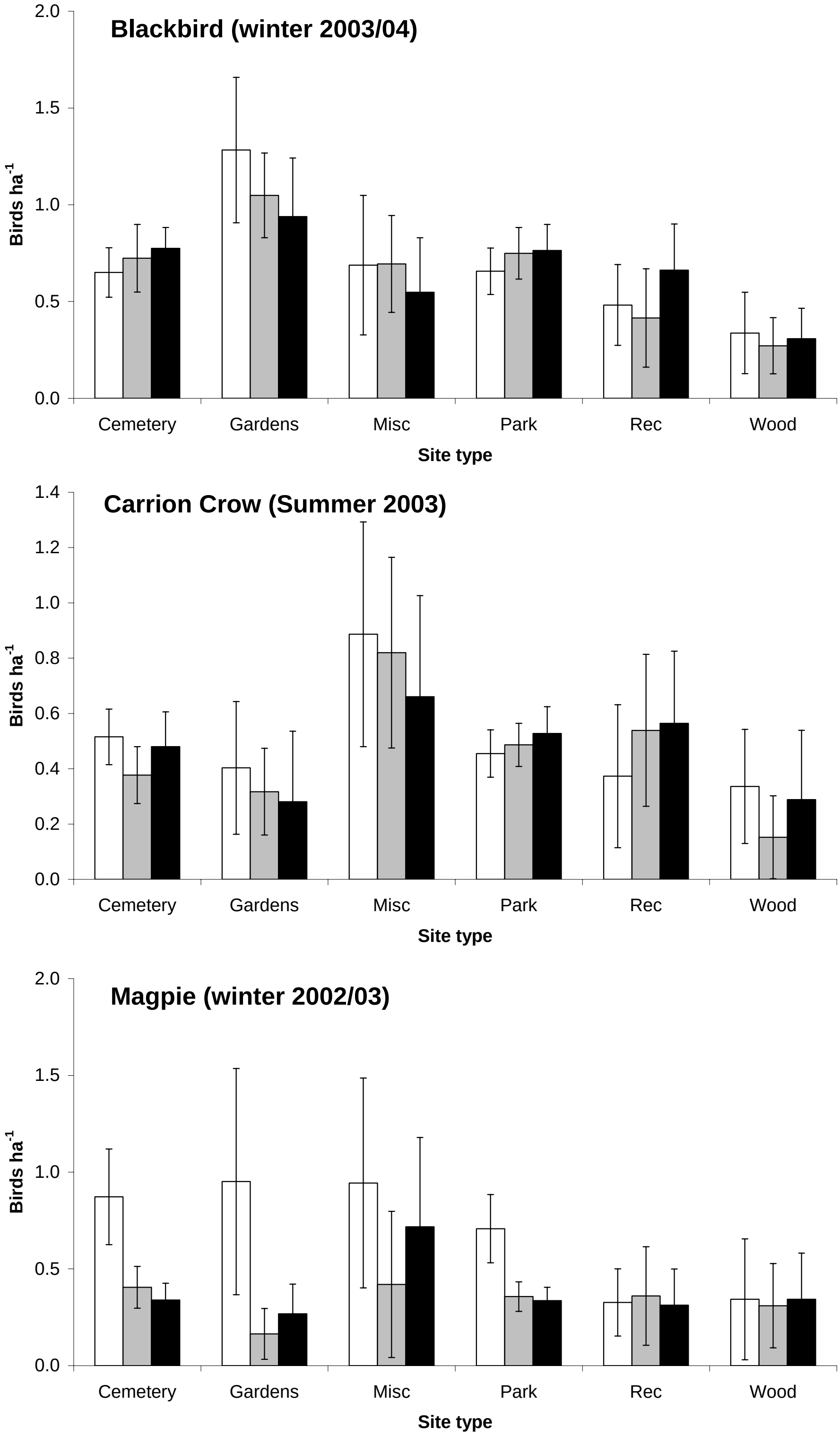


Figure 1.13 Mean (±95% CL) density of species in different site types in different visit periods where white bars = visit 1, shaded bars = visit 2, black bars = visit 3. Each species showed a significant site*visit interaction.

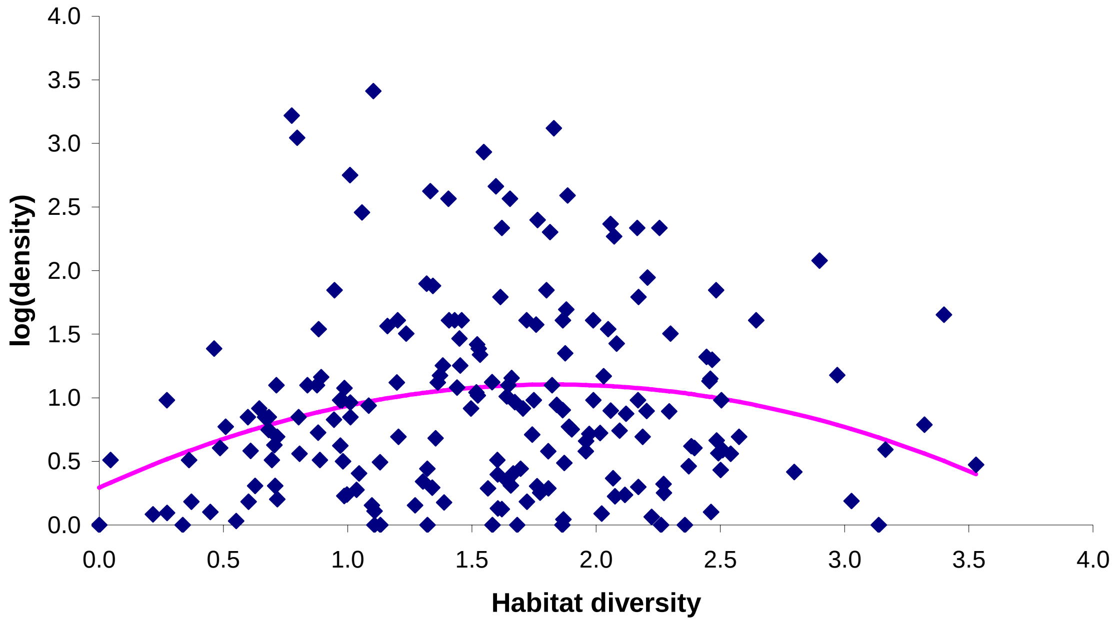


Figure 1.14 The relationship between Woodpigeon density and habitat diversity (Shannon index) in summer 2003. The curve was fitted from a quadratic regression.

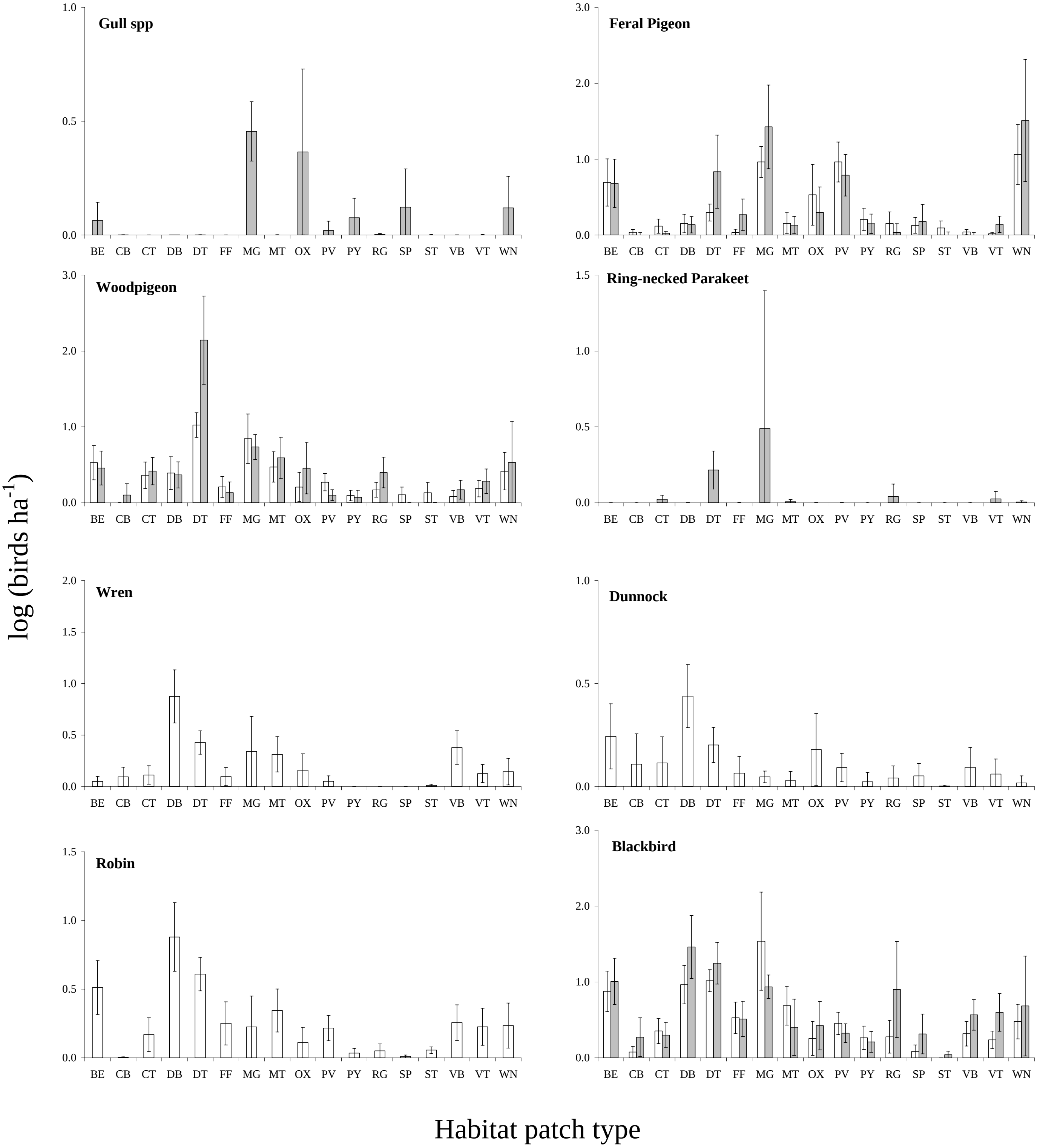


Figure 1.15 Mean (±95% CL) densities (log x+1 transformed) of species in different habitat types determined at the patch level in 2002/03 in different seasons where white bars = summer, shaded bars = winter. Results are not shown where there was no significant difference between habitat types or where *D* < 0.20 or *D* > 10. Habitat codes are given in Table 1.1.

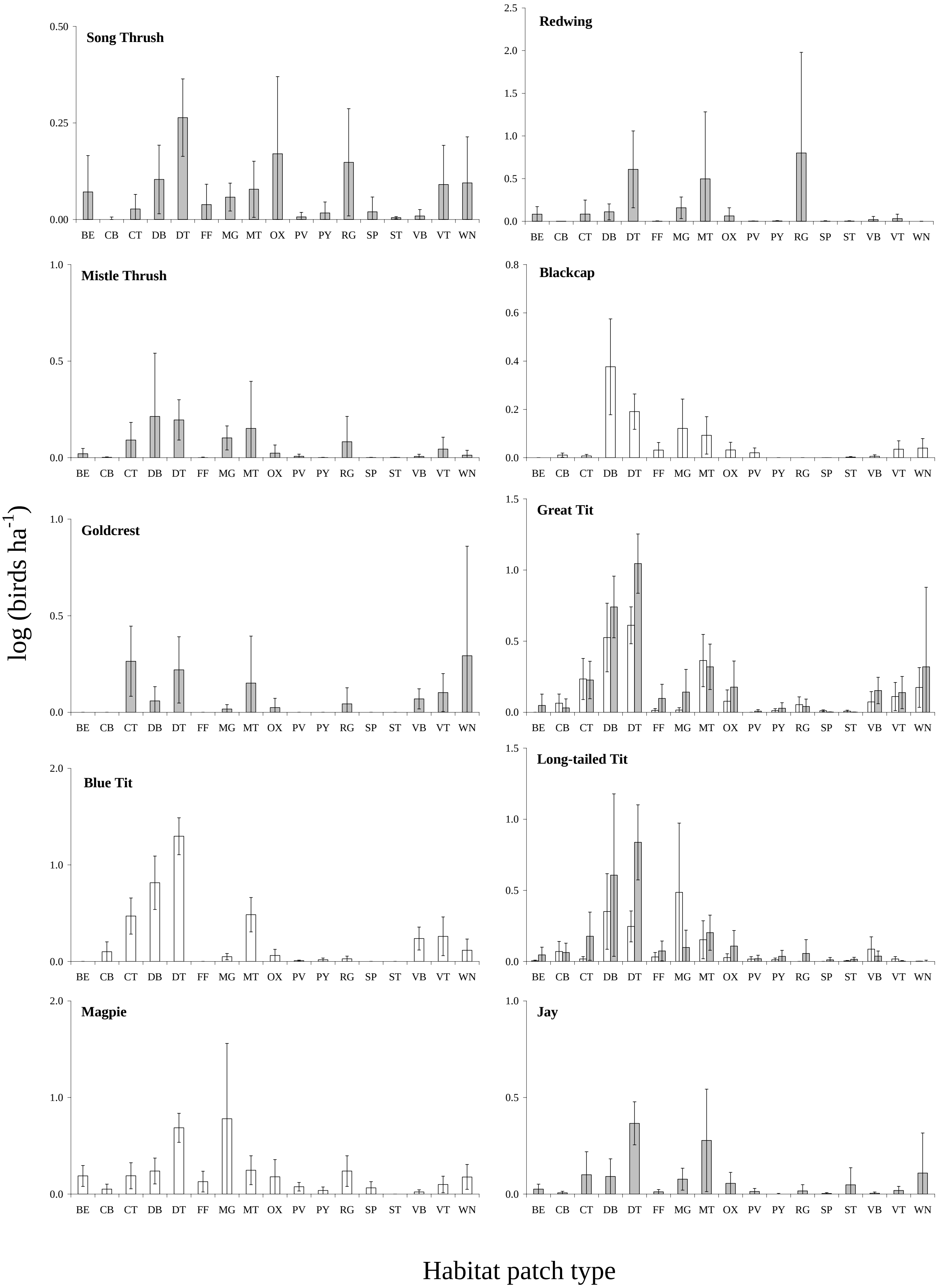


Figure 1.15 Continued.

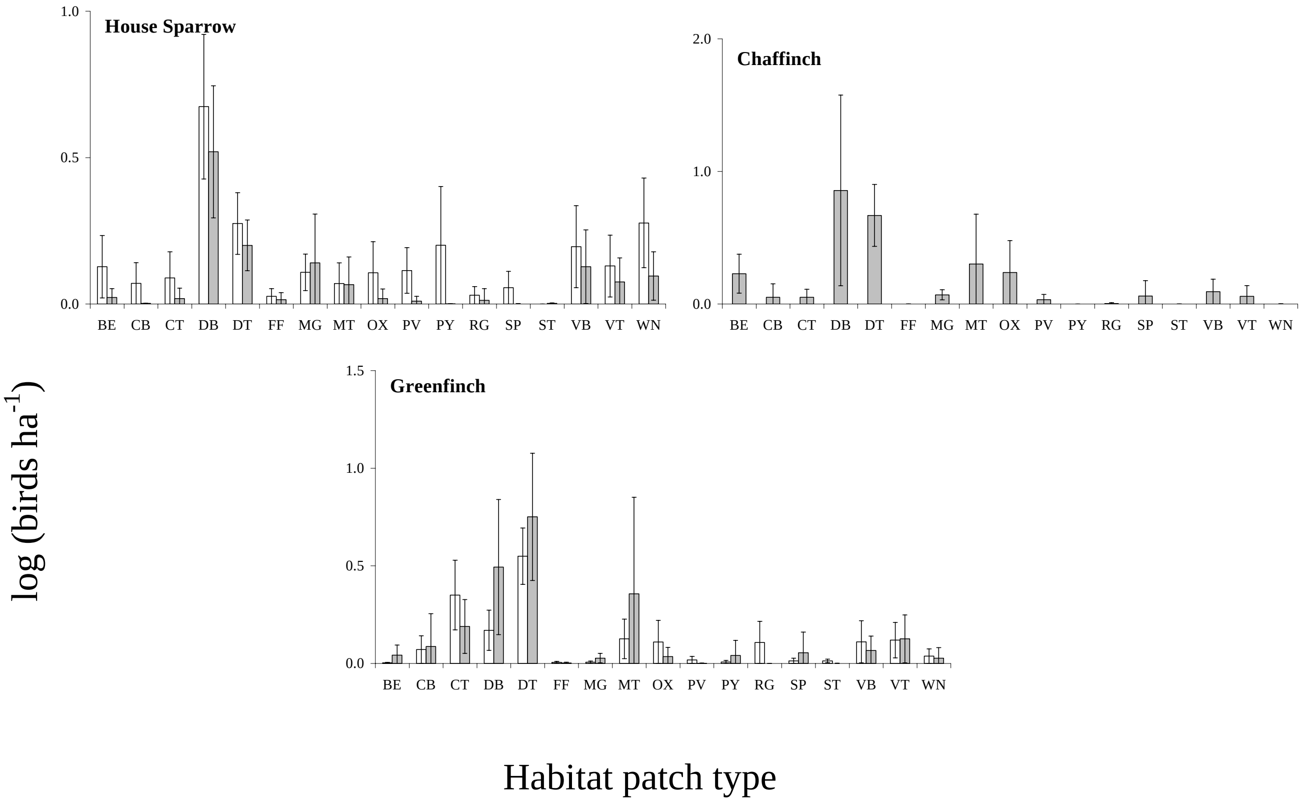


Figure 1.15 Continued.

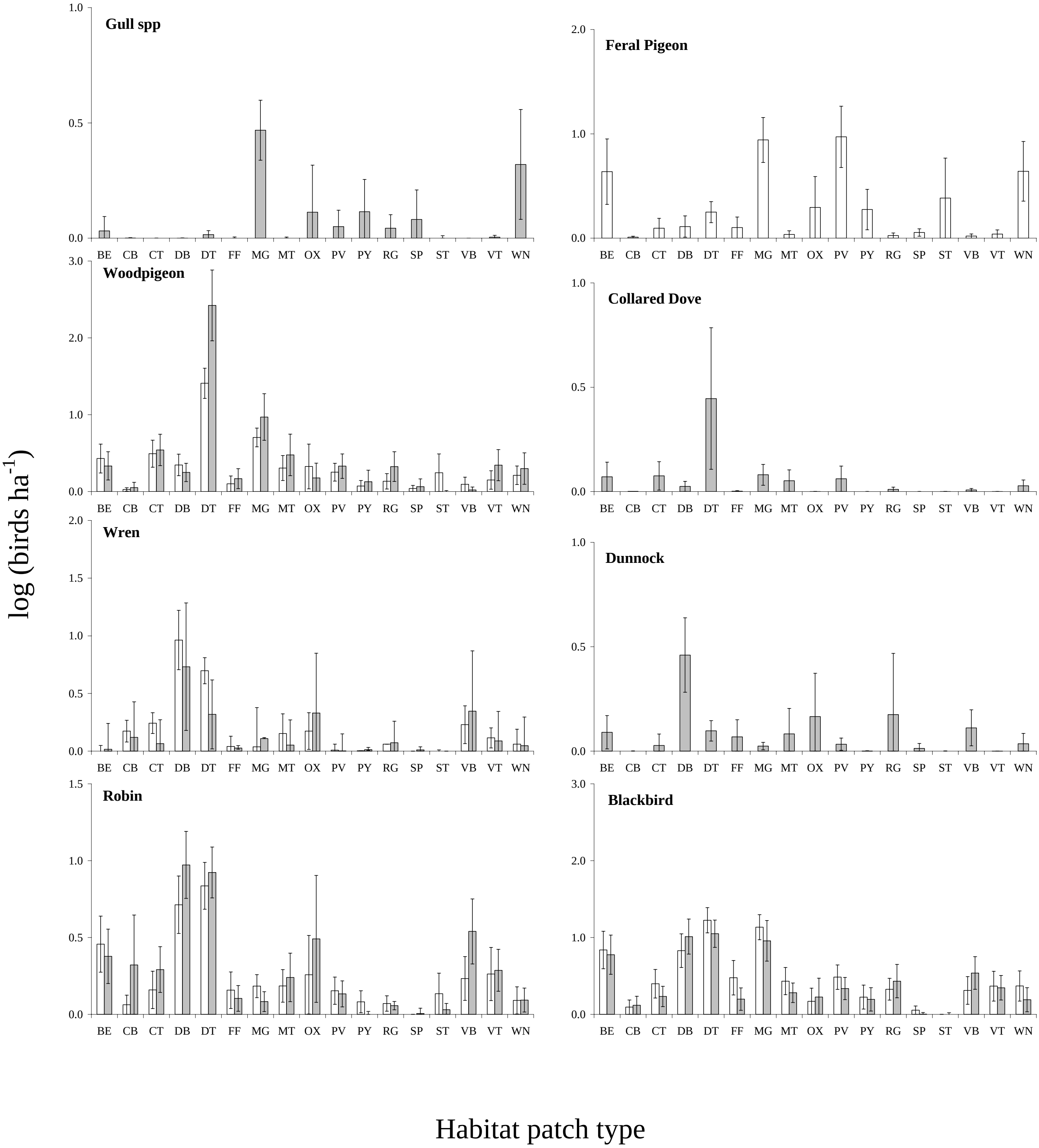


Figure 1.16 Mean (±95% CL) densities of species in different habitat types determined at the patch level in 2003/04 in different seasons where white bars = summer, shaded bars = winter.

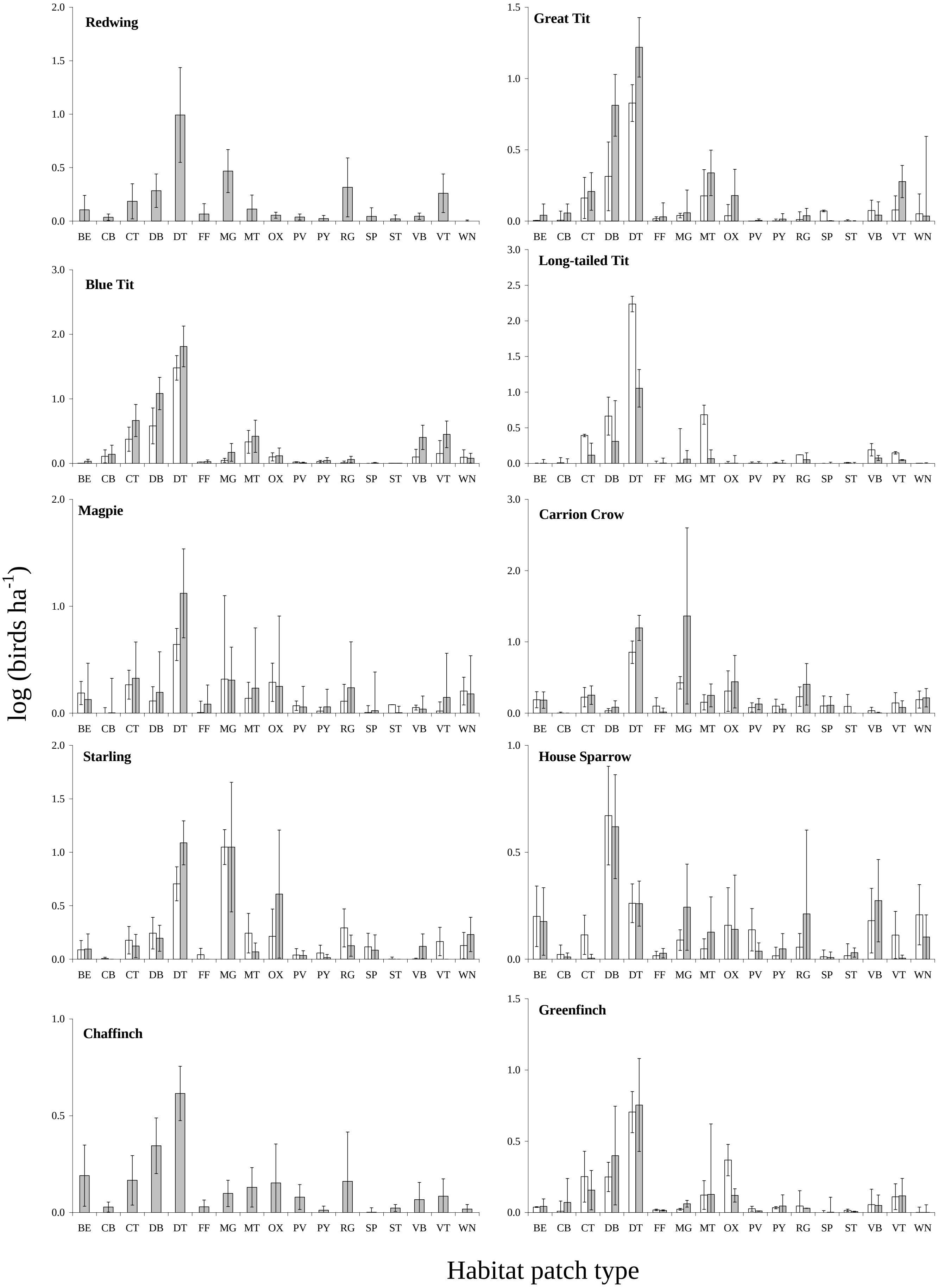


Figure 1.16 Continued.

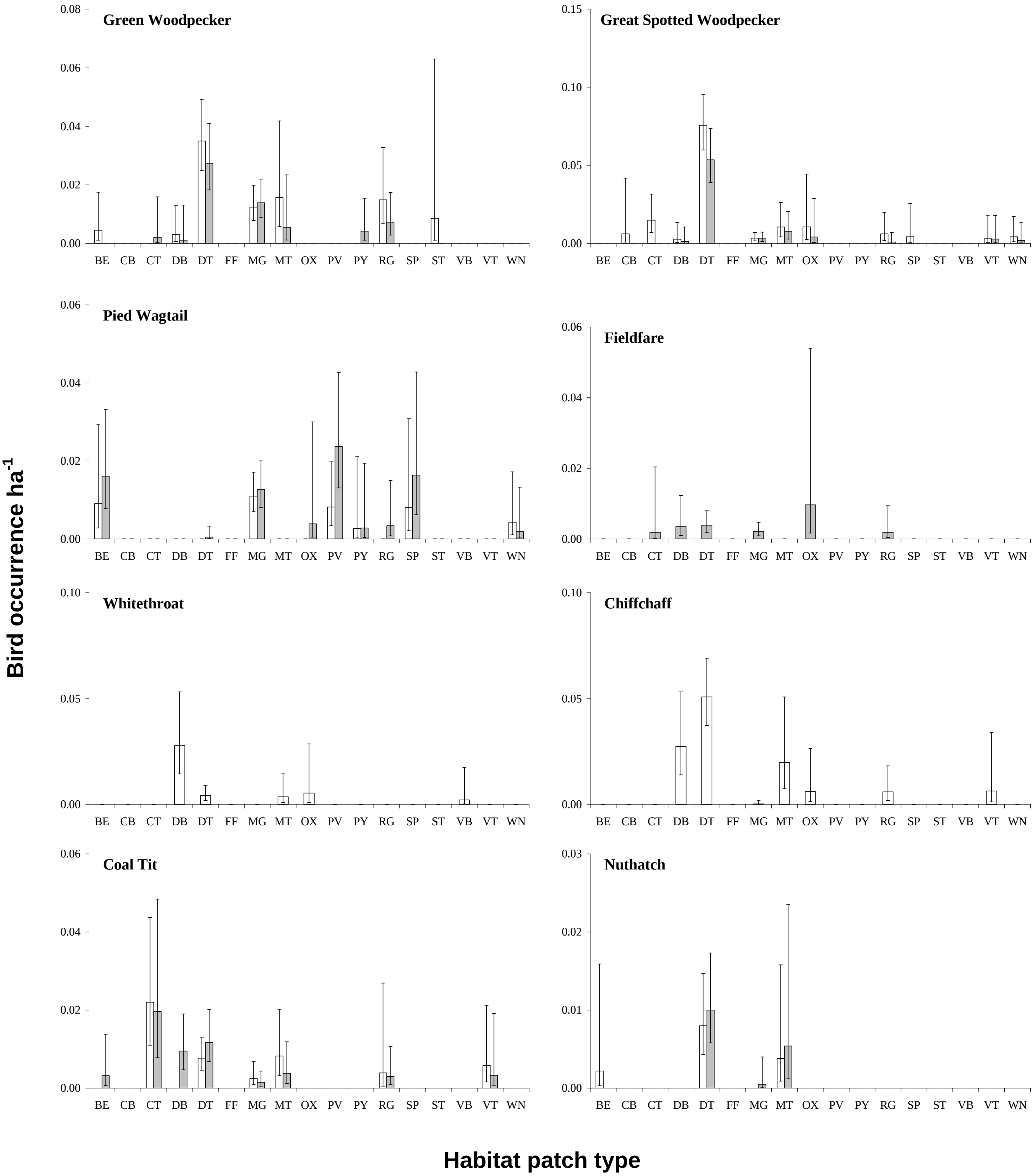


Figure 1.17 Mean (±95% CL) probability of occurrence per ha of species in different habitat types determined at the patch level in 2002/03 in different seasons where white bars = summer, shaded bars = winter.

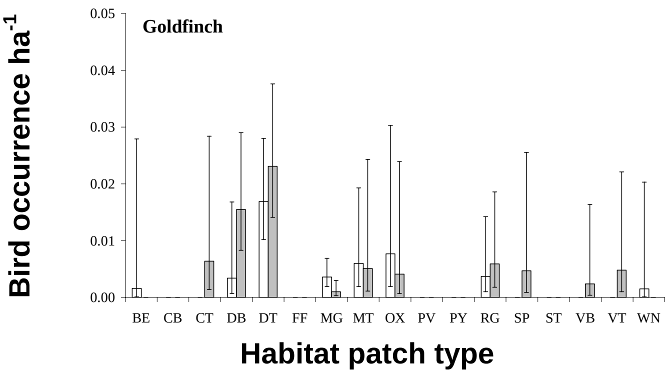


Figure 1.17 Continued.

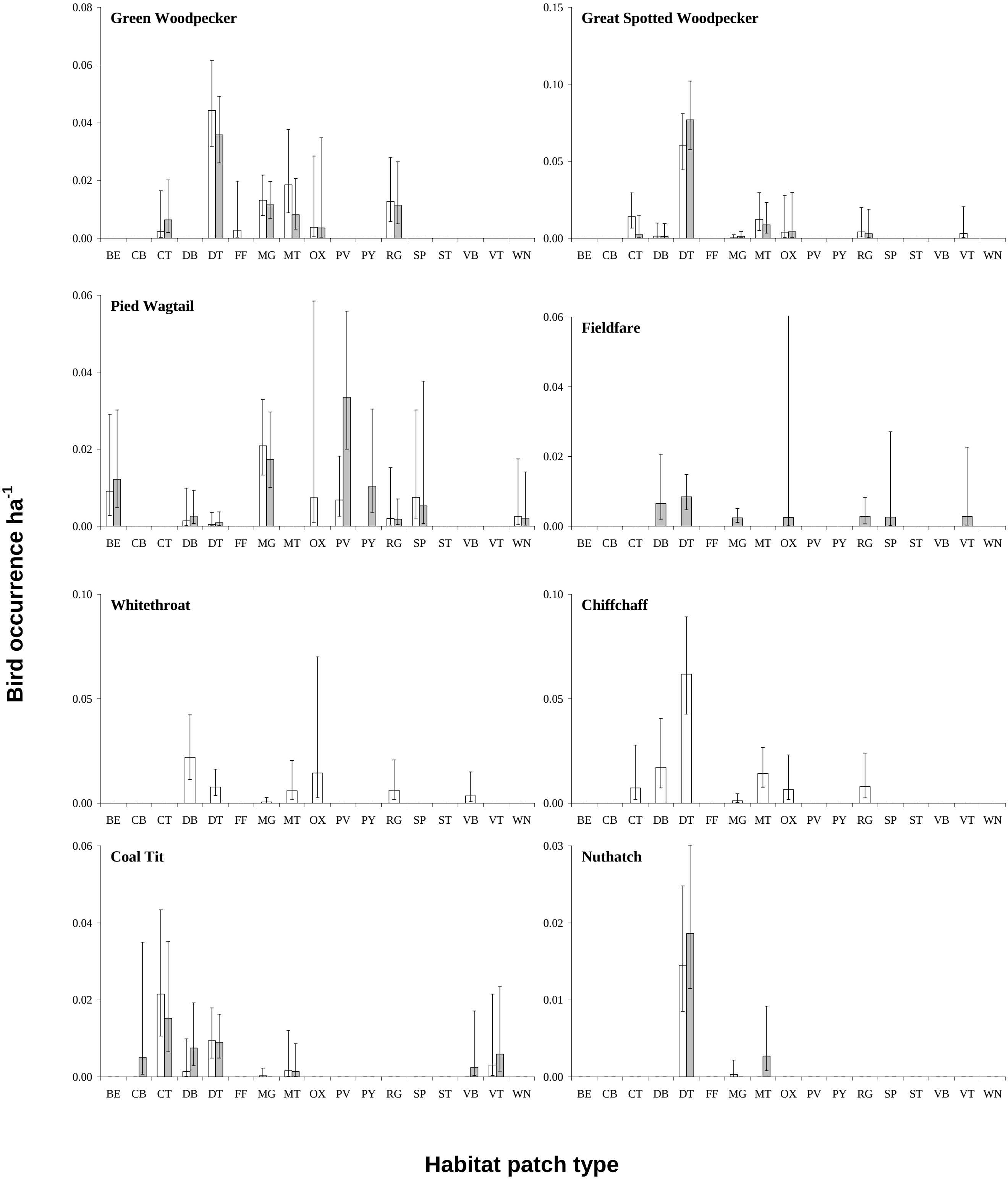


Figure 1.18 Mean (±95% CL) probability of occurrence per ha of species in different habitat types determined at the patch level in 2003/04 in different seasons where white bars = summer, shaded bars = winter.

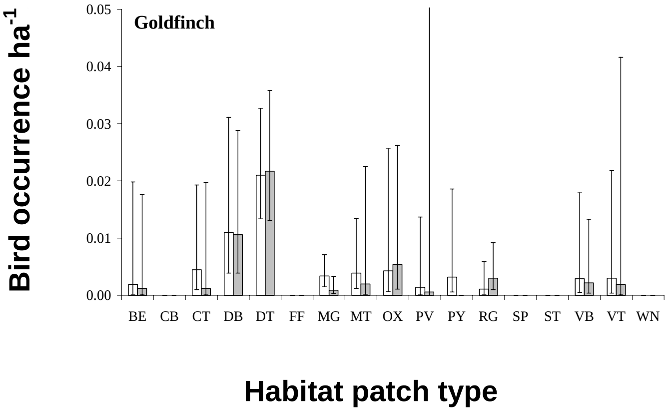


Figure 1.18 Continued.

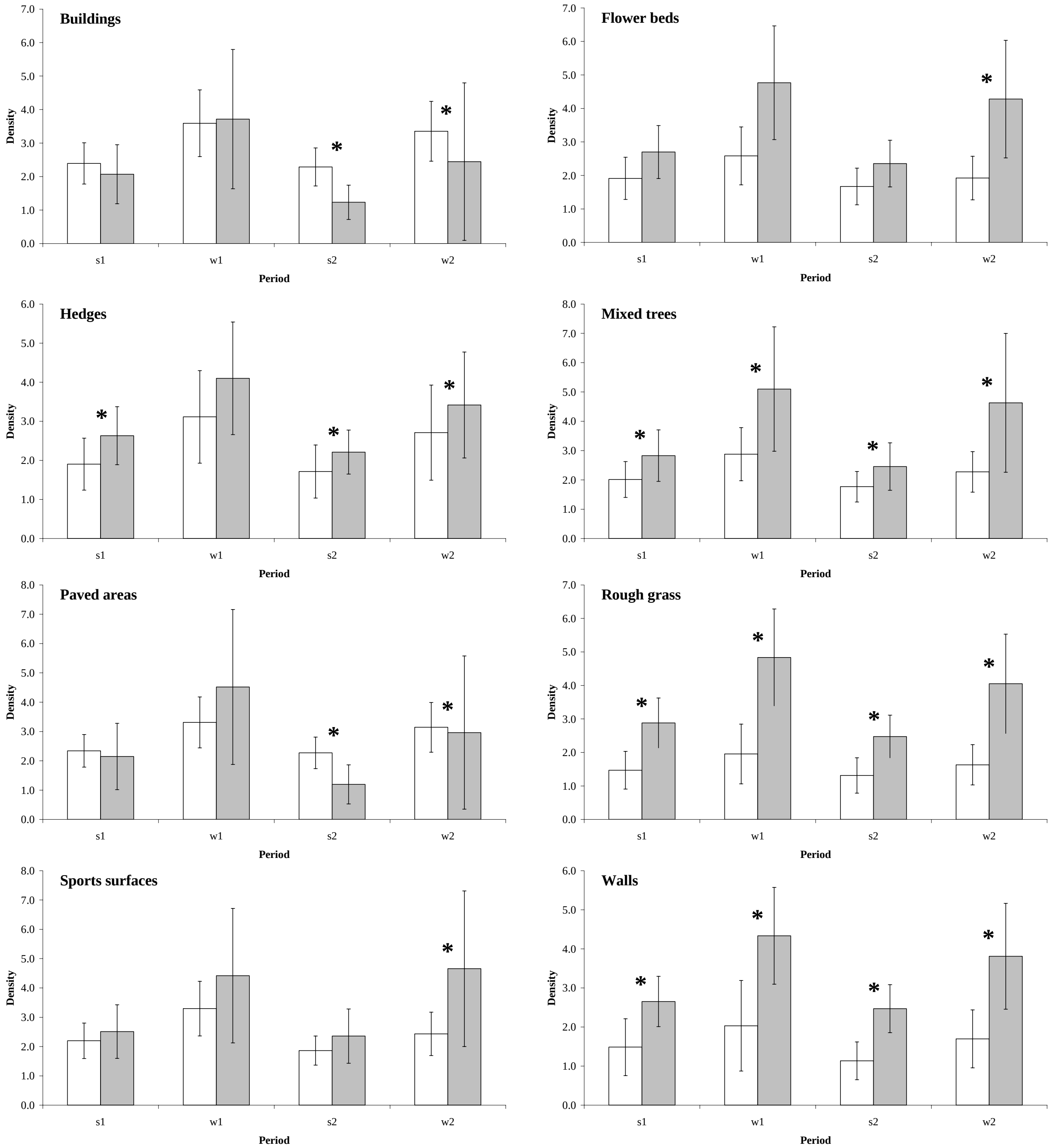


Figure 1.19 Mean (±95% CL) Grey Squirrel density in sites where a given habitat type was either absent (white bars) or present (shaded bars). Asterisks indicate where differences between medians were significant (Wilcoxon test). Periods : s1 = summer 2002, w1 = winter 2002/03, s2 = summer 2003, w2 = winter 2003/04.

CHAPTER 2 HABITAT ASSOCIATIONS OF BREEDING BIRDS IN LONDON'S PUBLIC GREEN SPACES

Dan Chamberlain, Su Gough, Howard Vaughan

1. INTRODUCTION

Surveys in the breeding season often include all individuals whether they show evidence of breeding or not (e.g. Breeding Bird Survey (BBS) in Chapter 3 and Garden BirdWatch in Chapter 4), yet it is often the breeding part of the population that is of primary interest. However, in the case of the BBS at least, there is a correlation between total numbers and breeding numbers on a national scale for most species (Freeman *et al*. in prep.). However, the distinction between total individuals and breeding individuals is likely to be important when considering habitat associations as breeders and non-breeders may occupy different habitats. For example, Krebs (1971) found that Great Tits bred preferentially in broad-leafed woodland. Non-breeders tended to occupy farmland hedgerow which was a poorer habitat for the (typically subordinate) birds who did manage to breed there. Numerous similar examples exist in other species (Newton 1998). The Great Tit example raises another issue, in that even if a species is attempting to breed, productivity may vary greatly between habitats. In surveying only breeding birds, assumptions are still being made about productive breeding. Nevertheless, by analysing data derived only from those individuals showing some evidence of breeding, it is possible to increase the likelihood of surveying that part of a population that contributes to the next generation.

2. AIMS

The aims of this chapter were essentially similar to those in Chapter 1, except that bird-habitat associations of species showing evidence of breeding were analysed.

3. METHODS

For the core survey, methods were similar to that used in the main survey (Chapter 1), only the location of all birds seen and heard were actually recorded onto maps of each study site. In addition, activity codes were used in order to interpret bird behaviour. The methods were therefore very similar to standard intensive mapping techniques (Bibby *et al*. 2000) used in monitoring surveys such as the Common Birds Census (Marchant *et al*. 1990). The only difference was that a much reduced number of visits was carried out (3 rather than a minimum of 10). The advantages of such methods is that they enable a more accurate assessment of whether a given species is likely to be attempting to breed at a given site by noting behaviour such as singing, feeding young or carrying nesting material. This fact, and the more accurate mapping involved, were the main differences between the core survey and the main survey. Note also that this survey had a largely professional element. There were 11 sites that used volunteers (out of 81) but even here, recruitment was specifically aimed at birdwatchers who had previous experience of such surveys. Recording and identification are therefore likely to be very accurate for this survey.

A survey was carried out on a sample of sites in 2002 (n = 44) and 2003 (n = 37). Note that in order to maximise the number of sites covered, a different sample of sites was carried out in the two years (i.e. the two years were mutually exclusive). Data were extracted on a patch-by-patch basis, where habitat patches were homogenous areas of habitat within each site defined by the observer (as in the main survey). There was no attempt made to identify territories as there would be in a full (i.e. 10+ visit) mapping survey. Instead, for birds showing evidence of breeding (henceforth referred to as breeding birds) the maximum count over all visits per season was taken as the analysis variable (after excluding records that were likely to have been the same individual). In the vast majority of cases, these were singing males. Habitat data were collected in the same was as the main survey. Counts of all birds were also taken in an identical manner to the main survey in both summer and winter and these data were included with the analysis of the main survey in Chapter 1.

Analytical methods were similar to those used in Chapter 1, where for individual species, log(density) was used as the independent variable in a normal regression model. However, the calculation of density was based on maximum count rather than mean count over the three visits in each season. This is a commonly applied method of determining breeding bird abundance (e.g. Chamberlain *et al*. 1999). Species richness was divided by area and log-transformed and then analysed with a normal regression model, using log(area) as an offset.

4. RESULTS

A summary of the occurrence (proportion of sites) of breeding birds and all birds recorded in the breeding season is given in Table 2.1. There were 43 breeding species in 2002 and 55 in 2003. A total of 64 species were recorded (note that gulls were identified to species as this survey used more experienced fieldworkers), 58 of these showing breeding evidence. Blue Tit was the most widespread species recorded in 2002 and Wren in 2003. There were 11 species that were potentially breeding in over 50% of sites in at least one summer: Blue Tit, Blackbird, Wren, Robin, Great Tit, Carrion Crow, Woodpigeon, Blackcap, Magpie, Starling and Dunnock. Generally occurrence rates were similar between years, but there were some notable differences. For example, breeding Great Tit were recorded in 80% of sites in 2002 but only 68% in 2003. Greenfinch showed a similar decrease from 48% to 24%. There were also some species that increased between years, e.g. Mistle Thrush from 25% to 51%. Also, Whitethroat was not recorded at all in 2002 but occurred on 16% of sites in 2003. It should be stressed that these differences do not necessarily reflect different populations in different years but could just be due to the different sites sampled as the sample was mutually exclusive between years.

There were some species of note recorded breeding. Starling and House Sparrow are both declining species and of some conservation concern. Starling was relatively common occurring in around 50% of sites (Table 2.1). However, House Sparrow occurred in 27% and 38% of sites in 2002 and 2003 respectively, relatively low proportions for a once abundant species. BAP species recorded breeding were: Song Thrush (c. 40%), Skylark, Linnet, Bullfinch and Reed Bunting (all >10% and mostly single records). There were also some woodland species of current conservation concern recorded including Willow Tit (recorded breeding in 11% of sites in the 2002 sample but absent in 2003), Lesser Spotted Woodpecker (potential breeding record from one site) and Goshawk (displaying bird at one site). However, Spotted Flycatcher was notable by its absence. Other interesting records included Ring-necked Parakeet (potentially breeding in c. 10% of sites), Kingfisher, Meadow Pipit, Lesser Whitethroat and Redpoll (isolated potential breeding records). Other notable species recorded but with no evidence of breeding included Turtle Dove, Woodlark and Wheatear.

4.1 Habitat Associations at the Site Level

There were some errors in habitat recording on several sites, which reduced the sample size for habitat associations to 33 and 31 sites respectively for the first and second years. Species richness at the site level was significantly lower in sites where statues where present in 2002 and was significantly correlated with mown grass cover in 2003. There were no other significant associations at the site level. Furthermore, in both cases, significance levels were weak ($P > 0.025$ in both cases).

For individual species density, significant associations with habitat at the site level for those species where model fit was deemed acceptable ($0.2 > D > 10.0$ – see Chapter 1) are shown in Table 2.2. There were few species where significant associations were found. In 2002, Robin, Wren and Blackcap all showed significantly higher density where pavement was present at the boundary. Robin density was also lower where railway was present in the surroundings. Coal Tit density was significantly lower where flower beds were present. In 2003, there was a significantly higher density of Woodpigeon, Starling and House Sparrow where buildings were present on the site (Table 2.2).

There were several species where model fit was underdispersed due to a large number of zeros and most counts only being single individuals where a species was detected. Even those species in Table 2.2 were only just within the range of acceptable model fit. Analyses were repeated using binomial logistic regression in order to see if there were significant associations with habitat and probability of species occurrence. Generally however, model fit was not improved and in many cases, models failed to converge. Those species showing significant effects of habitat variables and where $0.2 < D < 10$ are shown in Table 2.3. There were few consistent effects.

In Chapter 1, species richness and the density of several individual species were significantly greater if bushes were present on a site. Those analyses were repeated for birds showing evidence of breeding. There was no significant difference in species richness between sites with or without bushes in either 2002 ($F_{2,28} = 0.89$, NS) or 2003 ($F_{2,17} = 0.29$, NS). For individual species density, significant differences are shown in Table 2.4. There were seven species that showed a significantly higher density in sites where bushes were present (Blackcap, Dunnock, Great Tit, Magpie, Woodpigeon, Wren and Blue Tit). Robin also showed a higher density in sites with bushes in 2002, but showed a higher density in sites without bushes in 2003. Blackbird and Coal Tit also showed significantly higher density where bushes were absent. The analysis was repeated using binomial logistic regression to compare probability of occurrence. Results were similar to those in Table 2.4, with similar species showing similar associations. Exceptions included House Sparrow and Song Thrush, where probability of occurrence was significantly higher where bushes were present and Pied Wagtail where probability of occurrence was lower in the presence of bushes.

4.2 Habitat Associations at the Patch Level

At the patch level, there were few habitats where birds showing evidence of breeding were recorded. The data set was reduced by amalgamating categories which occurred on <5 sites into broader categories or by omitting categories where birds were never recorded. In 2002 this left only six habitat categories: deciduous trees (DT), mown grass (MG), mixed trees (MT), rough grass/nettles (RG), sports/hard surface (SP) and walls and buildings (WN). However, there were only three habitat types that met the variable selection criteria in this year: deciduous trees, rough grass and mown grass.

There was no significant difference in species richness between these habitat types. In 2002, the difference was almost significant ($\chi^2_6 = 12.31$, $P < 0.056$), where mixed trees and rough grass had the highest estimates and mown grass the lowest.

There were also few significant differences in individual species density between habitats. Only Blue Tit showed significant differences between habitats in 2002 (Fig. 2.1). In 2003, Wren and Robin had their highest densities in deciduous trees (compared to mown grass and rough grass). Blackbird densities were highest in mown grass. No other species showed significant effects.

As previously, count data were converted to presence/absence data and binomial models were fitted to the data. Several species showed significant differences in probability of occurrence between habitat types. Occurrence rates in different habitats in 2002 are shown in Fig. 2.2. Woodpigeon, Blue Tit, Great Tit and Blackbird were most likely to occur in mixed trees and least likely in walls/buildings or sports/hard surfaces. Magpie and Robin were both most likely to occur in rough grass/nettles. More species showed significant effects in 2003 when there was only a three-way comparison. In this year, deciduous trees had higher occurrence rates than mown grass or rough grass/nettles in Blackcap, Blue Tit, Chiffchaff, Chaffinch, Dunnock, Greenfinch, Great Tit, Green Woodpecker, Jay, Long-tailed Tit, Robin and Wren. Mown grass had highest occurrence rates for Collared Dove, Mistle Thrush, Magpie and House Sparrow. Rough grass/nettles had highest occurrence rates for Whitethroat. Woodpigeon and Blackbird had almost equal occurrence rates in deciduous trees and mown grass.

5. DISCUSSION

There were 58 species that showed some evidence of breeding in the core survey over two years. This compares with 64 species recorded in total whether showing breeding evidence or not and with 88 species recorded in either summer of the main survey (Chapter 1). Therefore, a high proportion of birds detected within London's green spaces are likely to be potential breeders. For individual species, there were a number where the occurrence rates actually exceeded that in the main survey (compare Table 2.1 with Appendix I). This was the case in at least one year in Blue Tit, Great Tit, Coal Tit, Robin, Wren, Blackcap, Whitethroat and Chiffchaff. This may represent a higher detection rate due to the more experienced observers used in the core survey. Alternatively, the sites used in the core survey may be better in some way for these species. In most other cases, the occurrence rates in the main survey exceeded those in the core survey, but not usually by large percentages. A notable exception was Feral Pigeon, which occurred in a high proportion of sites, but was suspected breeding in only 11% and 19% of core sites in 2002 and 2003 respectively. Hirundines and Swift also had a wide discrepancy between occurrence of all birds and likely breeders but these species are largely restricted to nesting in buildings and are likely to use green spaces only for foraging. However, the survey methods are not really appropriate for such highly mobile species.

At the site level, there were few species that showed significant associations between habitat and breeding numbers. In 2002, Robin showed a lower density in sites with nearby railway line (possibly indicating fewer breeders in sites within more urbanized areas) and Coal Tit showed a lower density in sites with flowerbeds. The variable which had consistent effects across species was pavement: Wren, Robin and Blackcap all had significantly higher breeding density where pavement was present. In 2003, only one variable, building presence, was significant. Woodpigeon, Starling and House Sparrow were at greater density where buildings were present. These results are similar to those in Chapter 1, where building presence was a consistent predictor of species richness and the density of several individual species. In Chapter 1, it was suggested that this association arose as sites with buildings had more key habitats (deciduous trees, deciduous bushes and mown grass). The associations with pavements and buildings could also represent correlated effects. However, it should also be noted that both Starling and House Sparrow commonly nest in buildings so these associations could represent nesting habitat availability in these species.

At the patch level, there were relatively few habitats used, especially in 2003 where the comparison was only between three habitat types. There was no significant difference in breeding density between different habitat types for most species. There was greater variation (and usually better model fit) for occurrence rates. Generally, mixed and deciduous trees had high occurrence rates for many species. This is in accord with habitat associations of bird density reported in Chapter 1. However, there was evidence for a few species that rough grass/nettles may be important habitat. This was the case for Robin (potential nesting habitat) and Magpie. There were also high occurrence rates on mown grass in several species. These were generally larger species that prefer to forage on the ground. Note however that these were not ground nesters. Breeding evidence in these cases would have been behavioural cues (foraging for young, collecting nesting material, territorial dispute, mating) rather than a detection of the nesting habitat.

The above example illustrates a problem when carrying out such surveys in that foraging habitat and also habitats that are used for song or territorial display are likely to be well covered as detectability will be high. However, it is difficult to draw conclusions on the value of different habitat patches for nesting in the majority of species. A survey that could identify the best nesting habitats would take far more effort than was possible in this survey. Therefore, making management recommendations (additional to those given in Chapter 1) is difficult. Buildings are likely to provide nesting habitat to Starlings and House Sparrows, especially where there are plenty of cavities. Rough grass/nettles was a relatively more important habitat compared to results from Chapter 1 and could potentially provide nesting habitat for Robin and possibly other species. Otherwise, the management recommendations suggested in Chapter 1 (especially provision of deciduous trees, mown grass and deciduous bushes) are likely to provide at least good foraging opportunities for a range of breeding species.

	2002		2003	
Species	**Breeding**	**All**	**Breeding**	**All**
Blue Tit	0.98	0.98	0.84	0.86
Blackbird	0.93	0.93	0.92	0.95
Wren	0.91	0.91	0.95	0.95
Robin	0.89	0.93	0.81	0.89
Great Tit	0.80	0.82	0.68	0.73
Carrion Crow	0.77	0.80	0.62	0.86
Woodpigeon	0.77	0.93	0.70	0.92
Blackcap	0.61	0.64	0.65	0.65
Magpie	0.59	0.75	0.68	0.76
Starling	0.55	0.82	0.49	0.70
Dunnock	0.52	0.52	0.49	0.49
Greenfinch	0.48	0.50	0.24	0.32
Chaffinch	0.43	0.45	0.46	0.49
Chiffchaff	0.41	0.41	0.43	0.43
Song Thrush	0.39	0.43	0.38	0.41
Green Woodpecker	0.36	0.36	0.32	0.32
Jay	0.34	0.48	0.22	0.24
Long-tailed Tit	0.30	0.34	0.38	0.43
Collared Dove	0.27	0.32	0.08	0.14
Great Spotted Woodpecker	0.27	0.34	0.24	0.30
House Sparrow	0.27	0.41	0.38	0.43
Mistle Thrush	0.25	0.30	0.51	0.59
Goldcrest	0.23	0.30	0.22	0.27
Coal Tit	0.20	0.20	0.19	0.19
Goldfinch	0.14	0.18	0.14	0.19
Feral Pigeon	0.11	0.64	0.19	0.59
Stock Dove	0.11	0.14	0.22	0.22
Willow Tit	0.11	0.11	0.00	0.00
Jackdaw	0.09	0.14	0.14	0.14
Ring-necked Parakeet	0.09	0.14	0.11	0.16
Willow Warbler	0.09	0.09	0.16	0.16
Kestrel	0.07	0.11	0.05	0.05
Pied Wagtail	0.07	0.11	0.14	0.30
Sparrowhawk	0.07	0.09	0.14	0.14
Linnet	0.05	0.09	0.08	0.08
Nuthatch	0.05	0.07	0.14	0.14
Treecreeper	0.05	0.11	0.08	0.11
Bullfinch	0.02	0.05	0.00	0.00
Grey Heron	0.02	0.02	0.03	0.03
Kingfisher	0.02	0.05	0.03	0.03
Lesser Redpoll	0.02	0.02	0.00	0.00
Lesser Spotted Woodpecker	0.02	0.05	0.05	0.05
Lesser Whitethroat	0.02	0.02	0.08	0.08
Canada Goose	0.00	0.00	0.03	0.05
Coot	0.00	0.00	0.11	0.14

Table 2.1 The proportion of sites in which each species occurred in the breeding season for species showing evidence of breeding and all species recorded. N = 44 in 2002 and 37 in 2003.

	2002		2003	
Species	**Breeding**	**All**	**Breeding**	**All**
Great Crested Grebe	0.00	0.00	0.03	0.03
Goshawk	0.00	0.00	0.03	0.03
Grey Wagtail	0.00	0.02	0.00	0.00
Garden Warbler	0.00	0.00	0.08	0.08
Little Grebe	0.00	0.00	0.05	0.05
Moorhen	0.00	0.00	0.11	0.14
Meadow Pipit	0.00	0.00	0.05	0.05
Mute Swan	0.00	0.00	0.08	0.08
Pheasant	0.00	0.02	0.05	0.05
Pochard	0.00	0.00	0.00	0.03
Reed Bunting	0.00	0.00	0.03	0.03
Rook	0.00	0.05	0.00	0.03
Skylark	0.00	0.00	0.05	0.08
Swallow	0.00	0.00	0.03	0.05
Turtle Dove	0.00	0.00	0.00	0.03
Tufted Duck	0.00	0.00	0.11	0.11
Wheatear	0.00	0.00	0.00	0.03
Whitethroat	0.00	0.00	0.16	0.16
Woodlark	0.00	0.00	0.00	0.03

Table 2.1 Continued.

Year	Species		*D*
2002	Wren	PAVE+	0.28
	Robin	ARAIL- PAVE+	0.28
	Blackcap	PAVE+	0.24
	Coal Tit	FF-	0.32
2003	Woodpigeon	BLDG++	0.24
	Starling	BLDG++	0.22
	House Sparrow	BLDG++	0.26

Table 2.2 Associations between habitat variables and species density for birds showing evidence of breeding at the site level.

Year	Species		*D*
2002	Mistle Thrush	AGARD-- FF- SP+	0.45
	Blackcap	RG++	1.31
	Carrion Crow	CT- WN+	0.93
	House Sparrow	CT-- DB+ WN--	0.97
	Greenfinch	RG+ WN-	2.81
2003	Dunnock	AGARD- BLDG+++	1.14
	Great Tit	BLDG++	0.95

Table 2.3 Associations between habitat variables and species probability of occurrence for birds showing evidence of breeding at the site level.

Species	Year	Bush present			Bush absent			*D*
		Mean	LCL	UCL	Mean	LCL	UCL	
Blackbird	2002	0.88	0.38	1.38	0.90	0.37	1.42	0.65
Blackcap		0.53	0.14	0.91	0.31	-0.10	0.71	0.39
Coal Tit		0.17	-0.18	0.51	0.50	0.14	0.86	0.30
Dunnock		0.37	0.08	0.66	0.17	-0.13	0.48	0.22
Great Tit		0.49	0.16	0.83	0.33	-0.02	0.69	0.29
Magpie		0.56	0.26	0.86	0.27	-0.05	0.58	0.24
Robin		0.93	0.45	1.41	0.60	0.10	1.11	0.60
Woodpigeon		0.61	0.22	1.00	0.21	-0.20	0.62	0.39
Wren		1.00	0.54	1.46	0.77	0.28	1.25	0.55
Blackbird	2003	0.45	0.16	0.74	0.60	0.25	0.95	0.22
Blue Tit		0.60	0.27	0.94	0.38	-0.02	0.78	0.29
Great Tit		0.38	0.06	0.70	0.31	-0.07	0.69	0.26
Robin		0.56	0.26	0.87	0.63	0.26	0.99	0.25
Woodpigeon		0.49	0.14	0.83	0.12	-0.29	0.54	0.31
Wren		0.70	0.35	1.05	0.68	0.27	1.10	0.31

Table 2.4 A comparison of density of birds showing evidence of breeding in sites with deciduous trees, mown grass and deciduous bushes present versus sites with deciduous trees, and mown grass present and deciduous bushes absent.

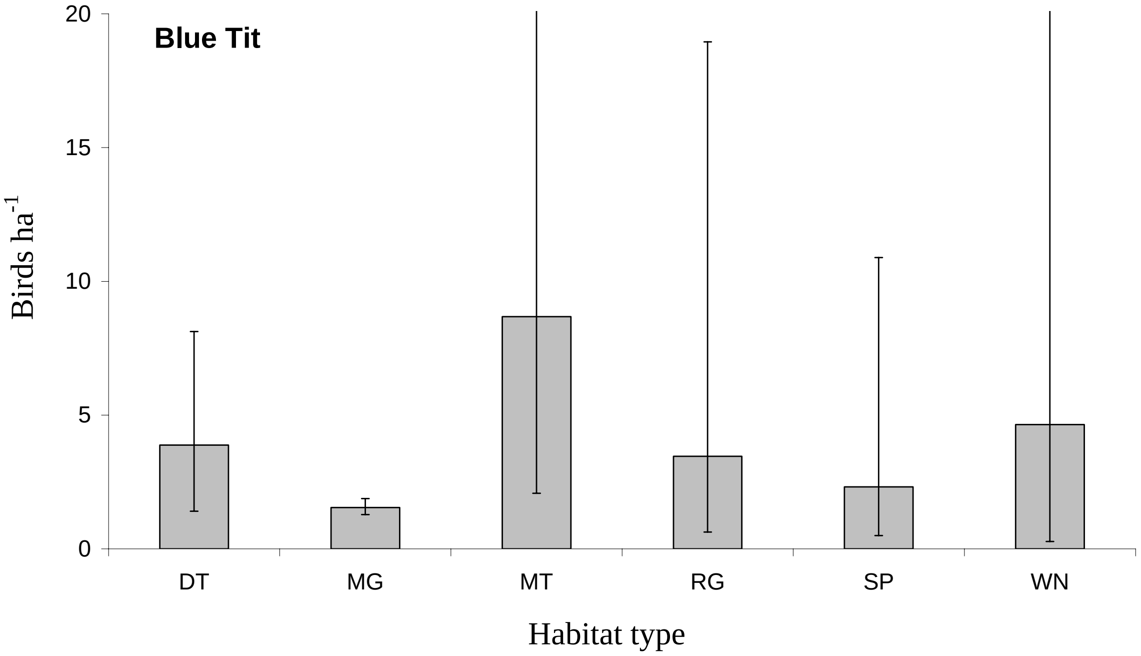


Figure 2.1 Mean (± 95% CL) Blue Tit density in different habitat types. Birds that showed evidence of breeding only are included. Note that in two habitats, MT and WN, upper confidence limits are off the y-axis scale. Respective values for these were 27.5 and 72.4.

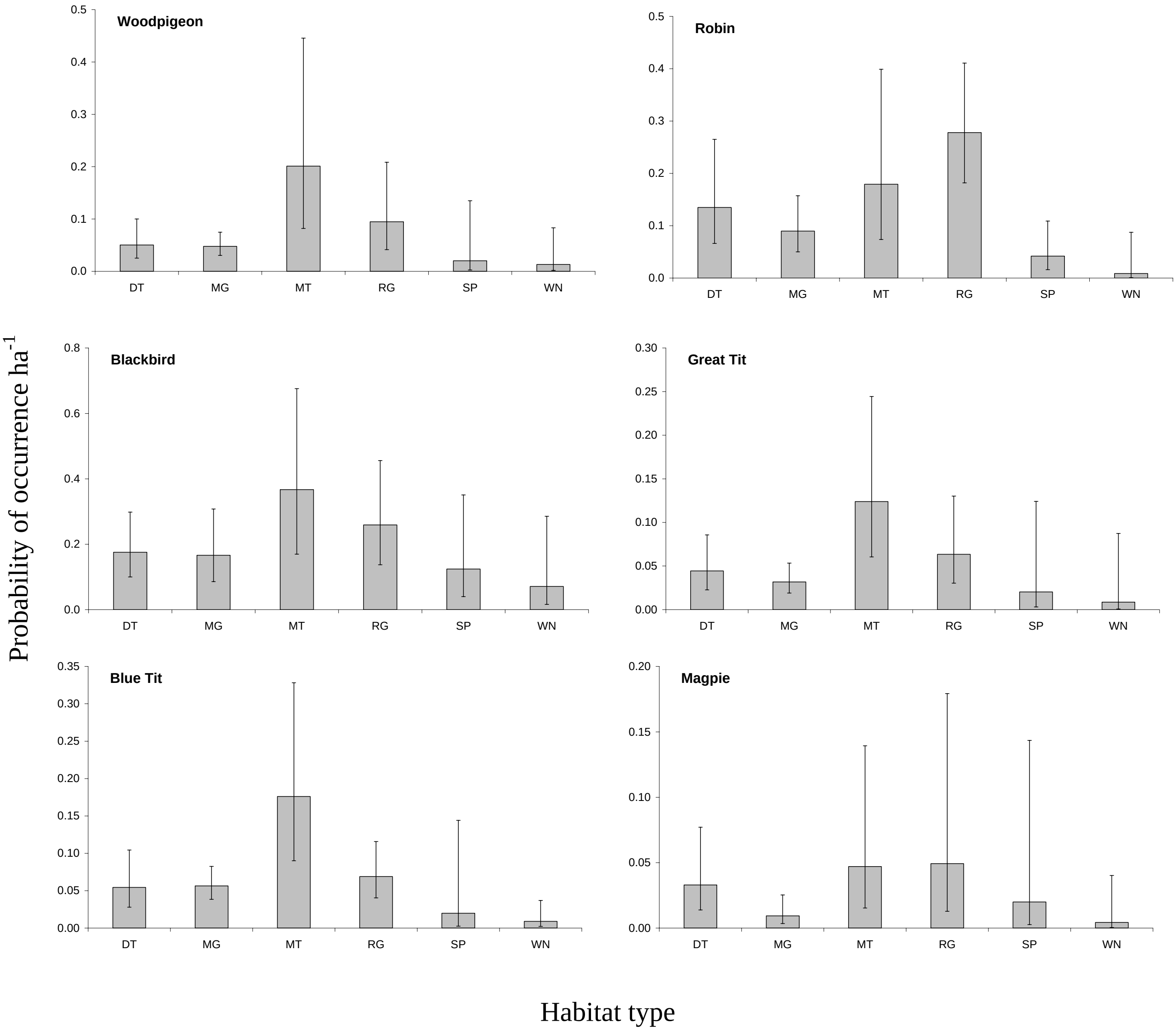


Figure 2.2 Probability of occurrence (± 95% CL) in different habitat types of species showing evidence of breeding in 2002.

CHAPTER 3 BROAD-SCALE HABITAT ASSOCIATIONS OF BIRDS IN GREATER LONDON: AN ANALYSIS USING BREEDING BIRD SURVEY DATA

Dan Chamberlain, Steve Freeman, David Noble

1. INTRODUCTION

The broad-scale distribution of almost all bird species in London is fairly well known, thanks largely to the London Atlas (Hewlett 2002). There are also a number of Breeding Bird Survey (BBS) squares. The BBS is the main annual monitoring scheme for relatively common terrestrial birds in the UK (Raven *et al*. 2003). The BBS produces both national and regional indices of bird population change on an annual basis that are used to inform and influence conservation policy at the highest level. For example, the BBS is the main data contributor to the Wildbird Indicator, one of the government's 15 headline 'Quality of Life' indicators (Anon 1999). There is a sufficient sample of BBS squares in Greater London to enable trends to be estimated for 16 species (Raven *et al*. 2003). The most recent trends published suggest that London's birds are going through a period of major change. There were significant increases between 1994 and 2003 in Woodpigeon, Collared Dove, Wren, Robin, Blue Tit, Great Tit, Magpie, Carrion Crow and Greenfinch. However, there were significant decreases in Blackbird, Song Thrush, Starling and House Sparrow (Raven *et al*. 2003).

Whilst we know a reasonable amount about the distribution and population change of species in Greater London, we know less about birds' habitat associations within this area. Analysis of BBS data in an urban setting has estimated that a large proportion of Woodpigeon, Dunnock, Blackbird, Song Thrush, Starling, Jackdaw, House Sparrow and Greenfinch are associated with human habitats (Gregory & Baillie 1998, Newson *et al*. in press). For Blackbird, Starling and House Sparrow the highest densities are found in suburban and urban habitat (Newson *et al.* in press) There has been more work from overseas on urban bird-habitat associations where urban bird distributions and communities are of increasing interest due to concern over urban encroachment into important wildlife habitats (e.g. Blair 1996, Germaine *et al*. 1998, Kluza *et al*. 2000, Cam *et al*. 2000, Fernández-Juricic & Jokimäki 2001, Odell & Knight 2001). The effects of fragmentation of forest habitats, particularly on neo-tropical migrants, and consequent changes in predation pressure have often proved the main focus for such research (Friesen *et al*. 1995, Gering & Blair 1999). Research on the effects of vegetation type and structure on birds in urban environments has also been undertaken. For example, Gavareski (1976) and Mills *et al*. (1989) showed that bird abundance is typically associated with native plant species in the USA. Clergeau *et al.* (2001) examined urban bird data from France, Finland and Canada and found that, in winter at least, the composition of urban bird communities was not influenced by the bird community of surrounding 'periurban' landscapes. This implied that local features were more important than surrounding landscapes.

2. AIMS

In this chapter, we aim to describe the wider bird community of Greater London (i.e. covering a range of habitats), to assess bird-habitat associations and also to consider bird population trends over time in relation to habitat. Analysis of BBS data enabled an assessment of general habitat gradients, including habitats not covered in the green spaces survey (Chapters 1 and 2), at relatively large scales.

3. METHODS

BBS is based on a random sample of 1-km squares stratified by regional population size (so there is a greater density of squares in Greater London than in northern Scotland, for example). For each square, two parallel 1-km long transects are identified. Each transect is divided into five 200m long sections. All birds detected in each section are recorded and their location marked into distance bands at 25m, 100m and over 100m from the transect (birds in flight are also recorded). The habitat in each section is also recorded using standard habitat codes (Crick 1992). Two survey visits are carried out per year, the first in April-mid May and the second from mid May to the end of June. There were between 35 and 64 1-km squares covered in the BBS between 1994 and 2002 and a total of 83 different squares were covered. For this analysis, only registrations within 100m from the transect line are analysed and it is assumed that detectability does not vary significantly between gross habitat types.

4. ANALYSIS

The extent of any variation in spatial and temporal trends between London sites comprising different habitats was considered between 1994 and 2002. Each BBS 200-m transect section was assigned to one of 17 broad habitat categories on the basis of habitat information provided by site surveyors (so there was a maximum of 10 different habitat types per season, one for each transect section). Trends in abundance and probability of occurrence were analysed in relation to 16 habitat types in Table 3.1 (excluding habitats identified as 'OTHH'). Abundance was determined for each transect in each square and a mean calculated across years. Only data from the first two distance bands were used. Detectability is likely to decrease the further away a given bird is away from the transect. If accurate estimates of density are required, the detectability function can be modelled using the program Distance (Buckland *et al*. 1994). This is however time consuming and is only applicable to the more abundant species. In this chapter relative differences between habitats and years are analysed rather than using more accurate estimates of density. In using a simpler measure of abundance an assumption of equal detectability between different habitats within the sample is being made. Restricting the data to a relatively narrow distance band of 100m is likely to make this assumption more robust.

A Generalized Linear Model was applied to the data with normal errors, relating mean abundance to habitat type. Transect sections from the same square were not considered as independent as there was likely to be an element of spatial autocorrelation and also each square had the same observer. The autocorrelation of counts within each square was adjusted using General Estimating Equations (GEE) within the GENMOD procedure in SAS (SAS Institute 1998) using a repeated measures model framework. Species richness was analysed using the same approach, where the dependent variable was the number of species recorded per year in each transect section. Model fit was described by the dispersion statistic D (see Chapter 1).

The analysis of probability of occurrence was approached in a similar manner, but assuming a binomial distribution and a logit link function in the model. The number of times a given species was present per transect section per square and the number of times that transect section had been visited over all years were used as respective events and trials in the model. No offset was used in this case. Both normal and binomial models were run separately for early and late visits.

There were some cases where the habitat changed for a given transect between years. In these cases, the habitat that appeared for the greatest number of years is included in the model and the other data discarded (including cases where there were an equal number of years).

Temporal trends - Temporal trends in overall abundance and habitat-specific trends between 1994 (when the survey was established) and 2001[1] were considered. Generalized Linear Models were applied to data from the 13 most widespread species, relating abundance to two multi-level factors; "site" (allowing for any geographical variation) and "year" (the corresponding parameter estimates being employed as annual indices of abundance in the standard manner (Raven *et al*. 2003)).

[1] Note that the temporal analysis was carried out in 2003 before the 2002 BBS data set had been input. Other analyses include 2002 data.

5. RESULTS

5.1 Basic Statistics

There were 83 BBS squares covered between 1994 and 2002 (range 34-62 per year). In total there were 103 species (and 2 hybrids) recorded during BBS in the period 1994-2002 in Greater London (Appendix III). Many of these species were recorded on only a few occasions. For example, 34 species were recorded on fewer than five occasions and 48 species were recorded in fewer than 10% of squares. The most frequent individual species were Blackbird, Starling and House Sparrow which occurred in more than 90% of transect sections. The most frequent species recorded also tended to be the most numerous. There were a number of BAP species recorded: Turtle Dove, Skylark, Song Thrush, Tree Sparrow, Bullfinch, Linnet, Reed Bunting and Corn Bunting. However, in most cases these were recorded at low abundance and occurrence, with the notable exception of Song Thrush which occurred in over 50% of squares and had a mean abundance of almost 2 individuals per square.

The distribution of the 83 BBS squares and an indication of mean species richness over the period 1994-2002 is shown in Fig. 3.1. Spatial patterns were indistinct, but there was some tendency for the squares with the highest species richness to be towards the north and west.

Fig. 3.2 shows the total percentage occurrence of each habitat type over all years for the early period (which had slightly higher coverage) based on all transects where habitat information was recorded (n = 4021). There were a number of cases where habitat was either not recorded or recorded incorrectly (n = 537 transects). The most frequent habitats were suburban buildings (SBIL), suburban gardens (SGDN), urban buildings (UBIL) and suburban parks (SPRK). Note also that many habitat types could not be assigned to one of the main 16 listed (OTHH), perhaps illustrating the drawbacks of using a general habitat coding system (Crick 1992) in an urban environment.

5.2 Habitat Associations

Species richness - The mean numbers of species recorded over all years and all squares in early and late visits in 16 habitat types (excluding ‘OTHH’) are shown in Fig. 3.3. The greatest mean number of individual species recorded was in rural garden habitat (RGDN) and rivers (RIVE), but both of these had low sample size. Relatively high species richness (>5 in both periods) was also found in lakes (LAKE), rural parks (RPRK), suburban gardens (SGDN) and parks (SPRK), urban parks (UPRK) and woods. The species richness was lowest in urban buildings (UBIL) and semi-natural grassland (SEMI). There was no evidence of strong seasonal trends. Further analyses will use maximum count of individual species over the two visits which is a standard procedure for estimating breeding numbers from BBS data (Raven *et al*. 2003).

Mean species richness per transect across all years was analysed in relation to the eight most widespread habitat types (Fig. 3.3) using a repeated measures model framework with normal errors. The model was fitted with an intercept term and the reference habitat was set as WOOD. A total of 80 squares and 787 transect sections were included in the analysis. The DSCALE option was also used to correct for overdispersion which was evident (D = 9.27). There was a significant effect of habitat (χ^2_7 = 22.03, P < 0.0025), where both urban and suburban buildings had lower richness than the reference habitat WOOD.

Species richness was also analysed in relation to remotely sensed landcover data derived from CEH’s CS2000 (Fuller *et al*. 2002) data base at the whole square level. Five continuous variables were analysed: arable cover, grass cover, woodland cover, suburban cover and urban cover. The quadratic term of each variable was also entered into the model initially. Model structure was the same as the previous analysis. As the landcover data were collected in 2000, the BBS data are restricted to the period 1998-2002. There were no significant quadratic effects. There were weakly significant (P < 0.04) negative effects of both arable cover and urban cover on species richness implying that both central areas and peripheral areas of Greater London tended to have lower than average species richness. No other variables were significantly associated with species richness.

Individual species occurrence - Binomial logistic regression models did not converge on a solution in several species. These were typically those species that were either very common, occurring in virtually all squares (e.g. Feral Pigeon, Blackbird, Blue Tit, Great Tit) or very scarce species occurring in very few squares (e.g. Grey Heron, Kestrel, Lapwing, Linnet). There were only eight species that showed significant differences in probability of occurrence between habitats (Fig. 3.4). Collared Dove showed highest occurrence rates in gardens. For Wren, Robin and Song Thrush rural gardens and parks and woods had the highest occurrence rates and farmland and urban buildings the lowest. Magpie and Goldfinch had relatively high estimates in rural habitats but they also had relatively high occurrence rates in farmland. Finally, both Starling and House Sparrow had the highest occurrence rates in suburban and urban habitats. Model fits for these species were generally good, being between 0.70 (Collared Dove) and 2.39 (House Sparrow).

Individual species abundance - There were several species that showed significant differences in density between habitat types. However, in several cases model fit was very poor where species were very abundant and often highly aggregated (e.g. $D > 10$ for Starling, Feral Pigeon, House Sparrow, Greenfinch) or very scarce (e.g. $D < 0.20$ for Green Woodpecker, Stock Dove, Whitethroat, Jay). Species showing a significant effect of habitat and where $0.20 > D > 10.0$ are shown in Fig. 3.5. A striking feature of Fig. 3.5 is that for six species the habitat with the highest density was rural gardens. These were Woodpigeon, Robin, Blackbird, Song Thrush, Great Tit, Magpie and Greenfinch. Moorhen reached its highest density in river habitat (although even here, rural gardens clearly had some high counts as shown by the large error bars for this habitat). Carrion Crow showed a greater affinity with parkland and farmland.

Temporal trends - Non-significant variability between years was found for only four of the species considered (Mallard, Collared Dove, House Martin, Magpie); the remaining nine (Canada Goose, Feral Pigeon, Wood Pigeon, Carrion Crow, Blue Tit, Blackbird, Starling, House Sparrow and Greenfinch) all fluctuated significantly from one year to the next (Fig. 6). There were increasing trends in Canada Goose, Wood Pigeon, Carrion Crow, Blue Tit and Greenfinch. Decreasing trends were apparent in Feral Pigeon, Blackbird (neither decreased strongly), Starling and House Sparrow. For a comparison of trends in London with those in the surrounding areas see Newson and Noble (2003).

There was a significant difference in bird abundance between habitats in each case. The GLM was extended to include an additional component, the interaction between this habitat categorisation and the year. A test of the significance of this interaction amounts to a test of the equality of temporal trends across all habitats. A significant result means that there is a difference between at least some of the habitat categories. Such significant results (significant at the 1% level in each case) were found for eight species: Starling, House Sparrow, Feral Pigeon, Wood Pigeon, Blackbird, Carrion Crow, Blue Tit and Magpie. Trends are shown in Fig. 3.7 for the more widespread habitat types. Feral Pigeon was virtually unrecorded in rural habitats (arable, grass and woodland), this species being a true urban specialist. There were clear increases in suburban parks and suburban buildings, but patterns were stable or fluctuating elsewhere. Woodpigeon was also increasing in several habitats, the increase being most clear in suburban buildings. For Blackbird, Blue Tit, Carrion Crow and Magpie, patterns were typically fluctuating, although the fluctuations were not consistent across habitats from year to year. Starling showed a slight decline in several habitats but a markedly large decline in woodland. Finally, House Sparrow showed general declines in all habitats, but declines were most severe in arable habitat and urban parks.

6. DISCUSSION

Species richness and the abundance and occurrence of several individual species were highest in suburban habitats illustrating the value of suburban gardens and parks compared to urban and rural habitats (farmland, grassland and woodland). Heavily built environments held the lowest richness and individual species abundance/ occurrence in most cases. The clear and obvious message arising from this work is that increasing urbanization (in the sense of replacement of green space with buildings) will have a general detrimental effect on the bird community. There were however, two exceptions, Starling and House Sparrow, where urban and suburban habitats tended to be better than rural habitats. A further more surprising message however, is that there were apparently negative effects of farmland on species richness. Suburban gardens and parks may therefore be better for a richer bird community than farmland which it often replaces under new housing developments. However, there are caveats on this. First, although species number may be lower in farmland than gardens, the individual species tend to be of higher conservation concern (e.g. more BAP species on farmland). Second, typical farmland is currently a poor habitat for many bird species (Siriwardena *et al*. 1998). Changes in agri-environmental policy could change patterns of bird communities across an urban-rural gradient.

Generally, BBS trends for Greater London reflect trends in the wider countryside (Raven *et al*. 2003). However, it was clear that for some species annual trends differed between habitat types. Increasing species often showed the greatest gains in urbanized or built-up habitats (e.g. Feral Pigeon, Woodpigeon, Blue Tit). This may be due to improved conditions in these habitats over time (e.g. lowering pollution levels) or may merely be the result of increases in 'source' habitats that are being picked up in less favoured 'sink' habitats. Habitat-specific declines may prove to be very important for declining species. For example, House Sparrow showed declines in parkland and farmland. Local declines have been reported previously in London (Hewlett 2002) and wider populations in arable landscapes are also declining (Siriwardena *et al*. 2002). Furthermore, Starling in woodland and Blackbird in urban parks both showed clear steady declines relative to other habitats. Conservation efforts may therefore need to be focused on specific habitat types for particular species.

For several species, there appear to be significant differences between the trajectories of population change in London and those in other parts of the country, although evidence of similar trends are seen in adjacent regions. Although much attention has been given to the House Sparrow, there are major declines for Starling, Song Thrush and Blackbird. The decline in the Starling is in fact less than in other regions but those for the two thrush species are the highest recorded nationally. Against this backdrop, it is important to try to involve more people in the annual BBS and pleasing to note that several volunteers from the London Bird Project have offered to take on this challenge. Currently, the BBS for London is only sufficiently robust to monitor sixteen species, but this should change as the number of squares covered increases.

Habitat	Code
Arable	ARAB
Grass (agricultural)	GRAS
Grass (semi-natural)	SEMI
Lake/pond	LAKE
Mixed farmland	MIXF
Other farmland	OFRM
Rural garden	RGDN
River/canal	RIVE
Rural park	RPRK
Suburban buildings	SBIL
Scrub	SCRU
Suburban garden	SGDN
Suburban park	SPRK
Urban buildings	UBIL
Urban park	UPRK
Woodland	WOOD
Unclassified	OTHH

Table 3.1 Habitat types used in BBS analysis combining data from 1994-2002. A number of transects were unclassified, either because there was no habitat information given for a particular BBS square, or because the classification was too vague (e.g. a number were classified as 'suburban, near road', 'urban, near railway line' etc.).

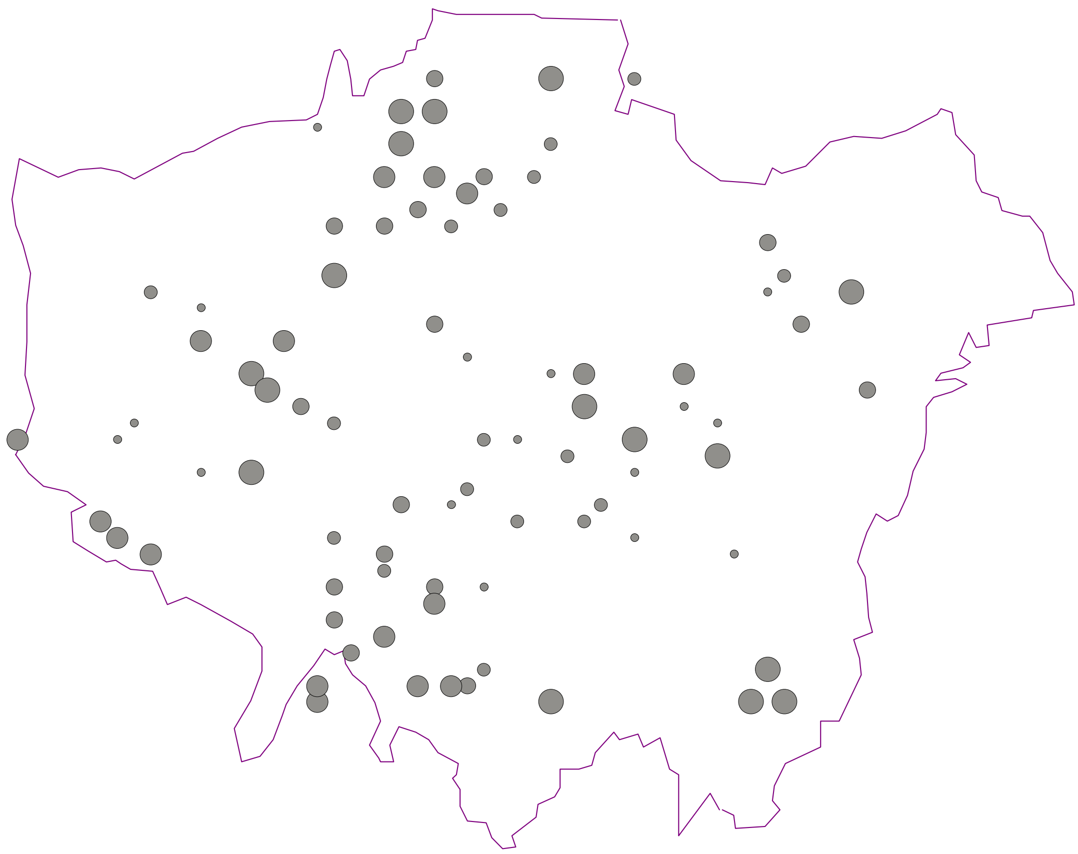


Figure 3.1 Mean species richness of BBS squares in Greater London between 1994 and 2002. Size 1 = 5-12 spp, 2 = 13-15 spp, 3 = 16-19 spp, 4 = 20-23 spp, 5 = 24-34 spp.

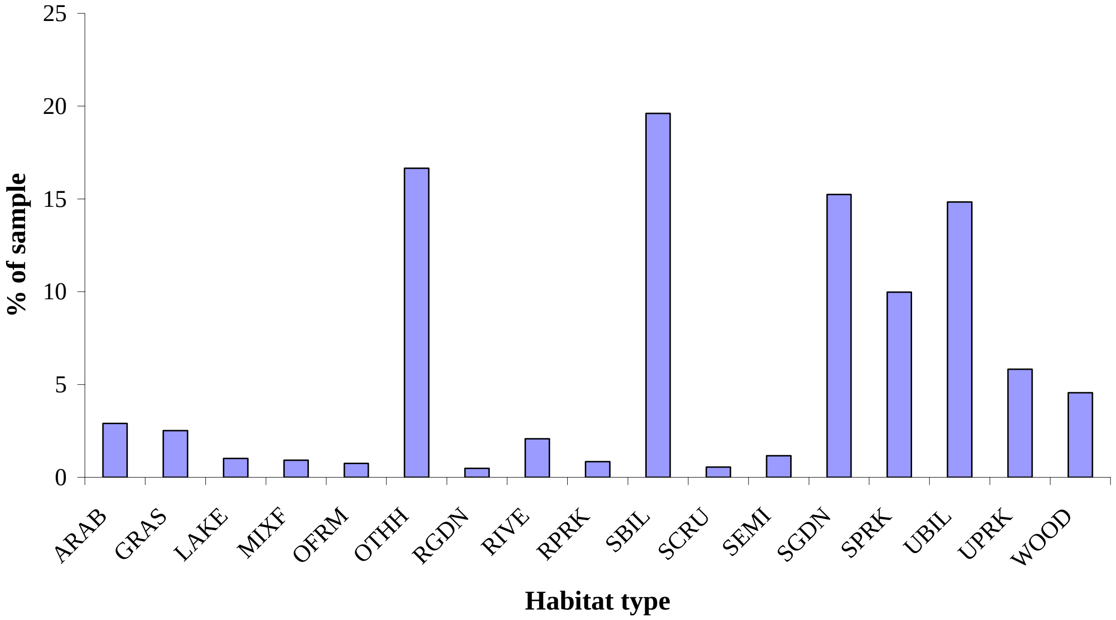


Figure 3.2 Percentage occurrence of habitat types, 1994-2002.

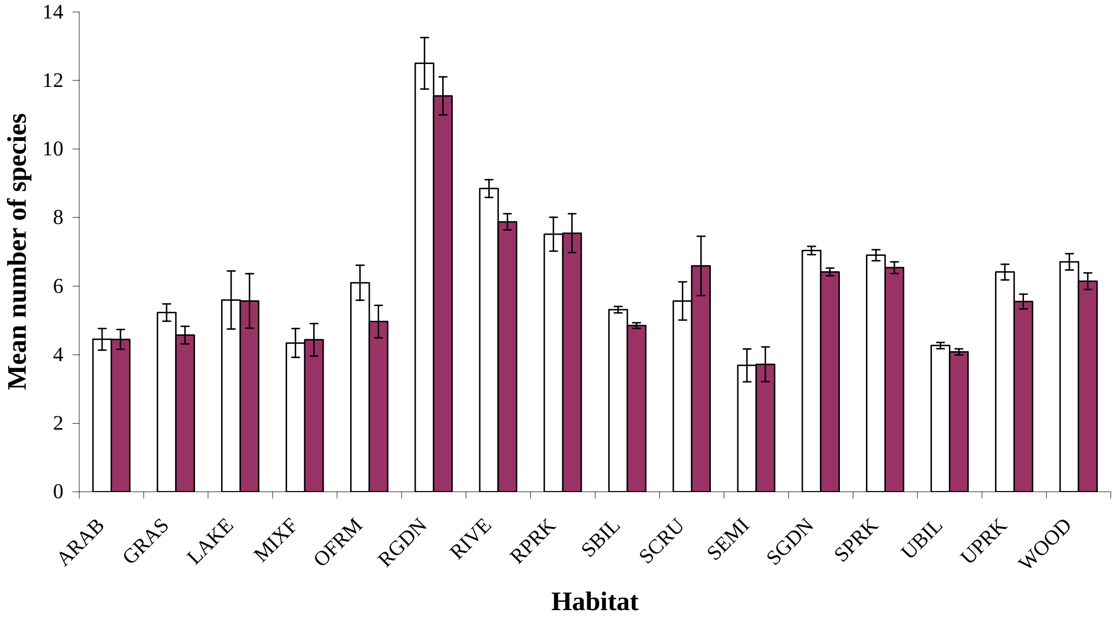


Figure 3.3 Mean (± SE) number of species per 1-km^2 in different habitat types 1994-2002. Sample size (no. transects) as follows (early,late): ARAB = 120,126; GRAS = 104-108; LAKE = 42,46; MIXF = 38,39; OFRM = 31,31; RGDN = 20,20; RIVE = 86,81; RPRK = 35,35; SBIL = 808,793; SCRU = 23-22; SEMI = 48-50; SGDN = 628,604; SPRK = 411,404; UBIL = 611,565; UPRK = 240,231; WOOD = 188,187.

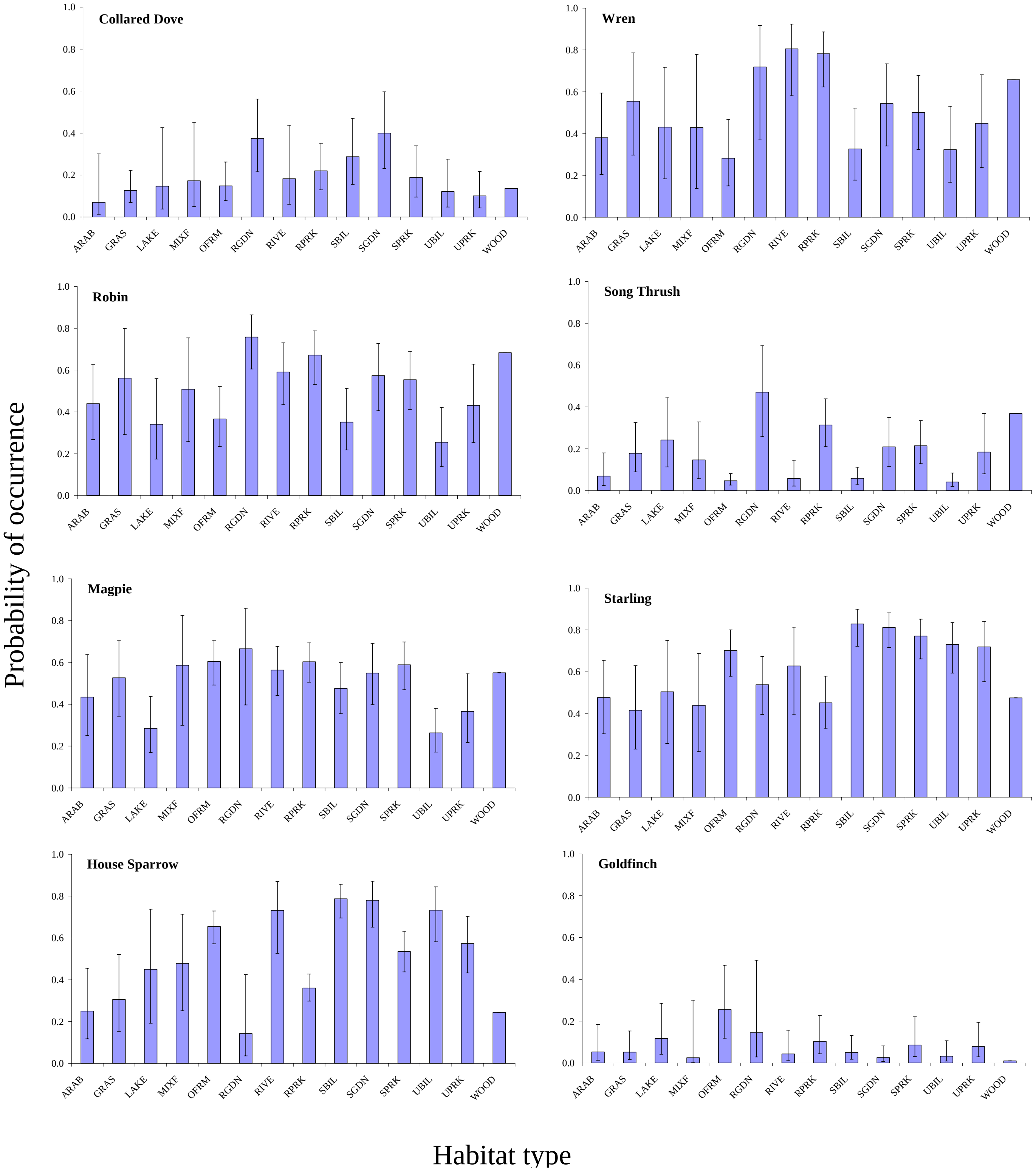


Figure 3.4 Estimated probability of occurrence (± SE) in different habitat types.

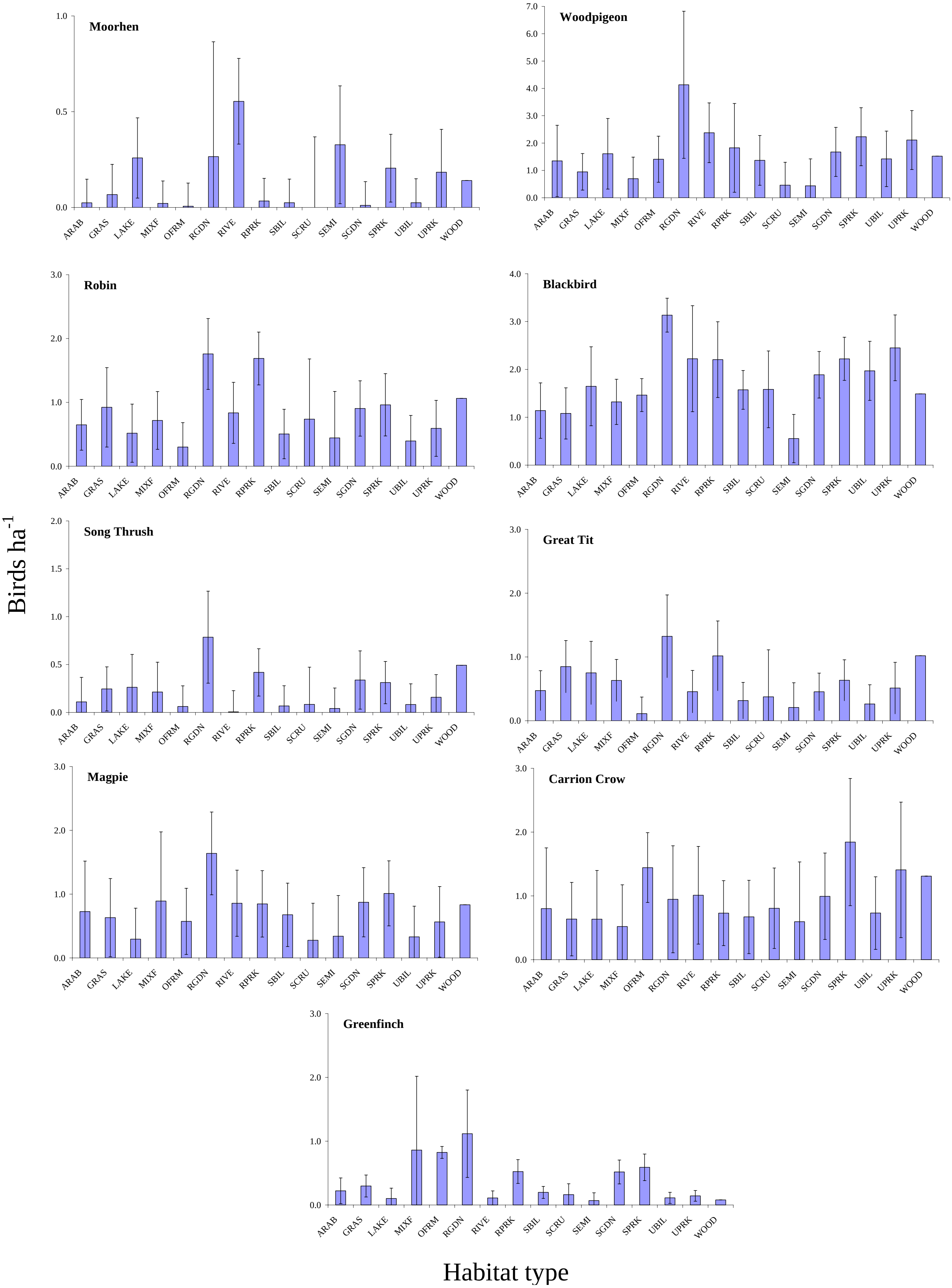


Figure 3.5 Estimated density (± SE) (birds/ha) of species in different habitats within Greater London.

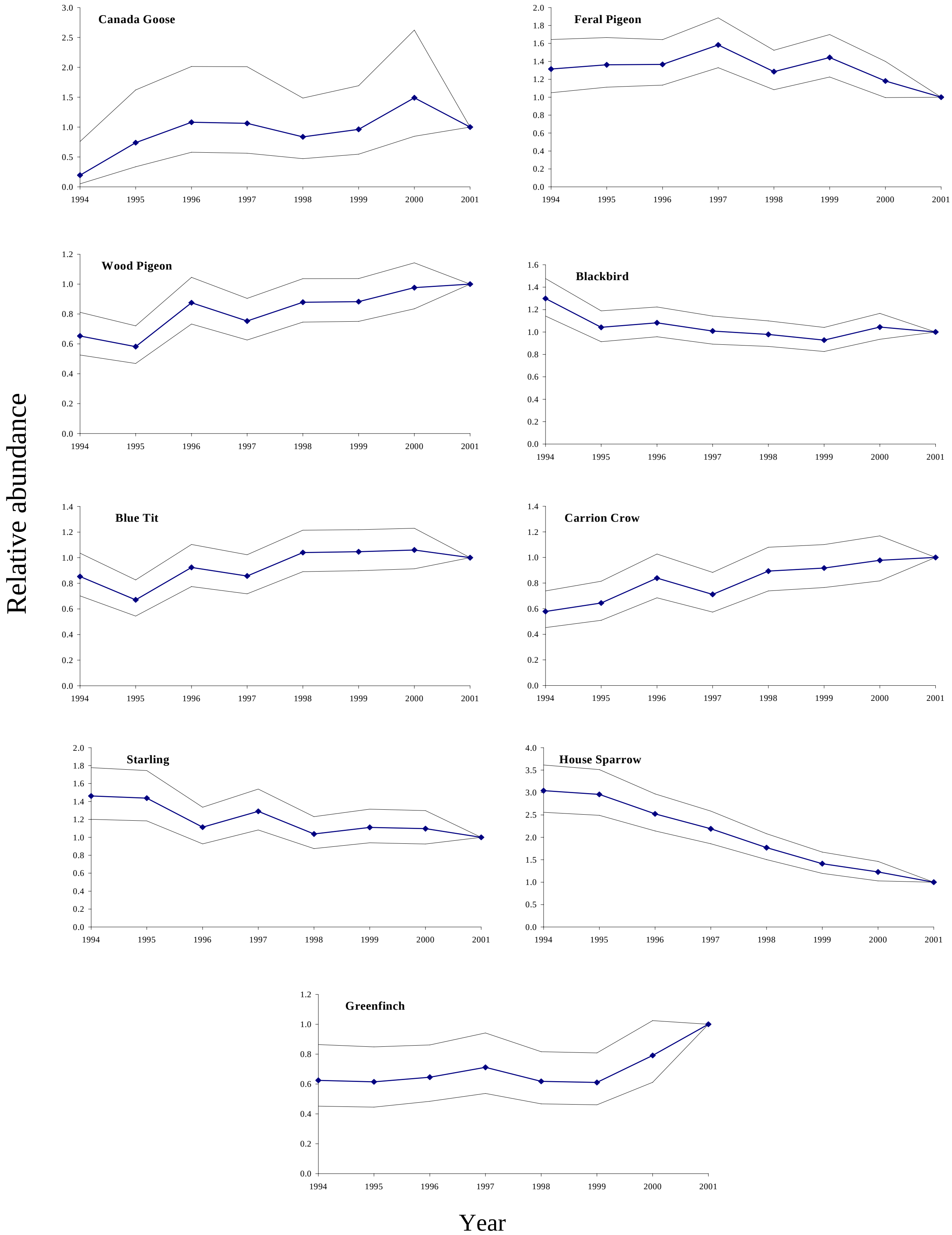


Figure 3.6 Annual variation in relative abundance (± 95% CL) of 8 bird species in Greater London. Relative abundance is set at 1 in year 2001 in each case.

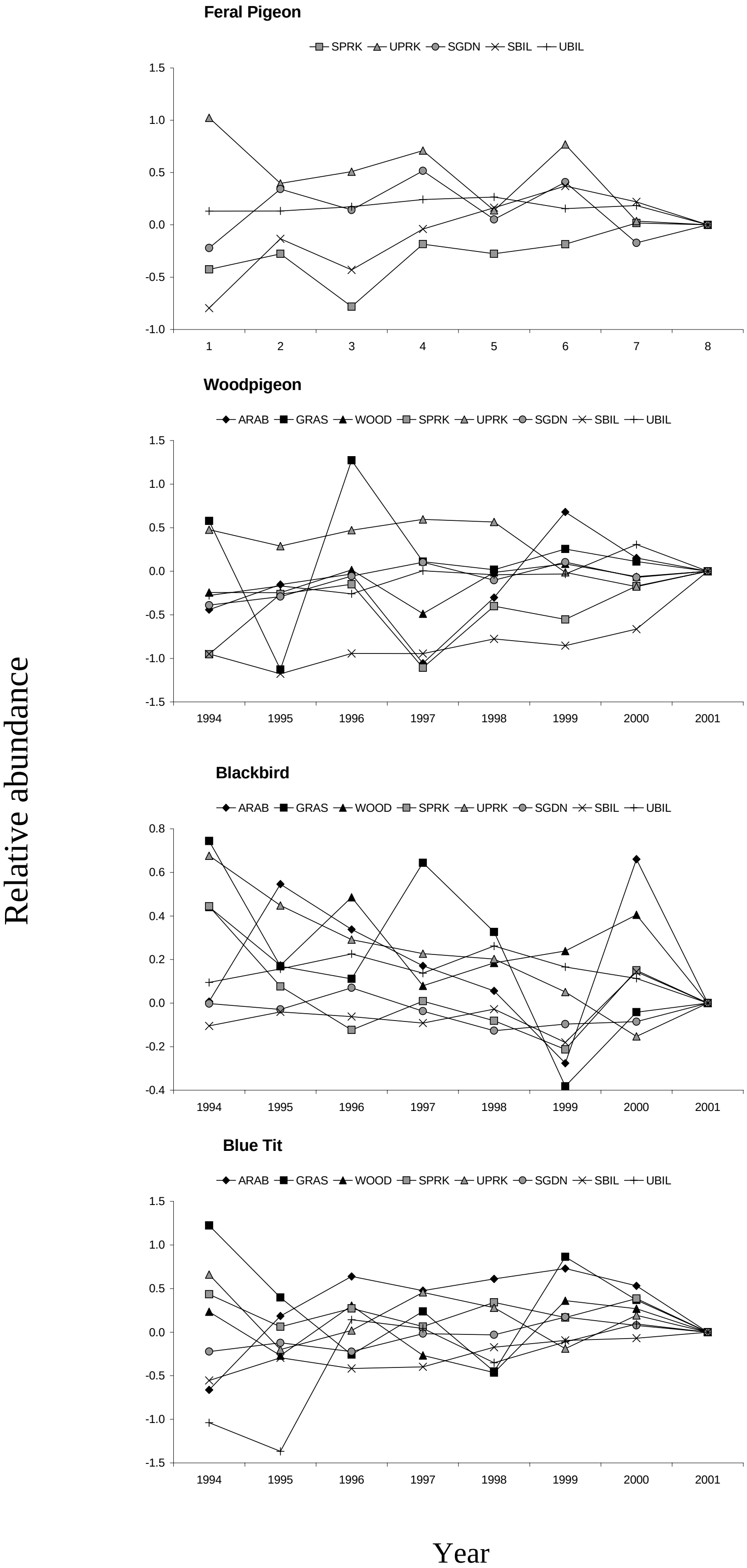


Figure 3.7 Relative change in abundance of birds recorded in BBS squares within Greater London between 1994 and 2001 in different habitat types. Only those habitats that were recorded in at least 10 squares in at least 1 year (Table 3.1) are shown. In each case, interaction between habitat and year were significant. Each habitat is presented relative to 2001 which is set at 0. (Note that House Sparrow was not recorded in woodland in 2000 – this point is not included on the figure).

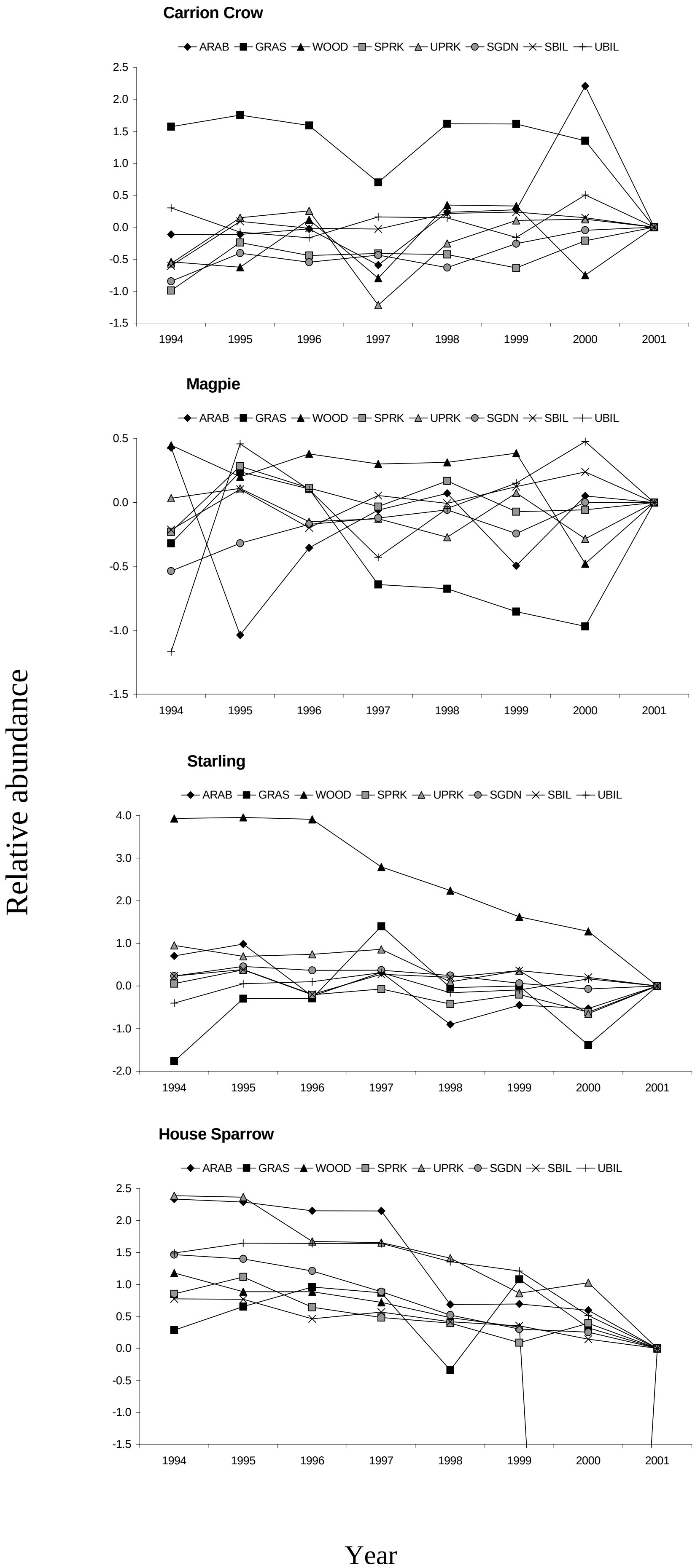


Figure 3.7 Continued.

CHAPTER 4 POPULATION TRENDS AND HABITAT ASSOCIATIONS OF GARDEN BIRDS IN GREATER LONDON

Dan Chamberlain, Steve Freeman, Mike Toms

1. INTRODUCTION

Private gardens are major contributors to urban biodiversity in the UK and are important habitats for birds (Cannon 1999) and other taxonomic groups (Good 2000, Ansell *et al*. 2001). Private gardens are estimated to cover c. 3% of the land area of the UK (Owen 1991), an area that exceeds the total area of nature reserves. Despite this, the ecology of birds in gardens is relatively unstudied in comparison with other major habitat types, perhaps mainly due to problems of access (Cannon 1999), yet it is clear in the Song Thrush at least that private gardens hold a significant proportion of the national population (Mason 2000). Densities of many other species are also high in urban and suburban habitats which is likely to reflect, in part, the quality of habitat provided by private gardens (Gregory & Baillie 1998). In Chapter 3, analysis of BBS data for London showed that the greatest species richness and also some of the highest densities of individual species were in gardens.

In terms of population monitoring, the only long-term study carried out in the UK has been the Garden Bird Feeding Survey (GBFS; Chamberlain *et al*. in press) which specifically monitors use of garden bird feeding stations in the winter. In several cases, trends in the use of supplementary food provided in gardens do not match population changes in the wider countryside, possibly due to changes in quality and type of food provided in gardens over time or due to an increased reliance on bird feeders as habitat quality in the wider countryside (especially farmland) deteriorates. A more recent and more comprehensive continuous monitoring scheme is the BTO/CJ Wildbird Foods Garden BirdWatch (GBW; Cannon 1998, Toms 2003). GBW has used volunteers to survey a wide sample of gardens throughout the UK since 1995. Part of the goal of GBW is to involve the public in scientific survey work. The methods used are therefore as simple as possible. Nevertheless, GBW data provide a unique opportunity to examine habitat associations of common garden birds. GBW is one of the largest volunteer monitoring surveys in the world (Cannon 1999) and as such represents a huge resource.

GBW data have been analysed to determine inter-annual changes in bird use of gardens (Cannon *et al.* in press) and to assess associations between bird occurrence and broad-scale habitat gradients (Chamberlain *et al*. 2004). Applying similar analyses at a local level will give a general picture of population trends and habitat associations specific to Greater London.

2. AIMS

The aims of this chapter are: (i) to describe occurrence rates of common garden species in Greater London gardens; (ii) to assess fluctuations in garden bird occurrence between years (i.e. long-term trends) and within years (i.e. seasonal patterns) in Greater London; (iii) to identify habitats that are closely correlated with garden bird occurrence in Greater London.

3. METHODS

Each site is subject to one observation period each week. The observation period is left to the discretion of the observer, but should be constant from week to week, i.e. survey effort varies between sites but not over time within sites. Furthermore, observers are requested to carry out observation periods at the same time every week and from the same vantage point. During each observation period, any species from a list of 41 common garden birds are recorded (Cannon 1998). In this section, only presence/absence data of the 41 commonest species are considered (the development of a novel statistical technique to analyse additional categorical abundance data collected in GBW is given in Chapter 5). In addition, a large number of habitat variables are recorded describing the site and the surrounding landscape. These are recorded as categorical or ranked variables rather than continuous variables (Table 4.1).

3.1 Statistical Analysis

Annual trends – Trends in bird occurrence in Greater London gardens were analysed using binomial logistic regression. As both intra- and inter-annual patterns were of interest, data for every individual week were used. Data from the same sites in successive weeks were not independent so a repeated measures model framework was applied to the logistic regression model. Furthermore, many bird species exhibit cyclical patterns in their occurrence over time. In order to adequately model these patterns, week terms were expressed as sine and cosine functions (Flury & Levri 1999). These methods follow the analysis used by Cannon *et al*. (in press) and full details can be found therein.

Habitat associations - Year was divided into two seasons ('winter' & 'summer', where summer covers weeks 13-31 and winter covers weeks 1-9 and weeks 43-52). For each species, separately in each season and in each year 1995-2002, the number of weeks in which the species was present in each garden was calculated as a proportion of the total number of weeks in which a survey was undertaken. Thus for an example species:

C_{ijk} = the proportion of weeks in which the species was present in garden *i*, in season *j* (winter or summer) of year *k*

is defined. Suppose then that a particular garden has an associated size vector z_i and habitat covariate vector h_i, where z_i is a 1x3 vector $\{s_i, m_i, l_i\}$ and $s_i = 1$ for a small garden (and 0 otherwise), and m_i and l_i are quantified similarly. The dimension of the habitat vector *h* will vary according to the number of levels into which the particular habitat is apportioned, but functions in a similar manner, with the level occupied by garden *i* identified by the digit '1' at the appropriate vector element, and all others by zero. Then

$$\text{logit}\ (C_{ij}) = S_j + \alpha' z_i + \beta' h_i$$

where S*j* is an effect representing the season (*j*=winter, summer), and the model is fitted separately each year *k* = 1995-2002. The significance of the habitat effect is recorded in each case, for each of 41 species regularly recorded in London gardens. With 41 species, 7 years and 46 types of habitat this clearly involves an enormous number of tests. This makes it impossible to interpret individual significant results with any confidence (Beal & Khamis 1991), and some account must be taken of this multiple testing. Because of the practical and philosophical difficulties involved (see e.g. Moran 2003) critical levels of significance are not reduced here. Rather, interpretable patterns are sought in the accumulated array of test statistics. In all cases model fit was described by the dispersion statistic *D* (see Chapter 1).

Species richness was also analysed in a similar way, but the number of species recorded per site per year per season was determined and analysed with Poisson regression. Note, however, that this is not true species richness as only 41 species are recorded in GBW (although recent developments in web-based data entry have included all species which will facilitate analysis of full species richness in the future).

In addition to considering separate habitat variables, composite variables were also derived from ordination techniques using the programme CANOCO (Ter Braak & Smilauer 2002). Methods were identical to those used on GBW data at a national scale by Chamberlain *et al*. (2004) where variables were reduced to a binomial (either present/absent or high/low) before analysis. If ecologically meaningful gradients were identified, these were analysed with respect to bird occurrence and species richness as above.

4. RESULTS

There were 953 GBW sites in the Greater London area available for analysis between 1995 and 2002. A summary of the occurrence rates of the 41 target species between 1995 and 2002 within the Greater London area is shown in Table 4.2 for summer (April-July) and winter (October-February) separately. Blackbird was the most common species, occurring in c. 90% of gardens in both summer and winter. Other species occurring in more than 70% of gardens were Blue Tit, Woodpigeon, Robin, Great Tit, House Sparrow and Starling. Occurrence rates were slightly higher in the winter for most species. Generally though, seasonal differences were not great, apart from winter migrants (Brambling, Fieldfare and Redwing). The largest differences were Coal Tit, Chaffinch, Black-headed Gull, Pied Wagtail, Blackcap and Siskin though for the latter four species, occurrence rates were generally very low all round. Furthermore, Black-headed Gull, Blackcap and Siskin populations are augmented by migrants in the winter. No species showed markedly higher occurrence rates in the summer.

The distribution of GBW sites and the species richness per 1-km in summer and winter is shown in Fig. 4.1. Note that this is any 1-km square that had an active GBW site (so more than one site could be in each square). The total number of squares was 307 and the total number of sites was 785 between 1995 and 2002 (there were 168 sites where grid reference data were missing or erroneous). There was some suggestion that species richness was greater towards the north and south of the Greater London area with fewer large squares across the centre. There were also fewer sites in this area however.

4.1 Annual Trends

There were only nine species that showed significant variation in probability of occurrence between years. These were Feral Pigeon, Wren, Robin, Long-tailed Tit, Starling, House Sparrow, Greenfinch, Goldfinch and Siskin. There were six species where there was a significant interaction between week and year terms, indicating the seasonal patterns shifted over time. These species were Blackbird, Song Thrush, Coal Tit, Long-tailed Tit, House Sparrow and Goldfinch. Fitted annual trends and mean probability of occurrence for these species are shown in Fig. 4.2.

For certain species, the model clearly did not adequately describe the trends in the raw data. In particular, Siskin had very poor model fit but this was the only species that was under-dispersed (D = 0.15). For other species, model fit tended to be better, although there were some instances, notably in Greenfinch and Coal Tit where there were seasons with much lower than average occurrence rates (these were usually in the first year when sample sizes were relatively small). There were significant increases in Feral Pigeon, Wren, Long-tailed Tit, Goldfinch and to a lesser extent, Robin. There were declines in Song Thrush, Coal Tit, Starling and House Sparrow. Starling, Greenfinch and more subtley, Blackbird, showed a decrease in seasonal amplitude of trends over time.

4.2 Effects of Habitat on Bird Occurrence and Species Richness

There were a little over 100 species/habitat pairings in which a relationship between probability of occurrence and habitat was significant (at the conventional 5% level) in each of the seven years. Bear in mind here that, if habitat and bird presence were unrelated, the probability of a specified species being significantly associated with a specific habitat every year is $0.05^7 = 7.8 \times 10^{-10}$, thus even from 41 x 46 species/habitat pairings the expected number to produce seven significant results is considerably less than one. Clearly, therefore, there is a marked effect of habitat upon the presence of birds in London gardens**.** What is perhaps of particular interest is that, between them, the ten species that are sufficiently common for it to be worthwhile recording numbers of individual birds in GBW (see Chapter 5) comprise only ten of the cases in which a relationship with a particular habitat type was significant every year. This is, in relative terms, a considerably smaller proportion than the number of similar relationships from the remaining species. This may suggest that the commoner species are less restricted by habitat and more generalist in their requirements, as befits their status as the commonest species. Habitat may be a more critical factor for the rarer species, some of which such as Treecreeper and Tawny Owl have very specific requirements.

However, although many significant associations were detected, the effects of different levels of habitat were often year-specific, e.g. a given species could show an increasing linear probability with an increase in a given habitat in one year and a curvilinear relationship in another year. There were relatively few cases where associations were consistent from year to year (in terms of the relative magnitude of the parameter estimates of different levels of habitat extent). Associations between species occurrence and habitat that were consistent across years are shown in Table 4.3. It is clear from this table that coniferous trees and hedges are important predictors of bird occurrence in London gardens for many species. Sparrowhawk, Wren, Blackcap, Goldcrest, Long-tailed Tit, Chaffinch and Goldfinch all showed increasing probability of occurrence with a greater number of large coniferous trees and a greater cover of high hedges. Certain species showed non-linear trends in relation to vegetation cover, with a peak in probability of occurrence at intermediate levels of either number of trees (Goldcrest,) or hedge cover (Collared Dove, Blackbird, Greenfinch). There were two species, Rook and Jackdaw, that showed negative effects of small coniferous tree numbers indicating a preference for open garden. Alternatively, the association may be linked to an urban-rural gradient as we may expect small coniferous trees to be representative of *Leylandii* and other exotic species that are most common in suburban landscapes, whereas these latter two species are particularly linked to open farmland. The presence of woodland and farmland within 100m of the garden was also significantly associated with several species. Great Spotted Woodpecker, Nuthatch and Bullfinch were more likely to occur in gardens near broadleaved woodland. Rook and Starling were less likely in such gardens. Adjacent farmland significantly increased the probability of Great Spotted Woodpecker, Long-tailed Tit and Nuthatch. These are not typical farmland species, so this association probably represents a greater occurrence in gardens in more rural landscapes (and so on the urban fringe). Feral Pigeon, very much an urban specialist, was less likely to occur where farmland was adjacent to a garden. There were no consistent effects of the proximity of parks to probability of occurrence in gardens for any species.

A more detailed analysis of the ten cases involving the commoner species is given in Table 4.4. The results of these analyses are remarkably consistent: in both winter and summer the habitat effects were found to be significant, and none of the interaction terms representing a change in this effect between years or between winter and summer were significant.

There were consistent associations between species richness and land use (rural and suburban > urban), lawn cover (positive), garden size (large and medium > small) and large coniferous trees (positive). In common with Table 4.3, these variables were analysed in combined-year models to assess consistency across years, seasons and garden size (Table 4.5). In contrast to the individual species results (Table 4.4) there were significant interactions: the association between species richness and landuse varied according to season and year and the associations between species richness and lawn cover varied according to year, although in these cases, results were only weakly ($P > 0.02$) significant (Table 4.5). When analysed by season, the only consistently significant effects were landuse and garden size.

4.3 Ordination of Habitat Data

Initially, Detrended Correspondence Analysis (DCA) was used on the data. This revealed a length of gradient of 2.1 indicating that the data were distributed linearly rather than unimodally and therefore Principle Components Analysis (PCA) was more appropriate (Jongman *et al.* 1995). The first two axes derived from PCA explained 12.0% and 9.8% of variation in the data respectively. A bi-plot of these axes is shown in Fig. 4.3. Note that each variable must be present in every site so sample sizes were reduced to 416 (i.e. over 50% of sites were not included due to some missing data). Presence of trees and berries, relative berry abundance to surrounding gardens and size were strongly negatively related to the first axis. The only variable that was strongly positively associated with the axis was presence of parks within 100m. This variable is of ecological interest as it appears to largely represent a gradient of tree and berry cover in gardens which are likely to be important to birds. For the second axis, there was one strongly negatively correlated variable, presence of fence. Positively correlated variables included mixed and broad-leaved woodland, scrub, semi-natural grass and stream presence within 100m of the garden. This gradient appears to represent gardens in more rural surroundings adjacent to more semi-natural habitat to gardens in more built up areas (possibly these are more likely to have fences as garden boundaries). Lower axes were less readily interpretable and will not be considered further.

Site scores from the first two axes were used as continuous variables in binomial logistic regression. Year and season were treated separately. As previously (Table 4.3), consistent associations across years were sought. Axis1 has no consistent significant effects in any species. Axis 2 was significant in at least 7 out of 8 years for Robin, Long-tailed Tit, Chaffinch and Bullfinch in winter and Chaffinch in summer (all positive effects) indicating greater probability of occurrence in sites in close proximity to semi-natural habitats such as mixed and broad-leafed woodland, scrub and grassland.

5. DISCUSSION

The distribution of GBW sites is not random, but shows a bias towards more southerly and northerly parts of Greater London. There is also likely to be a bias in the type of housing in the GBW sites. Rural and suburban gardens are over-represented in relation to true urban dwellings (Cannon *et al.* in prep.). Therefore the analysis presented here is probably representative of suburban bird communities rather than truly urban bird communities.

There was a tendency for species to occur at higher rates in the winter. The reasons for this may be twofold. First, species may be more detectable in the winter, particularly when reliant on bird food. Given that 95% of GBW sites provide food (Chamberlain *et al*. 2004), this seems to be a likely scenario. Second, resident populations may be augmented by migrant or dispersing individuals. Three species, Fieldfare, Redwing and Brambling, only regularly occur in the winter but there are several other species (e.g. Robin, Goldfinch, Chaffinch) whose winter populations are likely to be increased by partial migrants from the continent (Wernham *et al*. 2002). Furthermore, there is some evidence that certain granivorous species, particularly those that are declining, move into gardens from farmland in the winter (Chamberlain *et al*. in press). Higher winter densities were also evident in several species in urban green spaces (Chapter 1).

There were some species that showed clear temporal trends in Greater London gardens over the relatively short time span of 9 years. Table 4.6 compares trends between BBS and GBW data for 10 species (those that were GBW species and that had BBS trends determined in Chapter 3), where GBW data was restricted to the summer only (late March-late July). Starling and House Sparrow declined in both surveys. Blackbird declined in BBS but showed no significant annual trend in GBW. Collared Dove and Magpie showed no significant difference in either survey. Woodpigeon, Blue Tit, Carrion Crow and Greenfinch had increasing trends in BBS but no significant trends in GBW. Feral Pigeon showed significant increases in GBW but significant decreases in BBS. Differences in trend could be due to differing survey methods, as BBS considers abundance whereas GBW considers only presence. It may be expected that probability of occurrence is less variable than abundance data hence fewer significant trends are found for GBW. It is harder to explain the patterns for Feral Pigeon where trends are opposing in the two surveys. Increases in gardens relative to the general population could indicate that this particular habitat is good relative to the general urban habitat. It is even possible that an increase in gardens is reflective of a declining wider population or a decreasing quality of non-garden habitats as birds become more dependent on food provided by man (Chamberlain *et al*. in press). Note that the GBW trends here are based on presence/absence data. A more formal statistical comparison, using abundance class data from GBW, is presented in Chapter 5.

The habitat associations found were complicated and varied from year to year for most species. There were relatively few cases where habitats had consistent effects on species occurrence in all years considered. For many individual species and species richness the number of coniferous trees and the cover of hedges in the garden boundary were, not surprisingly, consistent determinants of species presence. Similar associations with bushes (ecologically more or less equivalent to garden hedges) were found in urban green spaces (Chapter 1). However, deciduous trees held the highest densities in urban green spaces but mature coniferous trees were a key determinant of species presence in gardens for several species. There may be genuine ecological reasons for this difference, but it is possible that co-related factors are affecting these results (see below).

More species were found in larger gardens and gardens with a greater cover of lawn, although the latter is probably merely reflective of the former association. The surrounding landscape is also an important aspect, with many species being more closely associated with gardens that were in more rural settings (close to arable farmland, grassland and woodland). Species richness was also greater in rural and suburban compared to urban landscapes. There were other intriguing associations, notably that the presence of several species was associated with vegetables in gardens. Further investigation is required at a finer resolution of habitat recording than is possible in GBW in order to assess whether these are genuine associations or whether vegetable presence is a surrogate for another unmeasured variable, possibly an urban-rural gradient.

A problem in interpreting these associations is that there is a high degree of colinearity in the data. For example, gardens with large coniferous trees are more likely in less urbanized gardens in close proximity to rural habitats. These gardens also tend to be larger. Teasing apart the effects of garden habitat and the surrounding habitat is therefore difficult. Attempts to reduce the large number of habitat variables to fewer surrogate variables using ordination were only partially successful. Ecologically meaningful axes were derived, but they explained only a relatively low amount of variation in the data indicating generally high variability. There was an indication that sites in close proximity to semi-natural habitats (axis 2) were more likely to have Robin, Long-tailed Tit, Chaffinch and Bullfinch, although the low variability explained by this axis should be born in mind when interpreting these associations. Nevertheless, species showing such associations would probably be negatively affected by increasing urban encroachment and infilling of currently existing green spaces.

Habitat	Code	Definition	Ranking
Age	AGE	Years old	1 (0-4) 2 (5-10) 3 (11-19) 4 (20-49) 5 (50+)
Altitude	ALT	Altitude above sea level	1 (0-50m) 2 (51-100m) 3 (101-250m) 4 (251-499m) 5 (500+m)
Arable farmland	ARAB	Occurrence within 100 yards	0 or 1
Barren	BARR	Occurrence within 100 yards	0 or 1
Berries	BERR	Berry-bearing plants present	0 or 1
Berry abundance	LBER	Berry abundance relative to surroundings	1 (fewer), 2 (same) or 3 (more) than in neighbouring land
Bog	BOG	Occurrence within 100 yards	0 or 1
Buildings	BUIL	% in garden boundary	1-5 (0% and quartiles)
Canal	CANA	Occurrence within 100 yards	0 or 1
Coastal proximity	COAS	Distance to coast	1-4 (0-5, 6-10, 11-50, 51+ km)
Coniferous hedge (high)	CHHE	% in garden boundary	1-5 (0% and quartiles)
Coniferous hedge (low)	CLHE	% in garden boundary	1-5 (0% and quartiles)
Coniferous trees (large)	CTTR	Number in garden	1 (0) 2 (1) 3 (2-4) 4 (5-9) 5 (10+)
Coniferous trees (small)	CSTR	Number in garden	1 (0) 2 (1) 3 (2-4) 4 (5-9) 5 (10+)
Coniferous wood	CWOD	Occurrence within 100 yards	0 or 1
Deciduous hedge (high)	DHHE	% in garden boundary	1-5 (0% and quartiles)
Deciduous hedge (low)	DLHE	% in garden boundary	1-5 (0% and quartiles)
Deciduous trees (large)	DLTR	Number in garden	1 (0) 2 (1) 3 (2-4) 4 (5-9) 5 (10+)
Deciduous trees (small)	DSTR	Number in garden	1 (0) 2 (1) 3 (2-4) 4 (5-9) 5 (10+)
Deciduous woodland	DWOD	Occurrence within 100 yards	0 or 1
Fence	FENC	% in garden boundary	1-5 (0% and quartiles)
Flower beds	FLOW	% of garden covered	1-5 (0% and quartiles)
Gardens	GARD	Occurrence within 100 yards	0 or 1
Land use	LAND	Surrounding landscape	1 (urban) 2 (suburban) 3 (rural)
Lawn	LAWN	% of garden covered	1-5 (0% and quartiles)
Marsh	MARS	Occurrence within 100 yards	0 or 1
Miscellaneous farmland	FARM	Occurrence within 100 yards	0 or 1
Mixed woodland	MWOD	Occurrence within 100 yards	0 or 1
Orchard	ORCH	% of garden covered	1-5 (0% and quartiles)
Other	MISC	% in garden boundary	1-5 (0% and quartiles)
Parks	PARK	Occurrence within 100 yards	0 or 1
Pastoral farmland	PAST	Occurrence within 100 yards	0 or 1
Railway	RAIL	Occurrence within 100 yards	0 or 1
Refuse	REFU	Occurrence within 100 yards	0 or 1

Table 4.1 Habitat variables recorded in GBW.

Habitat	Code	Definition	Ranking
River	RIVE	Occurrence within 100 yards	0 or 1
Scrub	SCRU	Occurrence within 100 yards	0 or 1
Semi-natural grassland	GRAS	Occurrence within 100 yards	0 or 1
Shrubs	SHRU	% of garden covered	1-5 (0% and quartiles)
Size	SIZE	Garden area	1 (small) 2 (medium) 3 (large)
Stream	STRE	Occurrence within 100 yards	0 or 1
Vegetables	VEGE	% of garden covered	1-5 (0% and quartiles)
Wall	WALL	% in garden boundary	1-5 (0% and quartiles)
Water body (large)	WATL	Occurrence within 100 yards	0 or 1
Water body (small)	WATS	Occurrence within 100 yards	0 or 1
'Wild' land	WILD	Occurrence within 100 yards	0 or 1

Table 4.1 Continued.

Species	Summer	Winter
Blackbird	0.911	0.895
Blue Tit	0.884	0.939
Woodpigeon	0.799	0.708
Robin	0.734	0.861
Great Tit	0.722	0.785
House Sparrow	0.722	0.642
Starling	0.703	0.668
Magpie	0.637	0.644
Collared Dove	0.519	0.487
Feral Pigeon	0.457	0.427
Dunnock	0.449	0.505
Greenfinch	0.415	0.374
Carrion Crow	0.374	0.365
Chaffinch	0.255	0.424
Jay	0.253	0.312
Wren	0.235	0.268
Great Spotted Woodpecker	0.212	0.214
Coal Tit	0.182	0.288
Song Thrush	0.117	0.133
Long-tailed Tit	0.100	0.154
Goldfinch	0.065	0.086
Nuthatch	0.049	0.071
Sparrowhawk	0.029	0.038
Blackcap	0.025	0.042
Tree Sparrow	0.025	0.023
Mistle Thrush	0.023	0.030
Jackdaw	0.019	0.015
Goldcrest	0.018	0.041
Rook	0.016	0.016
Bullfinch	0.008	0.005
Black-headed Gull	0.008	0.045
Siskin	0.006	0.021
Tawny Owl	0.005	0.007
Pied Wagtail	0.004	0.013
Marsh/Willow Tit	0.003	0.005
Brambling	0.001	0.003
Fieldfare	0.001	0.012
Treecreeper	0.001	0.003
Yellowhammer	0.001	0.001
Reed Bunting	0.000	0.000
Redwing	0.000	0.045

Table 4.2 Occurrence rates of the 41 target species between 1995 and 2002 within the Greater London area in summer (April-July) and winter (October-February) separately. Rates are based on all weekly observations from 953 sites. Species are given in order of occurrence in the summer.

Species	Significant variables
Sparrowhawk	CHHE+ CTTR+
Feral Pigeon	FARM-
Collared Dove	CLHE^{n}
Great Spotted Woodpecker	DWOD+ FARM+
Wren	CTTR+
Robin	ALT+
Blackbird	ORCH+ DHHE^{n}
Mistle Thrush	STRE- VEGE^{u}
Blackcap	CTTR+
Goldcrest	CTTR+ DLTR^{n} RAIL+ CSTR^{n}
Long-tailed Tit	FARM+ LAND^{sru} CTTR+ VEGE^{u}
Coal Tit	ALT-
Nuthatch	BUIL- DWOD+ FARM+ MWOD+ WALL-
Treecreeper	COAS+
Rook	DWOD- CSTR- VEGE-
Jackdaw	CSTR- WATS-
Starling	DWOD- VEGE+
Chaffinch	CTTR+
Greenfinch	CLHE^{n}
Goldfinch	VEGE^{u} CTTR+
Bullfinch	DWOD+

Table 4.3 Significant predictors of species occurrence in gardens within Greater London. Only variables that were significant in each year considered and that showed consistent relative ranking of parameter estimates for different habitat levels across years are shown. Habitat codes are given in Table 4.1. + = positive association between habitat and probability of bird occurrence, - = negative association between habitat and probability of bird occurrence, u or n = non-linear (peaked or troughed respectively) association between habitat and probability of bird occurrence.

(a) Collared Dove and CLHE

Model Parameters	Habitat	Interaction
size, CLHE (winter only)	$F_{4,788}$=11.97 P<.0001	
size, CLHE (summer only)	$F_{4,694}$=11.34 P<.0001	
size, year, season, CLHE	$F_{4,1482}$=23.32 P<.0001	
size, year, season, CLHE, year*CLHE		$F_{13,1469}$=0.62 P=.8384
size, year, season, CLHE, season*CLHE		$F_{4,1478}$=0.05 P=.9959
size, year, season, CLHE, year*CLHE, season*CLHE, year*season*CLHE		$F_{18,1447}$=0.21 P=.9999

(b) Robin and ALT

Model Parameters	Habitat	Interaction
size, height (winter only)	$F_{4,788}$=17.16 P<.0001	
size, height (summer only)	$F_{4,694}$=14.86 P<.0001	
size, year, season, height	$F_{4,1482}$=30.02 P<.0001	
size, year, season, height, year*height		$F_{22,1460}$=0.43 P=.9907
size, year, season, height, season*height		$F_{4,1478}$=0.16 P=.9608
size, year, season, height, year*height, season*height, year*season*height		$F_{28,1428}$=0.26 P=1.0000

(c) Blackbird and ORCH

Model Parameters	Habitat	Interaction
size, ORCH (winter only)	$F_{1,791}$=25.74 P<.0001	
size, ORCH (summer only)	$F_{1,697}$=11.22 P=.0009	
size, year, season, ORCH	$F_{1,1485}$=34.37 P<.0001	
size, year, season, ORCH, year*ORCH		$F_{6,1479}$=1.62 P=.1371
size, year, season, ORCH, season*ORCH		$F_{1,1484}$=1.83 P=.1764
size, year, season, ORCH, year*ORCH, season*ORCH, year*season*ORCH		$F_{11,1467}$=1.10 P=.3601

(d) Blackbird and DHHE

Model Parameters	Habitat	Interaction
size, DHHE (winter only)	$F_{4,788}$=6.87 P<.0001	
size, DHHE (summer only)	$F_{4,694}$=9.53 P<.0001	
size, year, season, DHHE	$F_{4,1482}$=15.56 P<.0001	
size, year, season, DHHE, year*DHHE		$F_{14,1468}$=1.45 P=.1245
size, year, season, DHHE, season*DHHE		$F_{4,1478}$=0.06 P=.9928
size, year, season, DHHE, year*DHHE, season*DHHE, year*season*DHHE		$F_{19,1445}$=0.54 P=.9433

(e) Starling and VEGE

Model Parameters	Habitat	Interaction
size, VEGE (winter only)	$F_{2,790}$=25.26 P<.0001	
size, VEGE (summer only)	$F_{2,696}$=13.67 P<.0001	
size, year, season, VEGE	$F_{2,1484}$=36.84 P<.0001	
size, year, season, VEGE, year*VEGE		$F_{12,1472}$=0.81 P=.6392
size, year, season, VEGE, season*VEGE		$F_{2,1482}$=0.55 P=.5799
size, year, season, VEGE, year*VEGE, season*VEGE, year*season*VEGE		$F_{18,1452}$=0.21 P=.9999

Table 4.4 Results of a series of logistic regression analyses of the relationship between presence of bird species quoted and a range of habitat types. Interaction refers to the likelihood ratio test of the highest order interaction.

(f) Starling and DWOD

Model Parameters	Habitat	Interaction
size, DWOD (winter only)	$F_{1,791}$=19.89 P<.0001	
size, DWOD (summer only)	$F_{1,697}$=17.65 P<.0001	
size, year, season, DWOD	$F_{1,1485}$=38.83 P<.0001	
size, year, season, DWOD, year*DWOD		$F_{6,1479}$=0.62 P=.7158
size, year, season, DWOD, season*DWOD		$F_{1,1484}$=0.31 P=.5755
size, year, season, DWOD, year*DWOD, season*DWOD, year*season*DWOD		$F_{12,1466}$=0.31 P=.9878

(g) Chaffinch and CTTR

Model Parameters	Habitat	Interaction
size, CTTR (winter only)	$F_{4,788}$=9.54 P<.0001	
size, CTTR (summer only)	$F_{4,694}$=17.49 P<.0001	
size, year, season, CTTR	$F_{4,1482}$=24.60 P<.0001	
size, year, season, CTTR, year*CTTR		$F_{24,1458}$=0.55 P=.9629
size, year, season, CTTR, season*CTTR		$F_{4,1478}$=1.28 P=.2773
size, year, season, CTTR, year*CTTR, season*CTTR, year*season*CTTR		$F_{30,1424}$=0.30 P=.9999

(h) Greenfinch and CLHE

Model Parameters	Habitat	Interaction
size, CLHE (winter only)	$F_{4,788}$=13.20 P<.0001	
size, CLHE (summer only)	$F_{4,694}$=10.25 P<0.0001	
size, year, season, CLHE	$F_{4,1482}$=23.33 P<.0001	
size, year, season, CLHE, year*CLHE		$F_{13,1469}$=0.38 P=.9775
size, year, season, CLHE, season*CLHE		$F_{4,1478}$=0.35 P=.8464
size, year, season, CLHE, year*CLHE, season*CLHE, year*season*CLHE		$F_{18,1447}$=0.23 P=.9997

Table 4.4 Continued.

(a) CTTR

Model Parameters	Habitat	Interaction
size, CTTR (winter only)	$F_{4,919}$=5.68 P<.0002	
size, CTTR (summer only)	$F_{4,1030}$=9.30 P<.0001	
size, year, season, CTTR	$F_{4,1949}$=15.98 P<.0001	
size, year, season, CTTR, year*CTTR		$F_{24,1925}$=1.35 P<0.12
size, year, season, CTTR, season*CTTR		$F_{9,1945}$=1.47 P<0.21
size, year, season, CTTR, year*CTTR, season*CTTR, year*season*CTTR		$F_{29,1892}$=0.33 P<.9999

(b) LAWN

Model Parameters	Habitat	Interaction
size, LAWN (winter only)	$F_{4,1328}$=2.98 P<.019	
size, LAWN (summer only)	$F_{4,1477}$=3.82 P<.005	
size, year, season, LAWN	$F_{4,2805}$=6.59 P<.0001	
size, year, season, LAWN, year*LAWN		$F_{24,2781}$=1.70 P<0.02
size, year, season, LAWN, season*LAWN		$F_{4,2801}$=0.15 P<0.97
size, year, season, LAWN, year*LAWN, season*LAWN, year*season*LAWN		$F_{30,2747}$=0.37 P<0.99

(c) LAND

Model Parameters	Habitat	Interaction
size, LAND (winter only)	$F_{1,1352}$=76.09 P<.0001	
size, LAND (summer only)	$F_{1,1504}$=40.34 P<.0001	
size, year, season, LAND	$F_{2,2854}$=110.4 P<.0001	
size, year, season, LAND, year*LAND		$F_{12,2842}$=1.30 P<0.20
size, year, season, LAND, season*LAND		$F_{2,2852}$=3.87 P<0.022
size, year, season, LAND, year*LAND, season*LAND, year*season*LAND		$F_{2,2882}$=3.53, P<0.030

Table 4.5 Results of a series of logistic regression analyses of the relationship between species richness (out of 41 selected species) and a range of habitat types. Interaction refers to the likelihood ratio test of the highest order interaction.

Species	BBS trend	GBW trend
Woodpigeon	Increase	NS
Feral Pigeon	Decrease	Increase
Collared Dove	NS	NS
Blackbird	Decrease	NS
Blue Tit	Increase	NS
Carrion Cow	Increase	NS
Magpie	NS	NS
Starling	Decrease	Decrease
House Sparrow	Decrease	Decrease
Greenfinch	Increase	NS

Table 4.6 A comparison of trends in BBS (1994-2002) and GBW (1995-2002) for selected species. NS = not significant.

Figure 4.1 Mean number of species (out of a maximum 41) per garden in each 1-km grid square in Greater London recorded during GBW. Summer size 1 = 3-16.5 spp, 2 = 17-30 spp, 3 = 30.5-34.3 spp, 4 = 41 spp; winter size 1 = 6-16.33 spp, 2 = 17-21.75 spp, 3 = 22-33 spp, 4 = 41 spp.

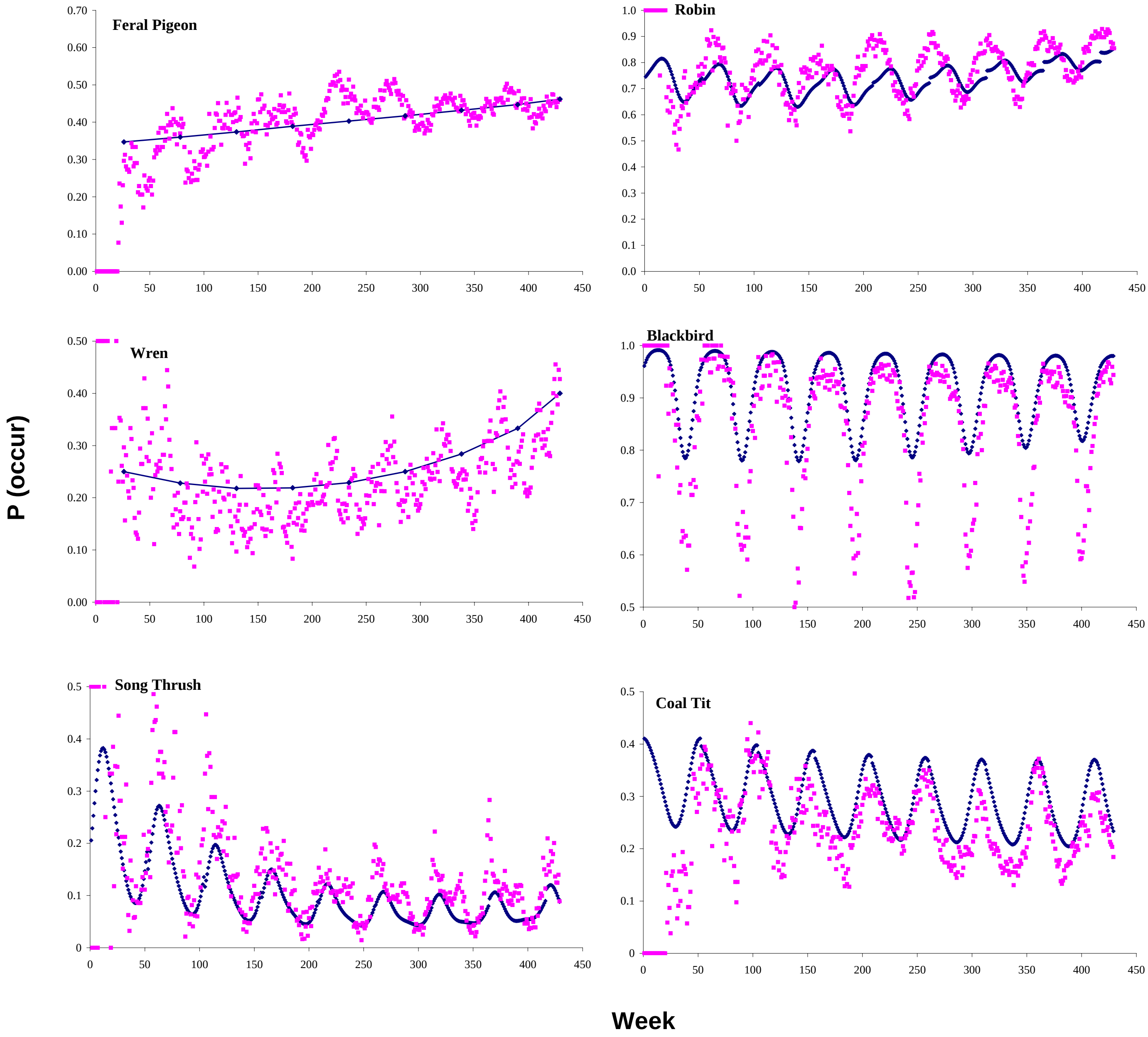


Figure 4.2 Weekly occurrence rates (grey dots) and probability of occurrence derived from logistic regression of selected species occurring in gardens. Only species showing significant annual effects or significant week*year interactions are shown. Week is expressed as a continuous variable from week 1 (January 1995) to week 450 (March 2002).

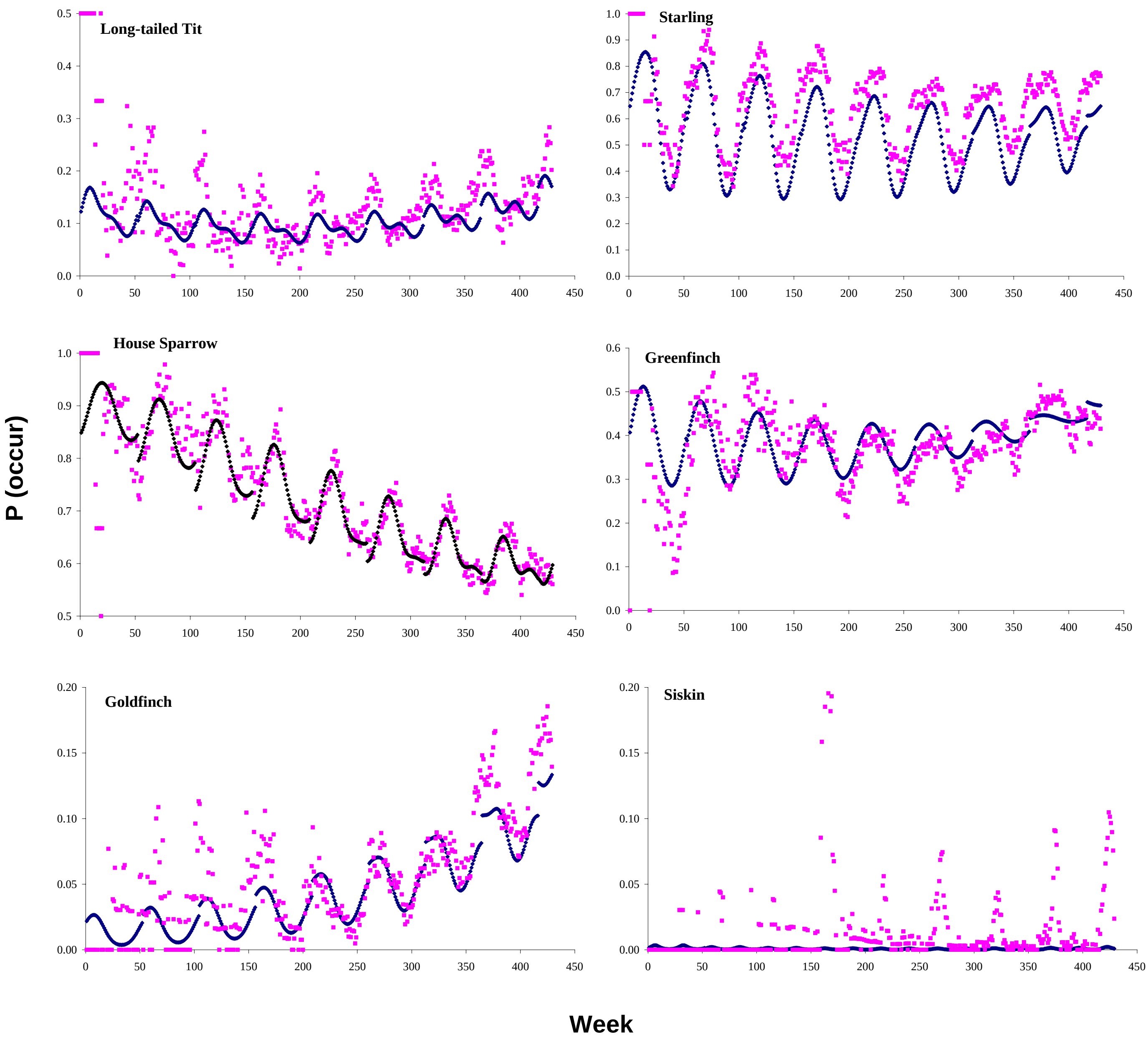


Figure 4.2 Continued.

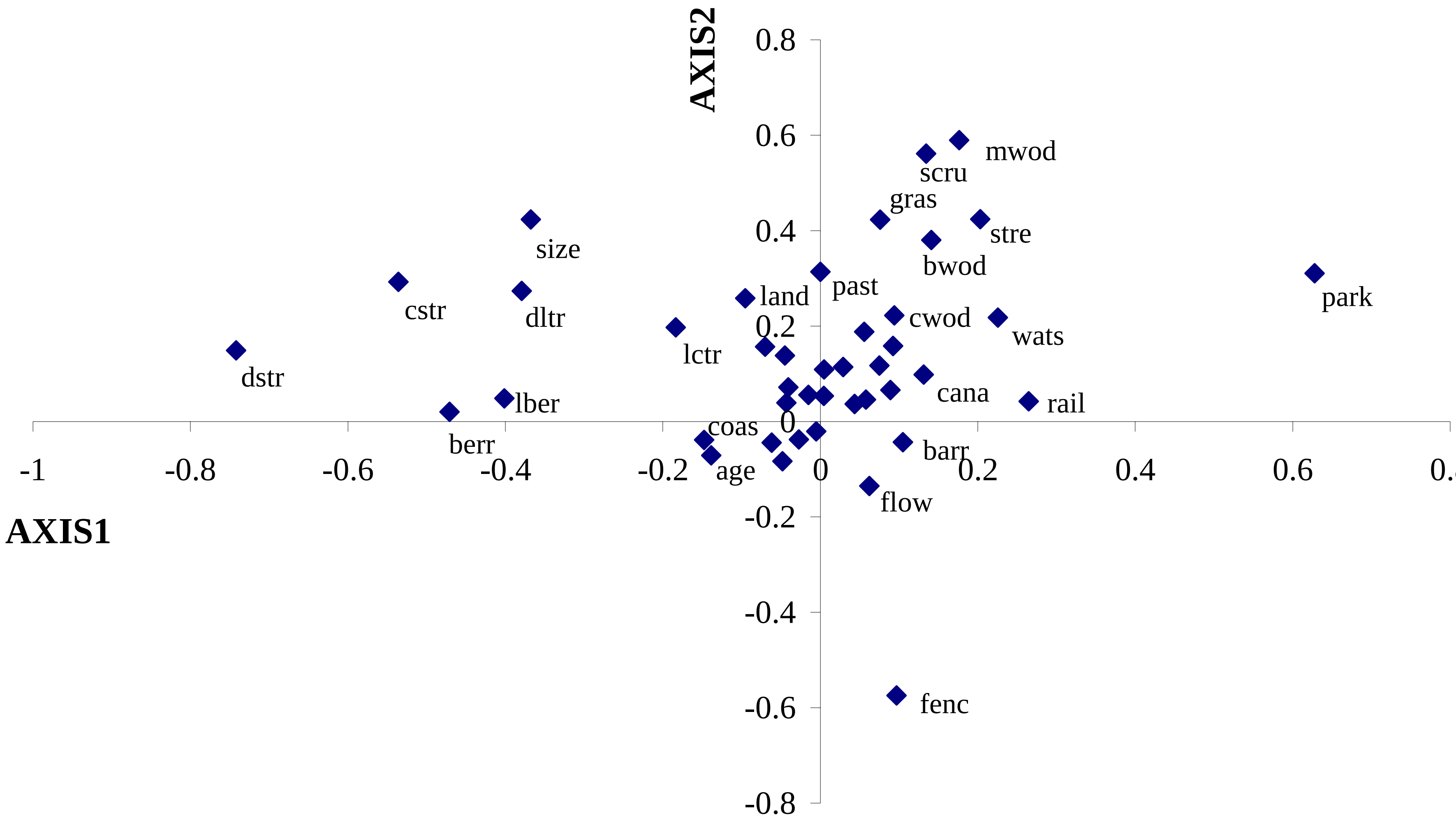


Figure 4.3 Bi-plot of scores from the first two axes derived from PCA on data describing the habitat of gardens in Greater London. N = 416.

CHAPTER 5 ORDINAL REGRESSION AND INTERVAL-CENSORED POISSON MODELS OF THE RELATIONSHIP BETWEEN GARDEN HABITAT AND BIRD ABUNDANCE IN GBW

Steve Freeman, Dan Chamberlain

1. INTRODUCTION

Cannon *et al*. (in press) determined change in national reporting rates of garden birds between 1995 and 2003, considering only presence/absence data from GBW. Similarly, Chamberlain *et al*. (2004) analysed variations in probability of occurrence of garden birds in relation to broad-scale habitat gradients using GBW data. Whilst these papers produced some very interesting results, the use of presence/absence data, rather than abundance data, must ultimately lose information. This is particularly likely to be the case for very common species (e.g. Blackbird, Blue Tit and Robin) which occur in a very high proportion of gardens. Furthermore, Cannon *et al*. (in press) make a comparison between BBS data (based on counts) with presence/absence data from GBW, a comparison that was repeated for Greater London data in Chapter 4. Cannon *et al*. (in press) detected some intriguing positive correlations, indicating similar trends in garden reporting rate and species abundance in the wider population. However, London-specific trends showed a close match between BBS and GBW in only two species, House Sparrow and Starling. It may be better to assess measures of abundance rather than presence/absence in both datasets where possible in order for an improved comparison of trends.

2. AIMS

For most species, only presence/absence in the garden is recorded in GBW. In Chapter 4, the trends in reporting rate over time and the relationship between the habitats in and around London gardens and the birds therein were investigated using only this data. For the ten most commonly encountered species however, additional, more detailed information on the counts of birds were available. In this section, how the relationship between abundance and garden habitat might be modelled is considered. The goal at this stage is to develop novel analytical techniques to determine bird-habitat associations of garden birds in London rather than present a complete list of results for all species. Furthermore, trends over time are derived from these count species from GBW and compared with BBS trends.

3. METHODS

Even for these common species, the data do not take the form of literal, integer-valued counts of individual birds. Rather, each week the species in question is assigned to one of five ordinal abundance categories, defined as ‘absent (X)’, ‘scarce (A)’, ‘not scarce (B)’, ‘common (C)’ and ‘very common (D)’. The categories are formally defined in terms of numbers observed, though this categorisation differs so that each species is categorised by numbers most suitable for its own data; thus category C corresponds to 11-20 individuals for the gregarious starling, but only 3-4 birds for the Greenfinch (Table 5.1). This affects the fine detail of the interpretation, but does not affect the workings of the analytical method employed here.

3.1 Analysis

It is a simple matter to assume some probability distribution for the weekly bird numbers and express the likelihood for the data in terms of the corresponding cumulative probability distribution. Here however, inference is based upon the technique of ordinal regression (McCullagh 1980, McCullagh & Nelder 1989), which is more readily accommodated by familiar GLM packages, such as SAS (Bender & Benner 2000), as employed in the present study. Models based on a specified probability distribution are considered later in the section. An additional advantage of the ordinal approach is not requiring the assumption of this underlying distribution for the data, most probably the Poisson, which may not be satisfactory in practice. Flocking may cause overdispersion in social birds, and data for less ubiquitous species may be prone to zero-inflation (Vandenbroek 1995). The ordinal regression technique, however, is as yet little used with ecological data (Thomson *et al*. 1998), though see Paradis *et al*. (2000) for an application to brood sizes. A brief introduction to the method is given in Appendix IV.

4. RESULTS

Habitat associations - Consider for example the Chaffinch data. From Chapter 4, presence of Chaffinches in gardens appears to vary with the number of large, coniferous trees (CTTR – habitat codes are given in Chapter 4). Consider the dependence of Chaffinch abundance on CTTR, based on an example set of data for Chaffinches recorded in week 1 (i.e. mid-winter) of 2001. An ordinal regression model was fitted based on garden size (small, medium or large) and CTTR (categories A (no trees) to E) as predictor variables. The results (Fig. 5.1a,b) suggest numbers of chaffinches increasing with garden size, and a greater number in gardens of CTTR category C (2-4 trees). (Recall that the greater the number of birds present, the lower the curve on the graph). However there is a marked change if data for the same species in week 26 (mid-summer) of the same year are considered. Although the cumulative proportions of gardens holding at least a designated number of Chaffinches remains very similar for gardens of CTTR category E (those with >10 coniferous trees), the same proportions drop markedly, compared to week one, for gardens in other categories (Fig. 5.1c,d).

Clearly, it would be facile to over-generalise from the analysis of these two weeks' data, as weather conditions at the time are likely to have an influence on birds' garden usage, for example. Taken at face value, the results of Fig. 5.1 appear to suggest that while gardens rich in conifers are favoured both in the breeding season and in winter, those with fewer trees of this kind are deserted to some extent by the summer. The mid-summer analysis was repeated for each of the years 1997-2000. Results (coefficients of the predictor variables) are given in Table 5.2. Greater numbers of birds are indicated by the model via smaller (more negative) values of the coefficients, thus the above pattern is approximately repeated in summer of 2000 and 1999 (coefficients for CTTR categories A-D are all positive, compared with zero for category E, and greatest in category A) but the pattern is lost in 1997-1998. However, note that sample sizes are smaller here, resulting in larger standard errors and a likelihood ratio test for the equivalence of all CTTR categories is actually not significant in 1997 ($\chi^2_4 = 4.53$, $P > 0.3$) or 1999 ($\chi^2_4 = 6.27$, $P > 0.1$).

Further analysis shows that the models can be reduced by removing the 'size' effect , without significant deterioration in the quality of the fit, for years 1997-2000. This simplification however has little effect upon the ranking of the coefficients associated with CTTR, and details are omitted here.

The annual pattern of variation between habitat types is also somewhat variable in winter (Table 5.2b), albeit with large standard errors in early years especially. More annual consistency is found when analysing the relationship similarly between Starling abundance and the proximity of broad-leaved woodland (Table 5.3). Note here that the predictor variable is a simple dichotomy, reducing the number of parameters and the complexity of the model. Every year, Starlings exhibit a preference for gardens away from broad-leaved woodland (as the coefficients in Table 5.3 are all positive). The difference is, however, more pronounced in winter (Table 5.3a) than in summer (Table 5.3b).

Reduced parameter models were produced by combining the data for all years (separately in weeks 1 and 26), and accounting for annual changes by introducing year as an additional predictor variable, a factor with five levels. Thus the framework of the model has cumulative proportions in the various abundance categories that are additive, hence proportional (on the appropriate scale), in the different years. The corresponding reduction in parameters can be expected to increase precision, and the occupancy proportions employed as single estimates of 'average' occupancy over the period. Under this model, for example, the habitat coefficient in the Starling/broad-leaved woodland relationship becomes 0.77 (standard error 0.15) in winter and 0.47 (0.13) in summer, both of which appear sensible in the light of Table 5.3. For the Chaffinch, a preference for gardens with large coniferous trees is now suggested both for winter and summer (Table 5.4) though the difference between gardens with one tree or less, and those with more (2-9) strengthens in the winter.

The structure of ordinal regression models does not permit the evaluation of estimated numbers of birds recorded in gardens of various types. For this, we have to assume some underlying distribution for the count data. Such an analysis of the Starling counts in relation to broad-leaved woodland is demonstrated below, assuming a Poisson distribution for the numbers of birds present.

Under the Poisson distribution, the probability p(x) of observing x birds is given by:

$$p(x) = \frac{\exp(-\lambda)\lambda^x}{x!}$$

where λ is the model parameter, the mean of the Poisson distribution. Thus the probability P(x) of observing x birds or less is

$$P(x) = \sum_{X=0}^{x} p(X)$$

At an arbitrary garden *i* in the survey the number of birds counted is known only to be in some interval $\{L_i \leq x_i \leq U_i\}$, and the probability of such a count readily follows as $P(U_i) - P(L_i - 1)$. The likelihood function for the entire data is formed by multiplying together these probabilities for every garden in the survey. In the manner of a standard linear model, variation between gardens is accommodated via extending to site-specific parameters λ_i, related to independent variables of interest in the form of a covariate vector $\mathbf{x}_i$, thus:

$$\log(\lambda_i) = \beta' x_i$$

where the log link function is used to ensure fitted counts remain positive and $\boldsymbol{\beta}$ is a vector of estimable coefficients.

This Poisson model was used to estimate counts of Starlings, in weeks 1 and 26 separately, and quantify the difference in average numbers in gardens with and without surrounding broad-leaved woodland (Table 5.3c). The results of a simple model (model (b) in Table 5.3c) in which DWOD is the only variable reinforce those of the ordinal regression, with an aversion to gardens reporting the proximity of broad-leaved woodland, which is most pronounced in winter (week 1). Converting the coefficients back from the log scale, in week 1 there are an estimated 2.38 birds per garden in the vicinity of broad-leaved woodland but 5.23 birds elsewhere. A likelihood-ratio test confirms the significance of this result (X^2_1 = 2(3713.8-3595.6) = 236.4, P<0.001). This result is confirmed by applying more complex models in which variation between gardens of different sizes is controlled (model (f), showing that gardens in the vicinity of broad-leaved woodland receive only 43.8% of the birds recorded elsewhere) and between years (model (d), in which this percentage is similar at 46.6%). In these more complex models the specific values of the fitted counts will obviously vary between gardens of different sizes, or in different years, while retaining the percentages quoted. Full details are not given here, though they are calculable from the coefficients of Table 5.3c. Similar results also are obtained from the analysis of data in week 26, with the simple thumbnail sketch of model (b) implying 3.75 birds per garden in the vicinity of broad-leaved woods and 5.07 otherwise.

Returning to the additive ordinal model, an interesting correspondence is revealed between two species (Greenfinch and Chaffinch) and the surrounding land use (rural, suburban or urban). These were the two species which, in the previous section, appeared in significantly different proportions of rural, suburban and urban gardens every year. The ordinal model was fitted, with additive year effects, and obtained the results of Table 5.5. Both species show a marked aversion to urban gardens. However, both showed a preference for suburban over rural sites in winter that was much reduced or reversed by the summer.

Blackbirds showed a preference for gardens in which the proportion of the boundary taken up by high deciduous hedges took a value in the middle of the possible range, i.e. as opposed to gardens with boundaries either dominated by, or largely devoid of, such hedges (Table 5.6). This phenomenon was consistent between the summer and winter weeks. Comparison of the maximised log-likelihood values with those for a reduced model in which the year effect was omitted (thus imposing identical cumulative frequencies every year in each abundance class, for gardens of the same size) revealed a negligible change in fit for week 26 (X^2_4 = 1.03, P=0.90) and, hence, little effect upon the habitat coefficients. Though the corresponding change in week 1 also failed to reach significance (X^2_4 = 5.76, P=0.22), it was rather greater and correspondingly altered habitat coefficients more markedly, though maintaining their ordering in terms of magnitude (Table 5.6). Cumulative proportions of small gardens, with varying extent of high deciduous boundary hedge, recording Blackbirds in the various abundance categories are shown in Fig. 5.2.

A preference by Starlings for gardens with increasing amounts of vegetables is revealed by the model of Table 5.7. Starlings become commoner as the proportion of vegetables increases from zero to 50%. Though the results suggest gardens with >50% vegetables are the least favoured category, this is based on only four such gardens, little utilised by Starlings, making it impossible to interpret meaningfully.

Temporal trends - This section is concluded by noting that the technique employed to model interval-censored Poisson data also permits us to model the BBS and GBW data simultaneously, with a common trend in both, and to thereby test for a difference between trends indicated by the two surveys. This works by multiplying together the likelihood functions for each and fitting a model to this joint likelihood. Here a model is adopted with separate year effects for the two surveys in 2001, but with the numbers of birds in the two surveys in the other years additionally constrained to remain in the same proportion, relative to the 'base' year 2001, in each survey. The maximised log-likelihood value for this model can be compared with the sum of the log-likelihoods for models fitted individually to the two sets of data, to derive a valid test statistic for the hypothesis that trends in the two surveys are equivalent.

As an example, a joint analysis of the data for week 26 of the GBW (in the breeding season to coincide with BBS) is considered in combination with the BBS data. Annual estimated trends are shown in Fig. 5.3, both under the joint model (where the two can be seen to fluctuate in parallel, by virtue of the model structure) and under separate individual models. The most striking similarity was for House Sparrow, where both BBS and GBW trends were almost identical to the joint model (and therefore to each other). For other species, the models were fairly similar, but there were some notable differences in individual years (e.g. Collared Dove, Starling, Chaffinch). In many cases, with the exception of Blackbird and Starling, trends were fairly stable in both BBS and GBW. All species apart from Robin, Blackbird and Dunnock showed significant differences between trends. However, due to the very large sample size of GBW (c.1000 gardens in Greater London), even very small differences may be detected as significant, so in this case, visual interpretation is probably more biologically meaningful.

It should be noted that in this final analysis the models take no account of differences between gardens, or between BBS squares, within the individual surveys. Thus, any decline/increase in a particular year may be in part due to a decreased or increased number of favoured sites surveyed. In principle, the model can be adapted to control for this sort of variability, enhancing the credibility of the test, though the enormous increase in the numbers of parameters to estimate is a significant computational burden. However, the joint analysis of multiple surveys is one of some potential.

5. DISCUSSION

This largely methodological chapter has presented the development of statistical procedures for analysing categorical abundance data in GBW. The techniques outlined here will provide the opportunity to assess the features affecting the abundance rather than just the presence/absence of selected species in gardens. The examples given were based on the simpler significant habitat associations with species presence presented in Chapter 4, but the technique developed enables a better quantification of precise effects. In brief, it is shown that: Starlings are less abundant in gardens that are in close proximity to deciduous woodland and peak in abundance in gardens that have intermediate cover of vegetables; Blackbird abundance is greatest in gardens with an intermediate high deciduous hedge cover; Chaffinch abundance is highest in gardens with intermediate numbers of deciduous trees and highest in suburban gardens; and, Greenfinch abundance is higher in suburban gardens. Furthermore, the technique allows an estimation of mean abundance in different habitat categories if an underlying Poisson distribution is assumed. These statistical techniques have great potential in examining species hypotheses about effects of particular habitat types on abundance in the commoner species recorded in GBW.

An extension to the technique has allowed, for the first time, a statistically rigorous comparison of abundance trends in GBW and BBS that reveals whether temporal trends in gardens are equivalent to those in the wider countryside. For London populations, the matching between BBS trends and GBW trends was fairly close. There were some diversions between the two trends, but nothing too extreme, suggesting that GBW count data is probably a generally good monitor of population trends in the wider countryside for the ten species analysed. The fit was especially good for House Sparrow. Similar close correlations between garden trends and wider trends have been shown between reporting rate (based on presence/absence) in GBW and BBS (Cannon *et al*. in press) and over a longer time span for abundance at garden feeders and Common Bird Census data (Chamberlain *et al*. in press). The House Sparrow population may be particularly well monitored in gardens because it is one of the few truly urban species.

Species	**A**	**B**	**C**	**D**
Collared Dove	1	2	3-4	5+
Dunnock	1	2	3	4+
Robin	1	2	3	4+
Blackbird	1	2	3	4+
Blue Tit	1	2	3-4	5+
Great Tit	1	2	3-4	5+
Starling	1-5	6-10	11-20	21+
House Sparrow	1-5	6-10	11-20	21+
Chaffinch	1	2	3-4	5+
Greenfinch	1	2	3-4	5+

Table 5.1 Numbers of birds in each abundance category (A-D) recorded in GBW for the 10 most abundant species.

Week 26		2001	2000	1999	1998	1997
No. of observations		305	266	193	104	85
Coefficients	S (small)	1.84(0.61)	0.26(0.52)	0.53(0.58)	-0.22(0.79)	-0.44(0.87)
	M (medium)	0.26(0.33)	-0.33(0.36)	-0.28(0.40)	-0.89(0.58)	-0.97(0.62)
	L (large)	0.00(-)	0.00(-)	0.00(-)	0.00(-)	0.00(-)
	CTTR A	1.96(0.54)	1.83(0.62)	1.83(0.74)	1.66(1.05)	0.12(1.30)
	CTTR B	1.69(0.57)	0.75(0.62)	1.67(0.76)	-0.05(1.06)	-0.83(1.38)
	CTTR C	1.43(0.55)	0.62(0.60)	1.59(0.76)	0.74(1.08)	-0.88(1.31)
	CTTR D	1.73(0.62)	0.51(0.68)	1.28(0.84)	-0.30(1.13)	-1.28(1.37)
	CTTR E	0.00(-)	0.00(-)	0.00(-)	0.00(-)	0.00(-)
-2 log L	{size} model only	493.22	487.36	377.36	208.05	168.14
	{size + CTTR} model	480.49	470.05	371.09	196.48	163.61
X^2 (test of equality between habitat types)		12.73	17.31	6.27	11.57	4.53

Table 5.2a Model results for ordinal regression models fitted to Chaffinch data in week 26 of 5 separate years. The test statistic X^2, referred to χ^2_4 tables, is a test of the equality of bird numbers in all gardens of the same size, irrespective of the numbers of large coniferous trees.

Week 1		2001	2000	1999	1998	1997
No of observations		273	192	143	94	70
Coefficients	S	0.91(0.45)	1.82(0.52)	1.61(0.58)	0.69(0.78)	-0.24(0.78)
	M	0.07(0.31)	0.23(0.34)	0.53(0.40)	-0.40(0.56)	-0.45(0.60)
	L	0.00(-)	0.00(-)	0.00(-)	0.00(-)	0.00(-)
	CTTR A	0.32(0.57)	1.52(0.69)	-0.26(0.75)	0.81(0.91)	1.30(1.05)
	CTTR B	0.21(0.59)	0.98(0.71)	0.54(0.81)	0.94(1.01)	1.76(1.11)
	CTTR C	-0.53(0.57)	0.83(0.70)	-0.21(0.77)	0.61(0.93)	-0.95(1.04)
	CTTR D	-0.16(0.63)	1.56(0.81)	-0.56(0.84)	-0.16(1.00)	-0.18(1.12)
	CTTR E	0.00(-)	0.00(-)	0.00(-)	0.00(-)	0.00(-)
-2 log L	{size} model only	703.21	533.64	384.44	243.22	203.76
	{size + CTTR} model	694.84	525.73	380.27	240.50	185.91
X^2 (test of equality between habitat types)		8.37	7.91	4.17	2.72	17.85

Table 5.2b Model results for ordinal regression models fitted to Chaffinch data in week 1 of 5 separate years. The test statistic X^2, referred to χ^2_4 tables, is a test of the equality of bird numbers in all gardens of the same size, irrespective of the numbers of large coniferous trees.

Week 26		2001	2000	1999	1998	1997
No. of observations		439	373	277	155	
Coefficients	S	1.07(0.28)	1.30(0.31)	0.33(0.36)	0.87(0.44)	
	M	0.46(0.24)	0.08(0.26)	-0.53(0.31)	0.08(0.38)	
	L	0.00(-)	0.00(-)	0.00(-)	0.00(-)	
	DWOD present	0.44(0.24)	0.62(0.24)	0.13(0.28)	0.61(0.36)	
-2 Log L	{size} model only	1289.64	1075.37	789.41	450.06	
	{size+DWOD} model	1286.01	1068.62	789.22	447.11	
X^2 {test of equality between habitat types		3.63	6.75	.19	2.95	

Table 5.3a Model results for ordinal regression models fitted to Starling data in week 26 of 5 separate years. The test statistic X^2, referred to χ^2_1 tables, is a test of equality of bird numbers in all gardens of the same size, irrespective of the proximity of broad-leaved woodland.

Week 1		2001	2000	1999	1998	1997
No. of observations		410	292	212	147	109
Coefficients	S	0.26(0.29)	0.23(0.34)	0.83(0.40)	0.82(0.47)	0.58(0.50)
	M	-0.35(0.26)	-0.41(0.30)	-0.11(0.34)	0.35(0.40)	-0.16(0.43)
	L	0.00(-)	0.00(-)	0.00(-)	0.00(-)	0.00(-)
	DWOD present	0.54(0.24)	0.81(0.28)	0.90(0.34)	0.17(0.39)	0.87(0.48)
-2 log L	{size} model only	1160.31	830.84	601.34	445.78	340.43
	{size+DWOD} model	1155.10	821.89	593.92	445.59	336.95
X^2 (test of equality between habitat types)		5.21	8.95	7.42	.19	3.48

Table 5.3b Model results for ordinal regression models fitted to Starling data in week 1 of 5 separate years. The test statistic X^2, referred to χ^2_1 tables, is a test of the equality of bird numbers in all gardens of the same size, irrespective of the proximity of broad-leaved woodland.

(i) Week 1		Parameters in model					
Model Coefficient		(a) intercept (I) only	(b)I+ DWOD	(c)I+year	(d)I+year+DWOD	(e)I+size	(f)I+size+DWOD
Intercept		1.5521	1.6537	1.399	1.4967	1.5729	1.7164
size	small					-0.3937	-0.4669
	medium					0.1014	0.0683
	large					0.0000	0.0000
DWOD	absent		0.0000		0.0000		0.0000
	present		-0.7872		-0.7643		-0.8233
Year	1997			0.6613	0.6585		
	1998			0.3934	0.4004		
	1999			0.0914	0.0917		
	2000			-0.0066	-0.0064		
	2001			0.0000	0.0000		
- log likelihood		3713.8	3595.6	3603.1	3489.4	3642.5	3510.4

(ii) Week 26		Parameters in model					
Model Coefficient		(a)intercept (I) only	(b)I+DWOD	(c)I+year	(d)I+year+DWOD	(e)I+size	(f)I+size+DWOD
Intercept		1.5703	1.6231	1.6234	1.6759	1.6645	1.7433
Size	small					-0.7285	-0.7700
	medium					0.0604	0.0534
	large					0.0000	0.0000
DWOD	absent		0.0000		0.0000		0.0000
	present		-0.3021		-0.3029		-0.3794
Year	1997			0.1315	0.1248		
	1998			-0.0089	-0.0003		
	1999			-0.2151	-0.2188		
	2000			-0.0745	-0.0709		
	2001			0.0000	0.0000		
- log likelihood		3753.8	3728.2	3731.8	3706.1	3590.7	3550.0

Table 5.3c Estimated coefficients and maximized log-likelihood values under a range of Poisson models for starling numbers in (i) week 1 and (ii) week 26.

Large Coniferous Trees	winter (week 1)	summer (week 26)
0	0.62(0.32)	1.91(0.32)
1	0.64(0.34)	1.25(0.34)
2-4	-0.04(0.33)	1.12(0.33)
5-9	0.08(0.36)	0.93(0.36)
10+	0.00(-)	0.00(-)

Table 5.4 Ordinal regression coefficients with standard errors from models fitted to Chaffinch data, with additive predictor variables {size, year, CTTR}

Landuse / Habitat	Chaffinch		Greenfinch	
	winter (week 1)	**summer (week 26)**	**winter (week 1)**	**summer (week 26)**
Rural	-0.49(0.44)	-2.03(0.52)	-0.35(0.45)	-0.79(0.39)
Suburban	-1.22(0.24)	-1.82(0.38)	-0.86(0.23)	-1.07(0.21)
Urban	0.00(-)	0.00(-)	0.00(-)	0.00(-)

Table 5.5 Ordinal regression coefficients with standard errors from models fitted to Chaffinch and Greenfinch data, with additive predictor variables {size, year, LAND}.

High deciduous hedges (Percentage of boundary)	winter (week 1)		summer (week 26)	
	(a) with annual variation	**(b) no annual variation**	**(a) with annual variation**	**(b) no annual variation**
0	-0.38(1.06)	-0.50(1.06)	0.20(0.91)	0.19(0.91)
1-25	-0.73(1.07)	-0.85(1.06)	-0.18(0.92)	-0.18(0.92)
26-50	-0.58(1.08)	-0.70(1.08)	-0.71(0.94)	-0.71(0.94)
51-75	-0.05(1.40)	-0.13(1.39)	0.10(1.14)	0.10(1.14)
76+	0.00(-)	0.00(-)	0.00(-)	0.00(-)

Table 5.6 Ordinal regression coefficients with standard errors from models fitted to blackbird data, with additive predictor variables (a) {size, year, DHHE} and (b) {size, DHHE}

Vegetables (Percentage of garden covered)	winter (week 1)	summer (week 26)
0	-1.74(1.08)	-0.11(0.90)
1-25	-2.11(1.08)	-0.51(0.91)
26-50	-2.96(1.12)	-0.82(0.94)
51-75	0.00(-)	0.00(-)
76+	-	-

Table 5.7 Ordinal regression coefficients with standard errors from models fitted to starling data, with additive predictor variables {size, year, VEGE}.

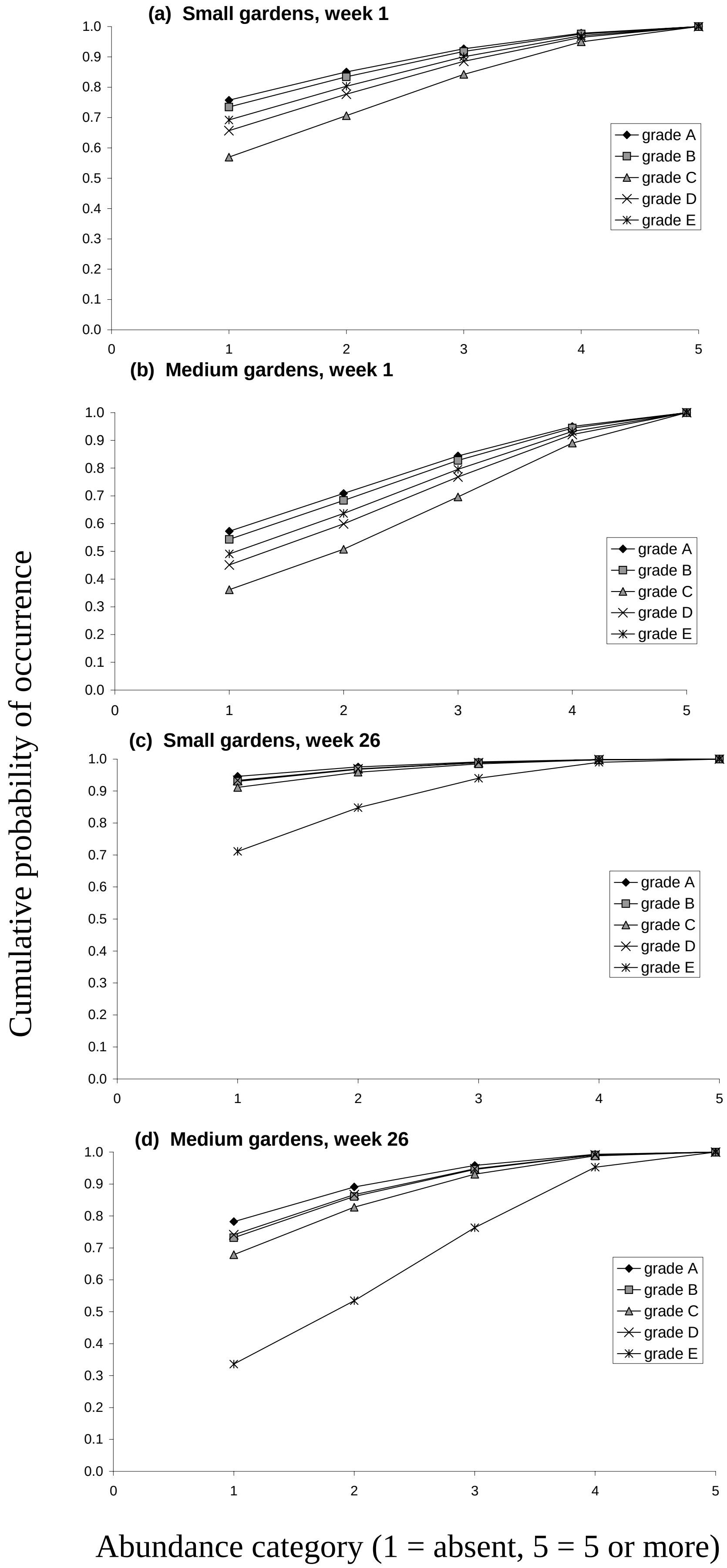


Figure 5.1 Probability of occurrence of different abundance categories of Chaffinch in relation to large coniferous trees (CTTR) in different seasons and gardens of different size in Greater London. Note that a greater number of birds is shown by a *lower* line.

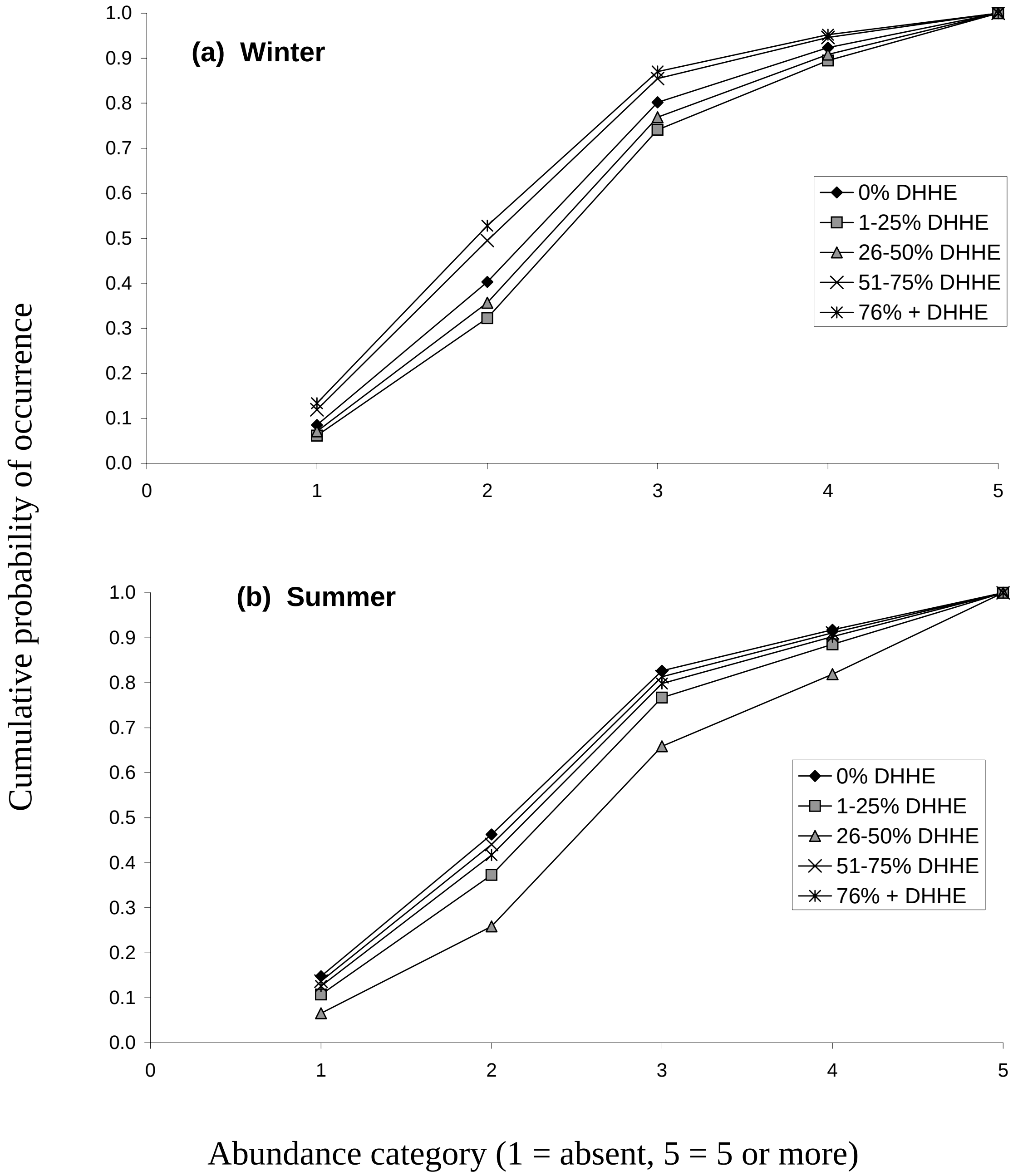


Figure 5.2 Probability of occurrence of different abundance categories of Blackbird in relation to high deciduous hedge cover (DHHE) in different seasons in Greater London.

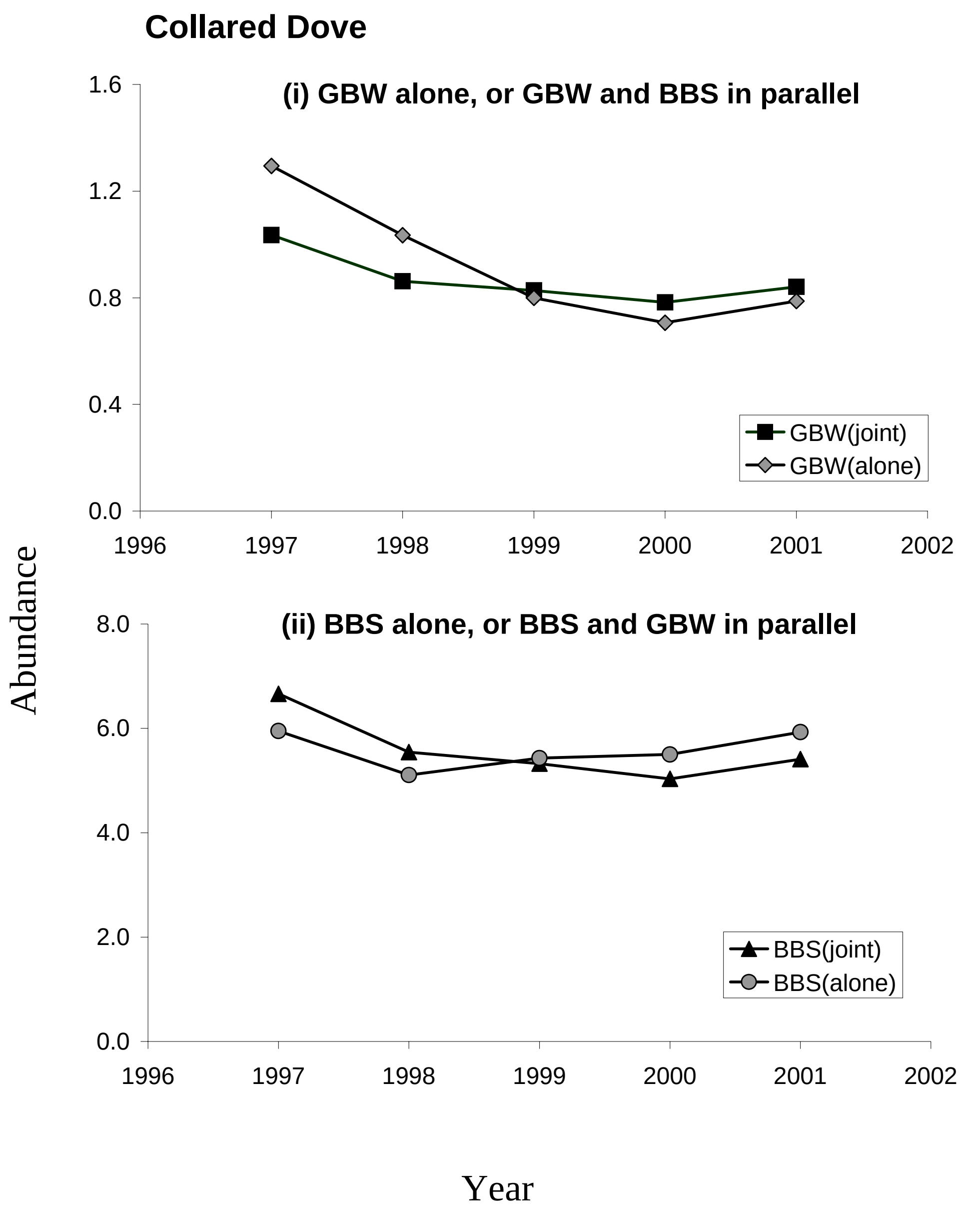


Figure 5.3 A comparison of trends in bird abundance from GBW and BBS data within Greater London, where GBW abundance is categorical. GBW data is from week 26 only to coincide with the breeding season.

Robin

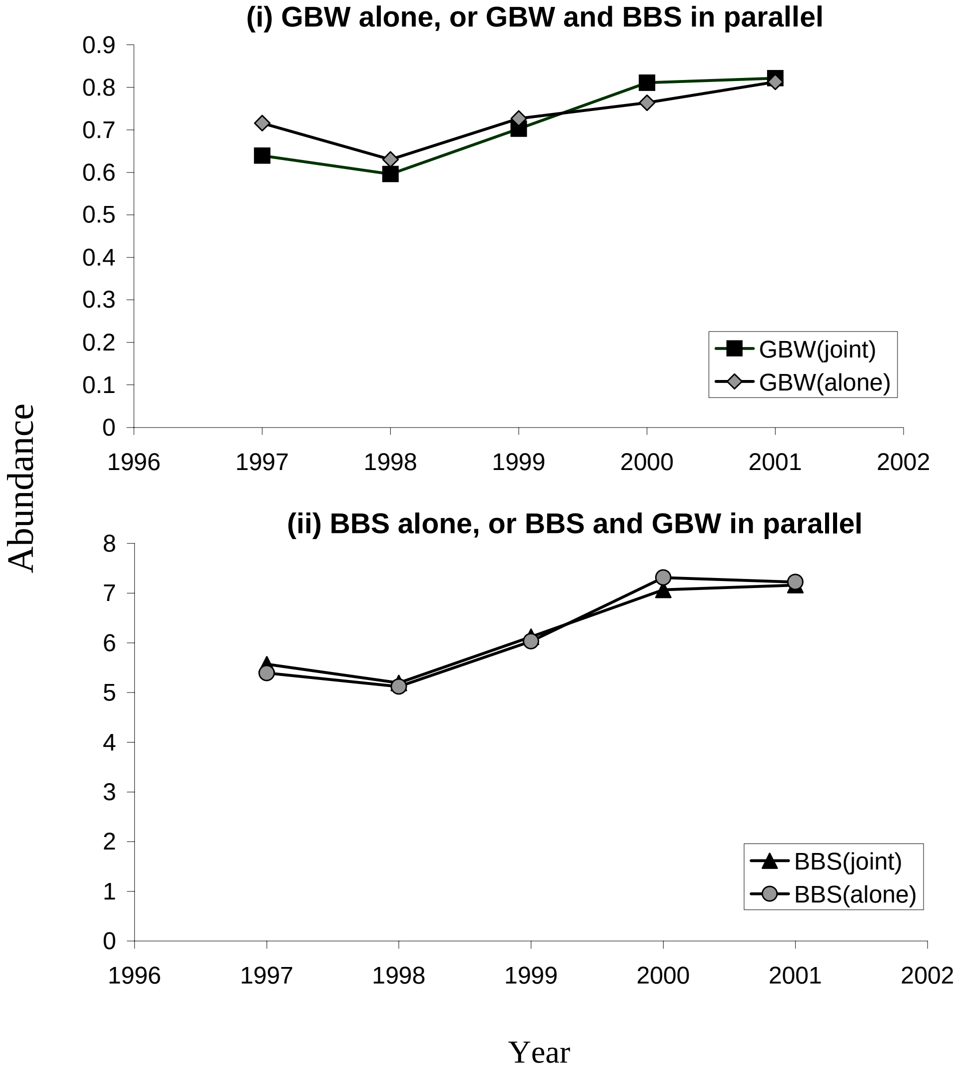


Figure 5.3 Continued.

Dunnock

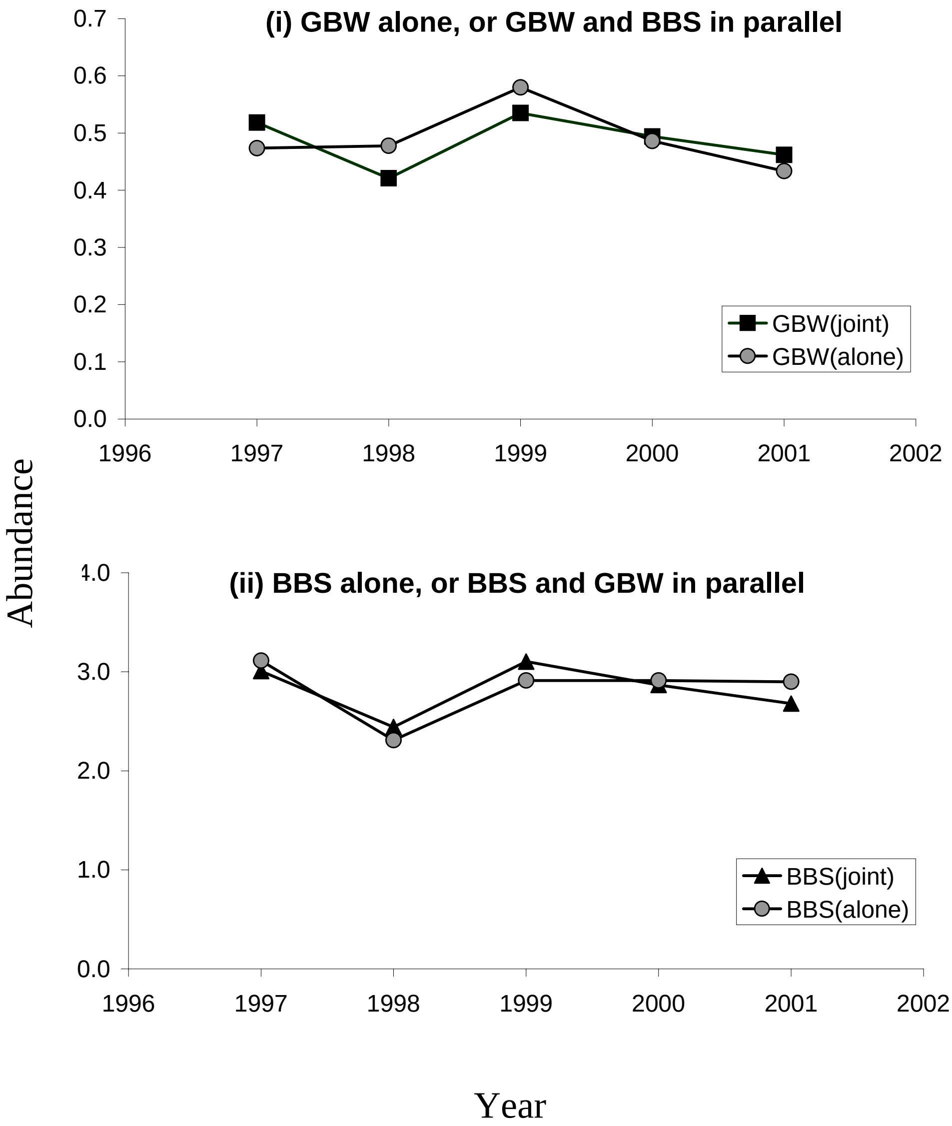


Figure 5.3 Continued.

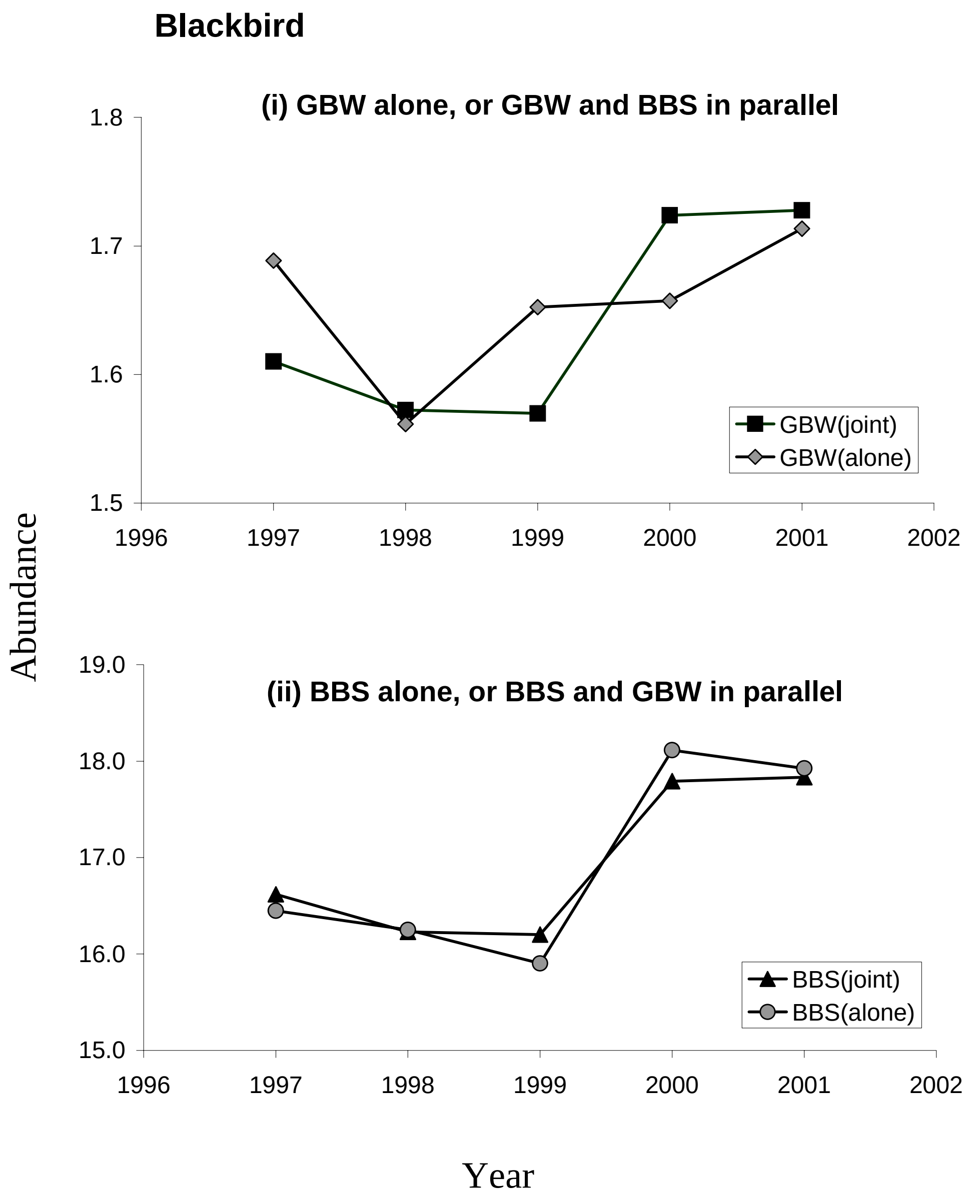


Figure 5.3 Continued.

Great Tit

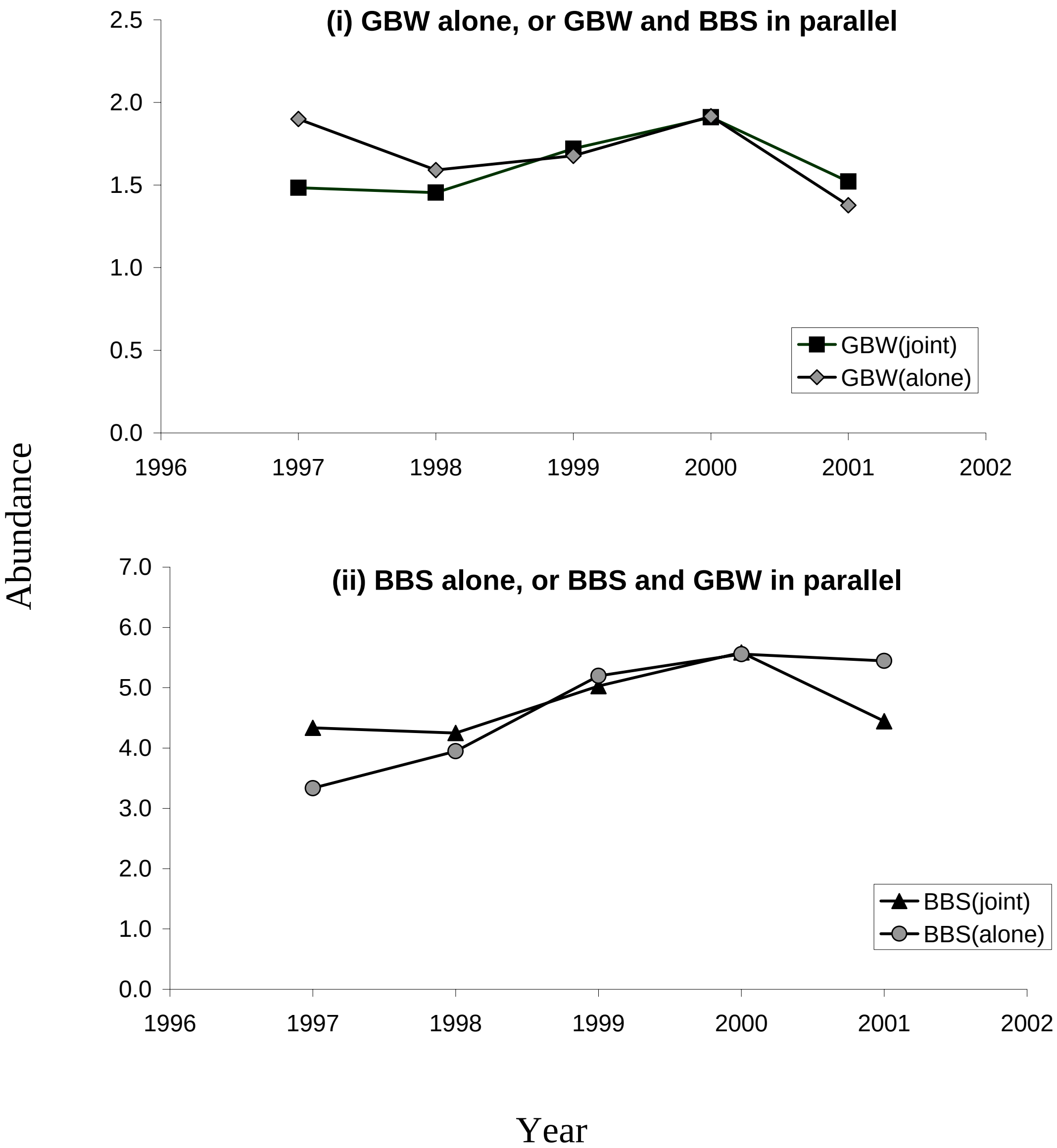


Figure 5.3 Continued.

Blue Tit

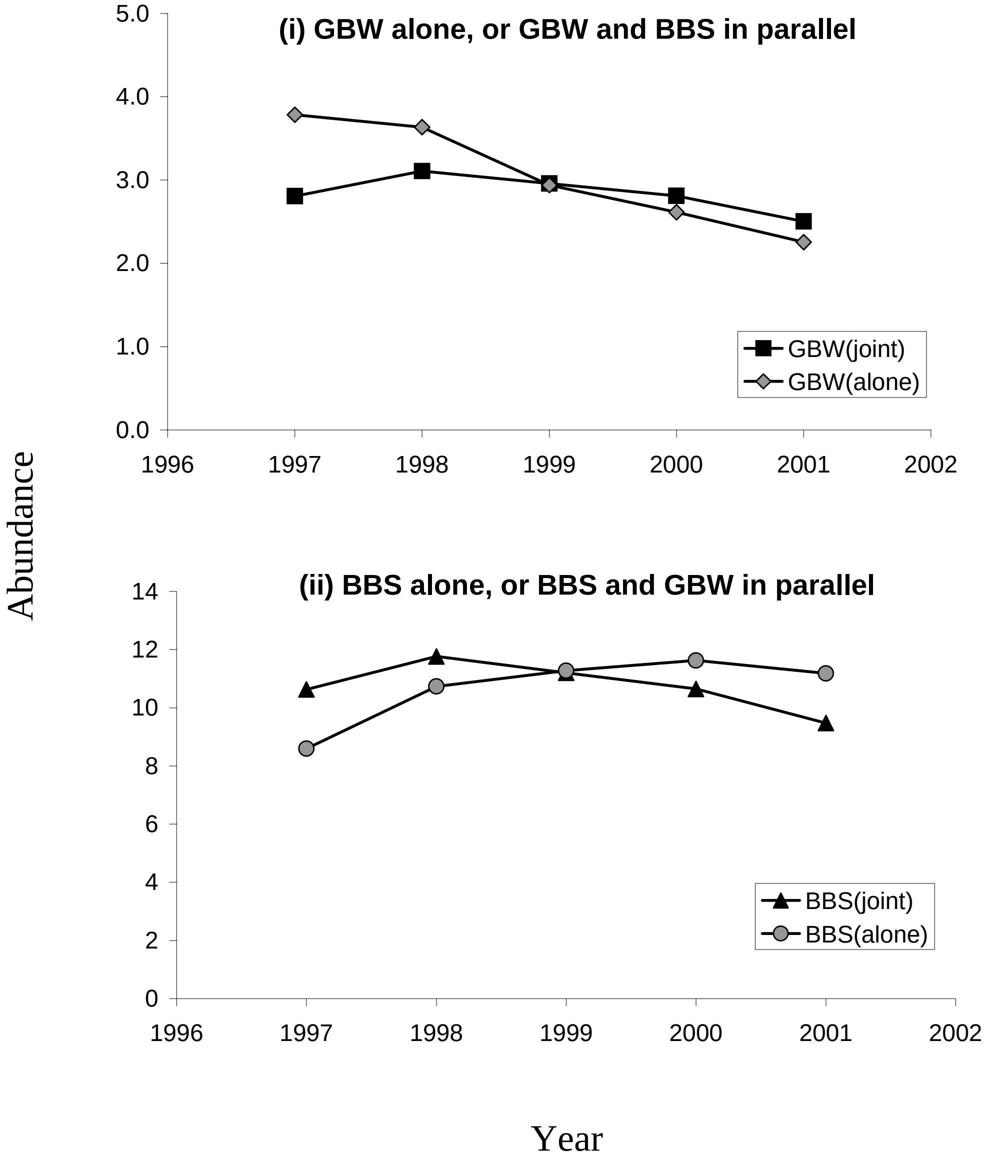


Figure 5.3 Continued.

Starling

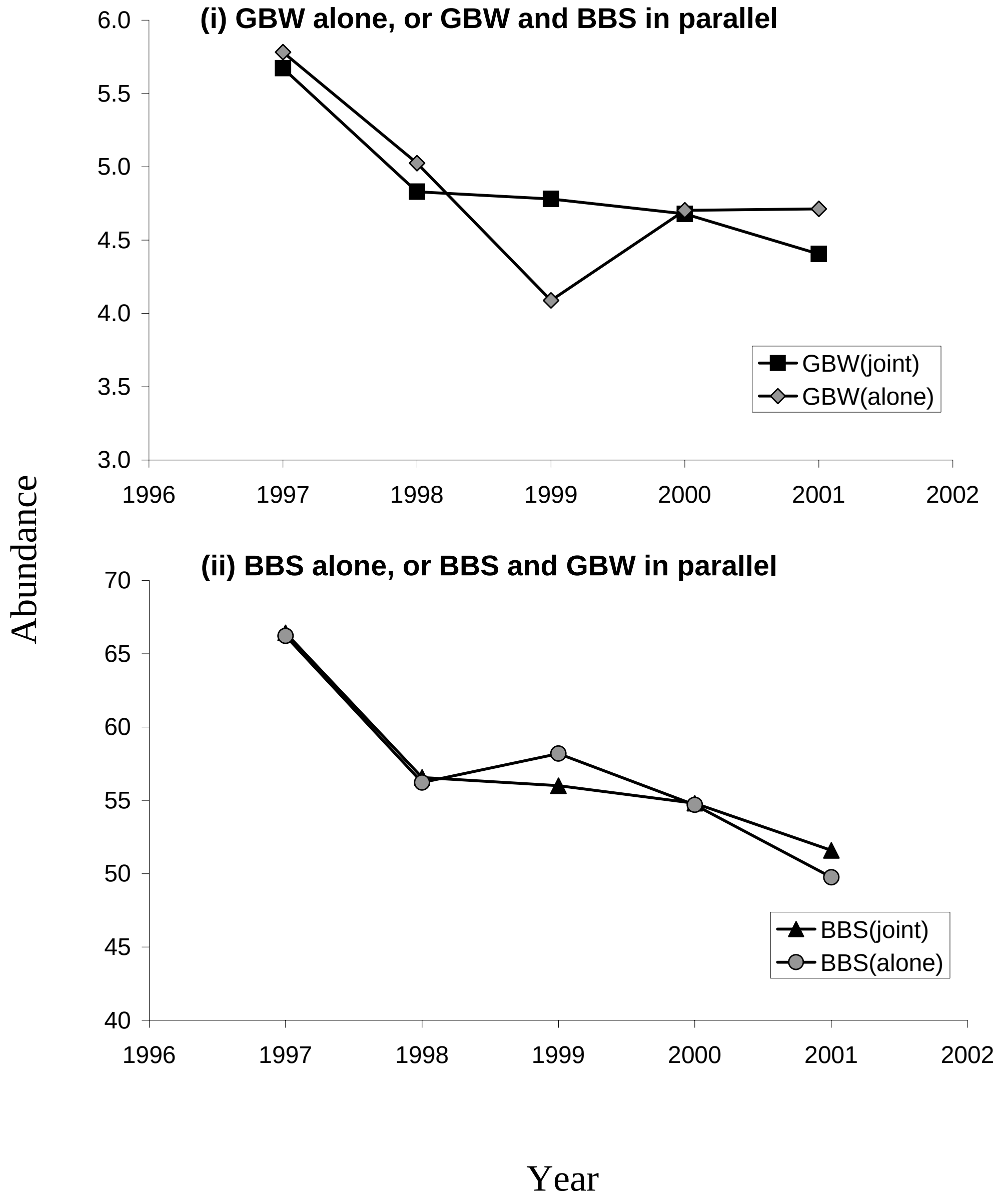


Figure 5.3 Continued.

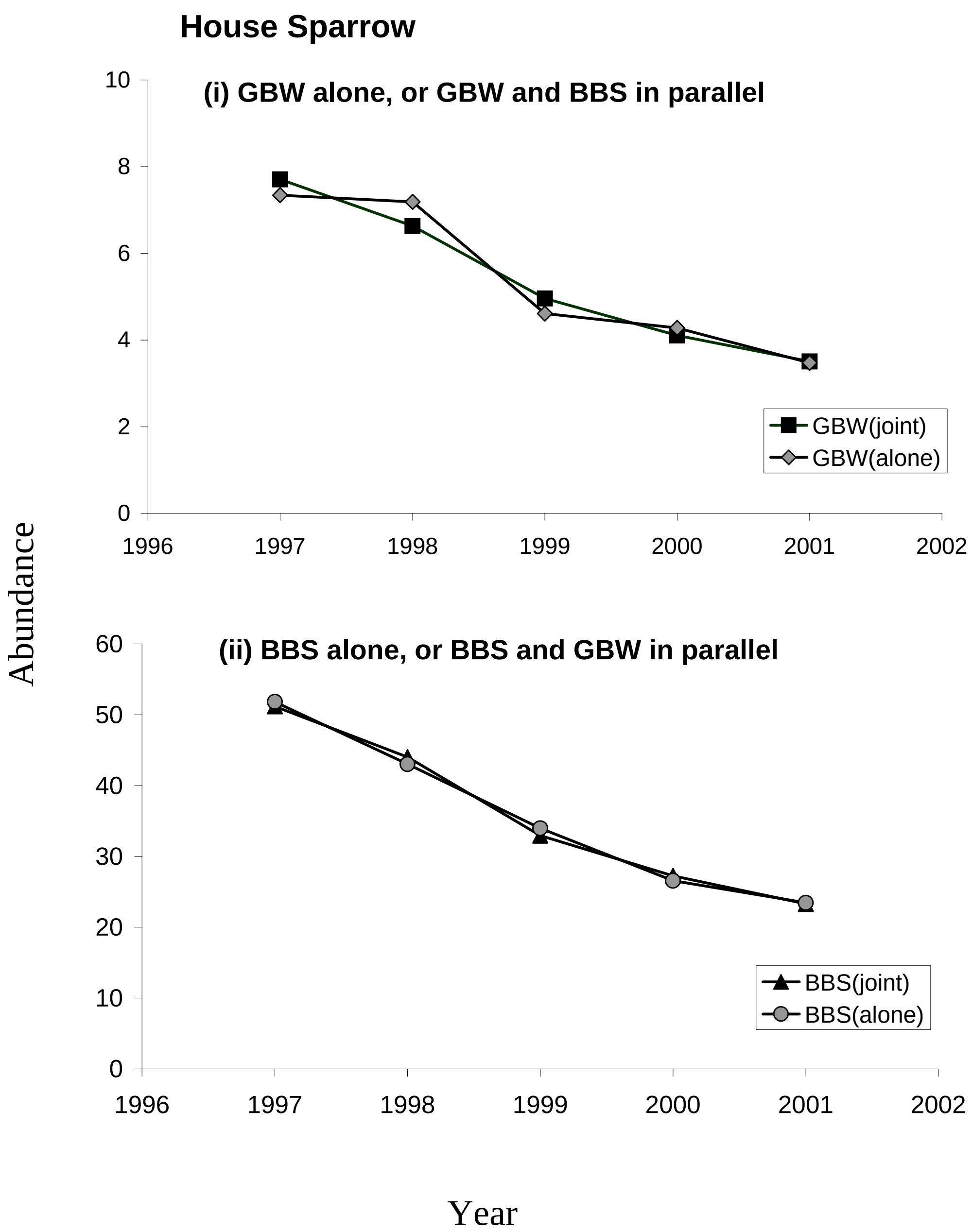


Figure 5.3 Continued.

Chaffinch

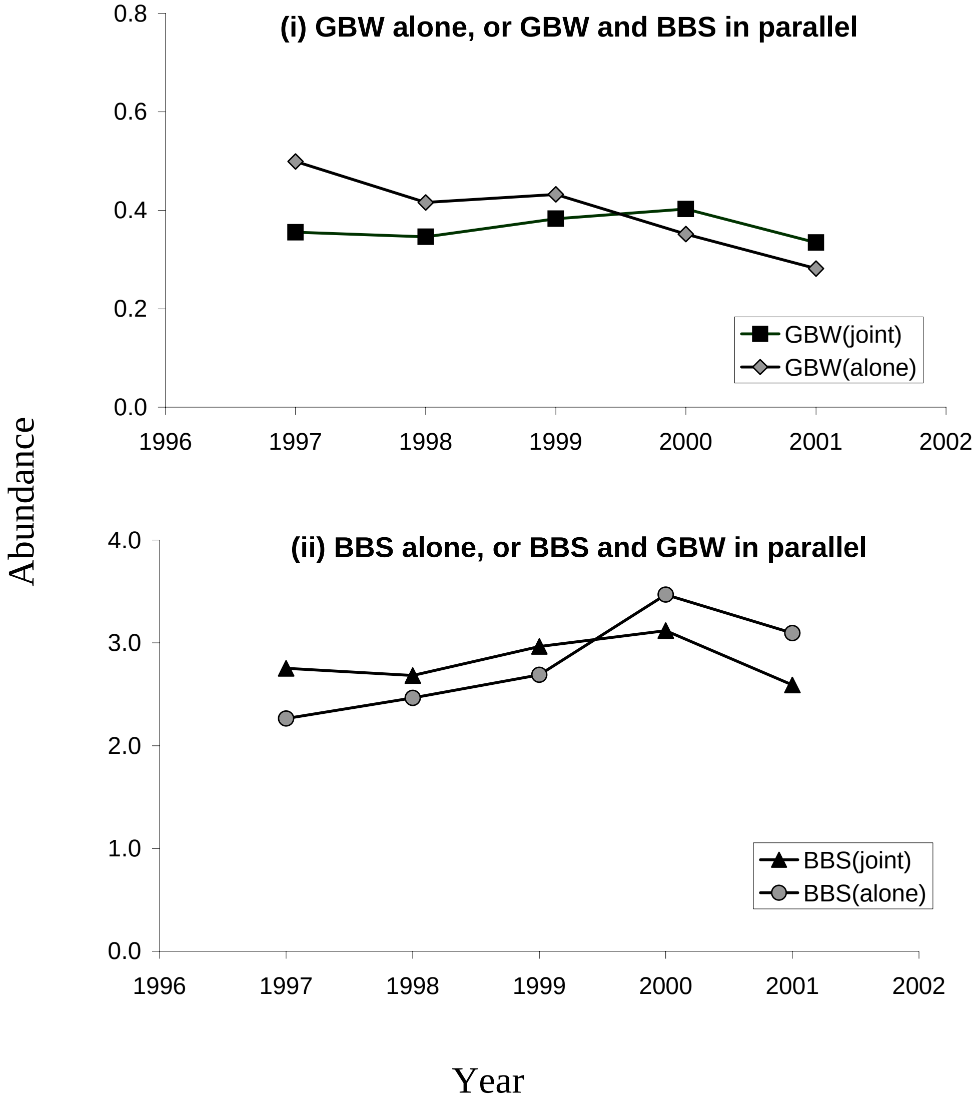


Figure 5.3 Continued.

Greenfinch

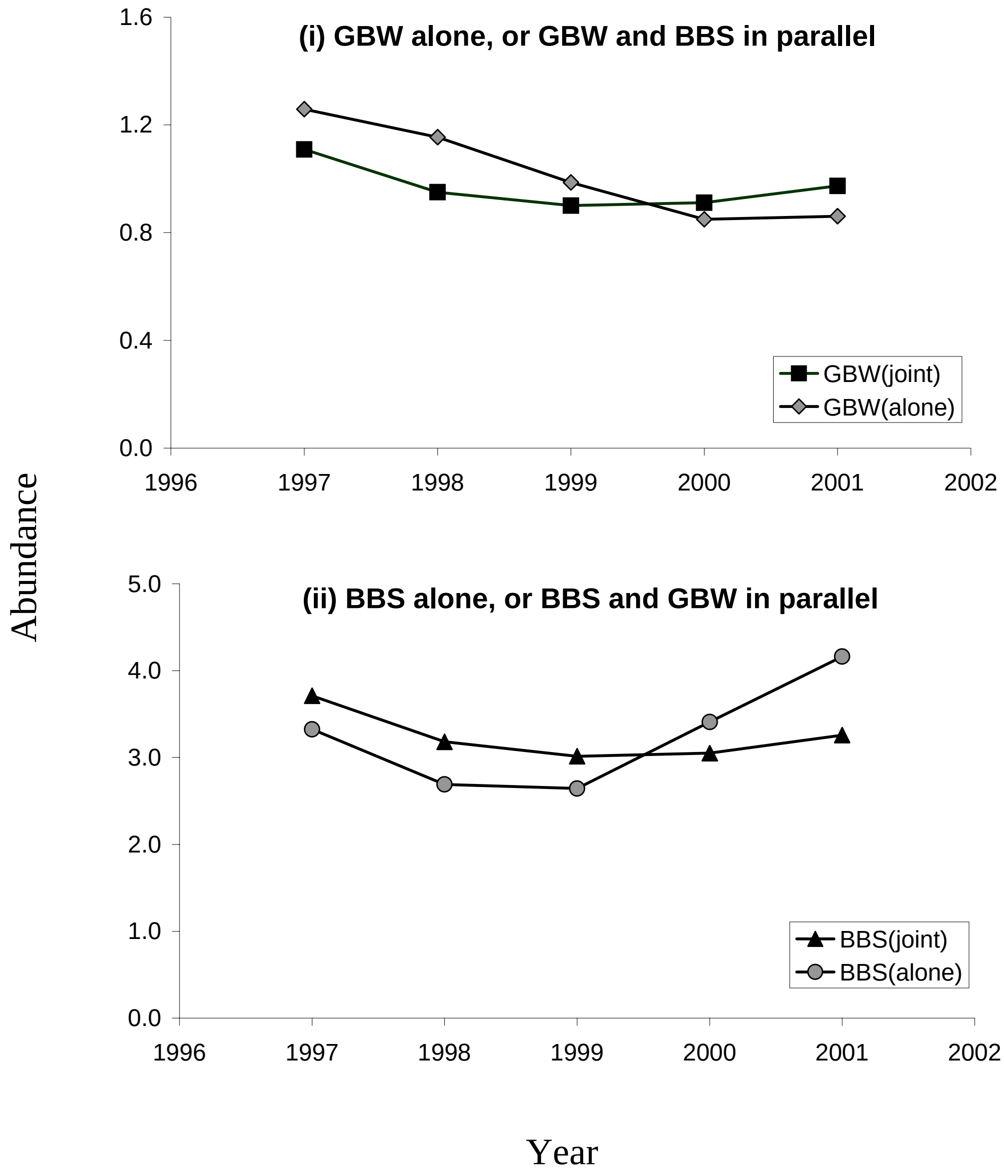


Figure 5.3 Continued.

CHAPTER 6 BIRDS IN GREATER LONDON: A COMPARISON OF SURVEYS

Dan Chamberlain

1. INTRODUCTION

There will be particular bias in any one bird survey methodology. Certain species will be recorded more frequently and other species will be under-recorded depending on the precise methodology and also the range of habitats covered. Building a complete picture of a total avifauna would be a very difficult undertaking, especially encompassing species that are difficult to detect (e.g. nocturnal species) or species that may have very particular habitat requirements. However, in an urban context, the data exist to compare diurnal terrestrial avifauna from different surveys.

2. AIMS

The aim of this chapter is to compare individual species' occurrence rates and densities (where possible) between three surveys: London Bird Project (LBP), Breeding Bird Survey (BBS) and Garden BirdWatch (GBW).

2.1 Statistical Analysis

The frequency of occurrence of species recorded in the London Bird Project (LBP), BBS and GBW was calculated by determining the number of times a given species was recorded (regardless of it's abundance) over all survey visits and dividing by the number of visits. For each BBS square there were only two visits (so if a given species was recorded on one of those visits, the frequency would be 0.5) and for each LBP site there were three visits each in spring/summer and winter. For GBW sites, there was potentially a survey every week of the year. For the following comparison, GBW data have been used only for those periods that coincide with the BBS and LBP (April – July and November – March). Differences in the rate of occurrence between surveys were assessed with a binomial logistic regression analysis using an events/trials model syntax. Furthermore, similarity indices were calculated (Magurran 1988) based both on species occurrence and on species density. This did not include GBW data as this was restricted to 41 species only.

3. RESULTS

When considering coverage from all three surveys, a large proportion of Greater London has bird data from at least one scheme. In total there were data from 617 individual 1-km squares (Fig. 6.1).

For the breeding season, a comparison of all three surveys was possible, although GBW is restricted to 41 species only. The recording rates of those 41 species, plus the 9 next commonest species recorded in the LBP are shown in Table 6.1. These are calculated using data from individual visits. All visits over all years in question were summed for each survey. The ratio of the number of times a species was present divided by the total number of visits was used as the analysis variable (note that this differs from previous occurrence data presented in Chapter 1 where occurrence over any visit rather than in each visit was used to calculate occurrence rates). In the majority of species, there was a significant difference in the probability of occurrence between surveys. No significant difference was detected between the three surveys for Feral Pigeon, Woodpigeon, Great Spotted Woodpecker, Mistle Thrush, Goldcrest, Marsh/Willow Tit, Long-tailed Tit, Magpie, Jay, Brambling, Chaffinch, Bullfinch and Reed Bunting. There was no significant difference detected between BBS and LBP for Kestrel, Mallard, Canada Goose, Moorhen, Grey Heron, Green Woodpecker, Stock Dove, Chiffchaff, Willow Warbler and Linnet (these species are not recorded on the standard paper recording form in GBW).

In many cases where a significant difference was detected, reporting rate was highest in BBS, illustrating the wider range of habitats and larger areas covered under this survey. This included common garden birds and birds that nest in buildings (Collared Dove, Dunnock, Blue Tit, Great Tit, Coal Tit, Jackdaw, Starling, House Sparrow and Greenfinch), but also a number of species that are more closely associated with farmland (Rook and Yellowhammer) or woodland (Marsh/Willow Tit, Siskin and Bullfinch). There were noticeably low frequencies in LBP for Collared Dove, Jackdaw, Coal Tit and House Sparrow. For the former three species, this may be associated with nesting habitat (buildings for the former two, conifers for the latter). The low proportion of House Sparrows, however, is likely to be a reflection of the wider decrease in this species. The dramatic declines in Hyde Park are well documented and these preliminary results do little to alter the view that London parks are no longer a prime habitat for this species.

A similar comparison between GBW and LBP was carried out for the winter period (Table 6.2) for 39 species (all GBW species except Black-headed Gull and Tawny Owl which were not covered adequately in LBP). Most species showed a significant difference in reporting rate. Species showing no significant difference were Sparrowhawk, Feral Pigeon, Great Spotted Woodpecker, Blackbird, Marsh/Willow Tit, Nuthatch, Magpie, Jay, Tree Sparrow, Brambling and Yellowhammer. Some notable differences were detected in Collared Dove, Coal Tit and House Sparrow where occurrence in gardens was much greater than that in green spaces. Conversely, occurrence of Carrion Crow, Song Thrush and Mistle Thrush was notably greater in green spaces.

Mean density was determined for BBS and LBP sites in the summer. For BBS data, the calculation was based on the summed count over the whole square divided by the transect area up to 100m distance from either side of the transect (i.e. 1000 x 100 x 2 m). The bird data were truncated at 100m for this calculation. For LBP, total bird numbers were divided by site area (280 sites had area data available). There was a big difference in density between surveys, with LBP densities far exceeding BBS densities in every species. There may be methodological reasons behind this. LBP is relatively intensive in that it covers the whole area of a given site whereas BBS uses line transect methodology. Detection probabilities may well differ between the two surveys. Even so, the counts for LBP were highly variable and there were many species where the difference was not significant or was significant at a relatively high P-value (Table 6.3). There is an implication therefore that large numbers (e.g. large flocks) may be more likely to encountered in parks, but overall there is a very large variation in bird numbers within London's green spaces. However, such are the differences that there is also likely to be an ecological reason underlying the differences: for many species green spaces are likely to hold very high densities compared to the surrounding countryside.

For BBS, distance sampling methodology can be applied when calculating densities which should to some extent minimise biases caused by decreasing detection with increasing distance away from the transect line (Buckland *et al*. 1994). This has been carried out for 9 species for four BBS habitat types (Chapter 3): rural parks, suburban parks, urban parks and all other habitats. The comparison with LBP densities is shown in Table 6.4. For 7 of the 9 species considered, densities are broadly comparable. Parks generally have higher densities than other habitats, although differences were not as extreme as in Table 6.3, illustrating the effect of adjusting for detectability. Only Blackbird and Feral Pigeon show much higher densities in the LBP data set compared to BBS park habitats. For Feral Pigeon, this may reflect a much greater proportion of sites from very urban green spaces where Feral Pigeons dominate. It is less clear why Blackbird should show such large differences.

Similarity indices (Magurran 1988) were determined for species presence (regardless of abundance) between LBP and BBS (this was not possible with GBW as the number of species is limited to 41). All indices used have a range between 0 (no common species) and 1 (identical species composition and occurrence). Jaccard's similarity index showed a value of 0.66. Similarity calculated using Sorenson's index was higher at 0.82. Clearly, the species recorded in both LBP and BBS were fairly similar, but Table 6.3 indicates that abundances vary between surveys for certain species. The Morista-Horn similarity index was calculated using individual species abundance which gave a value of 0.68, so even in terms of bird density, BHT and LBP are fairly similar.

4. DISCUSSION

Survey comparison - A key question in assessing these differences is whether they are due to ecological factors or whether they are due to differences in survey methodology. For some species where detection is easy biases are probably minimised. However, for more cryptic species, detection by sight only (as in GBW) may bias against species such as Wren and Song Thrush that are often detected by song and that showed relatively low occurrence in gardens. There were some intriguing differences between surveys in occurrence rates and abundance detected in certain species. Compared to LBP, in most cases, occurrence was higher in BBS. BBS covers a wider range of habitats than GBW or LBP and also in most cases, larger areas will be covered than in LBP. This may increase detection likelihood and therefore simple comparisons of presence/absence may be biased towards what is effectively a higher effort survey. Nevertheless, some differences were so striking that ecological differences may also have contributed. For some (generally scarce) species, LBP and GBW will not sample the primary habitats for woodland birds (Blackcap, Marsh/Willow Tit and Nuthatch) and farmland birds (Jackdaw, Yellowhammer and Reed Bunting). However, the much lower occurrence of generally widespread species in LBP such as Collared Dove, Coal Tit, House Sparrow and Goldfinch may indicate that urban green spaces are not the best habitat even within an urban context.

Occurrence rates were generally more similar between LBP and GBW. There were some notable differences. Most GBW participants provide bird food in their gardens (Chamberlain *et al*. 2004), and it is likely that most birds recorded will be visiting garden feeding stations. This may account for generally higher occurrence rates in GBW in species that commonly use feeders (e.g. Great Tit, Blue Tit, Coal Tit and House Sparrow). There were other species where occurrence rates were higher in LBP. In the breeding season, the higher rate of species such as Blackcap, Song Thrush and Long-tailed Tit may represent a better nesting habitat in green spaces than in gardens. Carrion Crow also had a much higher occurrence rate in green spaces than gardens which may be due to a preference for ground foraging in larger open areas.

A comparison of densities found that LBP density far exceeded that for BBS in virtually every species. However, when distance sampling was used and only BBS parkland habitat data were used, densities were broadly similar in most species. Two exceptions were Feral Pigeon and Blackbird where densities were much higher in LBP, even compared to urban parks. Given the general similarity across a range of species, it seems that this analysis provides better evidence of differences between densities in public green spaces and those in the wider countryside. The implications are that for Blackbird and Feral Pigeon, there was either a difference in the methods that provided a bias for these particular species, or LBP sites were biased towards these species in some way. For the other species, densities are fairly similar to other habitats within Greater London.

This chapter has presented a comparison of occurrence rates and densities of species in three different surveys which were analysed separately in Chapters 1 - 5. A problem with interpreting these data is that any ecological differences between the habitats covered in the surveys will be confounded by methodological differences. Nevertheless, the combined surveys give a good general picture of the bird community of gardens (GBW), public green spaces (LBP) and the wider countryside (BBS) within Greater London. It is suggested that any future comparison of surveys should use distance sampling when possible.

Species	GBW	LBP	BBS	Significance
Blackbird	0.91	0.95	1.00	***
Blue Tit	0.88	0.76	0.96	***
Woodpigeon	0.80	0.86	0.99	ns
Robin	0.73	0.71	0.98	*
Great Tit	0.72	0.52	0.94	***
House Sparrow	0.72	0.34	0.99	***
Starling	0.70	0.63	0.99	*
Magpie	0.64	0.65	0.95	ns
Collared Dove	0.52	0.20	0.74	***
Feral Pigeon	0.46	0.54	0.94	ns
Dunnock	0.45	0.34	0.88	***
Greenfinch	0.42	0.36	0.83	***
Carrion Crow	0.37	0.74	0.96	***
Chaffinch	0.26	0.36	0.68	ns
Jay	0.25	0.24	0.71	ns
Wren	0.24	0.66	0.93	***
Great Spotted Woodpecker	0.21	0.23	0.53	ns
Coal Tit	0.18	0.07	0.27	***
Song Thrush	0.12	0.32	0.78	***
Long-tailed Tit	0.10	0.22	0.52	ns
Goldfinch	0.07	0.11	0.61	**
Nuthatch	0.05	0.05	0.17	***
Sparrowhawk	0.03	0.04	0.11	***
Blackcap	0.03	0.29	0.63	***
Treesparrow	0.03	0.00	0.01	ns
Mistle Thrush	0.02	0.24	0.70	ns
Jackdaw	0.02	0.04	0.18	***
Goldcrest	0.02	0.11	0.30	ns
Rook	0.02	0.01	0.07	***
Bullfinch	0.01	0.02	0.12	ns
Siskin	0.01	0.00	0.00	***
Pied Wagtail	0.00	0.11	0.46	*
Marsh/Willow Tit	0.00	0.01	0.05	ns
Treecreeper	0.00	0.02	0.10	***
Fieldfare	0.00	0.00	0.01	ns
Yellowhammer	0.00	0.003	0.10	***
Reed Bunting	0.00	0.01	0.08	ns
Mallard	.	0.57	0	ns
Green Woodpecker	.	0.53	0.21	ns
Chiffchaff	.	0.47	0.2	ns
Moorhen	.	0.47	0.14	ns
Whitethroat	.	0.43	0.07	*

Table 6.1 Frequency of occurrence of 50 species recorded in three different surveys in Greater London in spring and summer 2002-03. GBW = Garden BirdWatch (400 sites), LBP = London Bird Project (299 sites), BBS = Breeding Birds Survey (83 sites). Occurrence is based on data per visit. Significance is based on binomial logistic regression, comparing probability of occurrence between surveys. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$, ns not significant. Species are given in order of occurrence in GBW and then in occurrence in LBP.

Species	GBW	LBP	BBS	Significance
Willow Warbler	.	0.37	0.04	ns
Linnet	.	0.33	0.04	ns
Kestrel	.	0.31	0.05	ns
Canada Goose	.	0.28	0.11	ns
Grey Heron	.	0.28	0.05	ns
Stock Dove	.	0.28	0.07	ns
Ring-necked Parakeet	.	0.14	0.09	**

Table 6.1 Continued.

Species	GBW	BHT	Significance
Blue Tit	0.94	0.84	***
Blackbird	0.90	0.92	ns
Robin	0.86	0.75	***
Great Tit	0.79	0.65	***
Woodpigeon	0.72	0.86	**
Starling	0.67	0.47	***
Magpie	0.64	0.70	ns
House Sparrow	0.63	0.25	***
Dunnock	0.51	0.27	***
Collared Dove	0.49	0.19	***
Chaffinch	0.43	0.39	*
Feral Pigeon	0.43	0.51	ns
Greenfinch	0.38	0.33	***
Carrion Crow	0.36	0.83	***
Jay	0.31	0.31	ns
Coal Tit	0.29	0.09	***
Wren	0.27	0.47	*
Great Spotted Woodpecker	0.21	0.24	ns
Long-tailed Tit	0.16	0.33	**
Song Thrush	0.14	0.29	**
Goldfinch	0.09	0.14	**
Nuthatch	0.07	0.07	ns
Redwing	0.05	0.26	***
Blackcap	0.04	0.01	***
Goldcrest	0.04	0.18	**
Sparrowhawk	0.04	0.04	ns
Mistle Thrush	0.03	0.25	***
Siskin	0.02	0.03	***
Tree Sparrow	0.02	0.00	ns
Rook	0.02	0.01	***
Jackdaw	0.02	0.04	***
Pied Wagtail	0.01	0.18	***
Fieldfare	0.01	0.05	***
Bullfinch	0.01	0.03	***
Marsh/Willow Tit	0.01	0.01	ns
Treecreeper	0.00	0.04	***
Brambling	0.003	0.001	ns
Yellowhammer	0.001	0.002	ns
Reed Bunting	0.00	0.01	***

Table 6.2 Frequency of occurrence of 39 species recorded in two different surveys in Greater London in winter 2002/03-03/04. GBW = Garden BirdWatch (396 sites), LBP = London Bird Project (285 sites). Significance is based on binomial logistic regression, comparing probability of occurrence between surveys. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$, ns not significant. Species are given in order of occurrence in GBW.

Species	LBP		BBS		T-test
	Mean	SD	Mean	SD	
Feral Pigeon	11.52	36.15	4.57	7.92	***
Blackbird	3.99	6.59	2.76	2.28	**
Starling	3.39	6.74	7.03	8.73	***
Woodpigeon	2.30	3.58	2.69	2.44	ns
Blue Tit	1.61	2.77	1.70	1.55	ns
Carrion Crow	1.31	4.45	1.43	1.39	ns
Robin	0.81	1.31	1.05	1.10	ns
House Sparrow	0.79	3.78	6.12	9.26	***
Wren	0.75	1.56	1.08	1.12	*
Magpie	0.71	1.31	1.11	0.91	**
Great Tit	0.60	1.44	0.67	0.66	ns
Greenfinch	0.33	0.84	0.43	0.60	ns
Long-tailed Tit	0.24	1.43	0.11	0.16	ns
Chaffinch	0.21	0.59	0.33	0.66	ns
Dunnock	0.21	0.61	0.45	0.61	**
Blackcap	0.17	0.59	0.17	0.29	ns
Collared Dove	0.15	0.62	0.63	1.01	***
Mistle Thrush	0.14	0.43	0.15	0.23	ns
Goldfinch	0.10	0.52	0.13	0.24	ns
Jay	0.09	0.28	0.11	0.15	ns
Great Spotted Woodpecker	0.08	0.25	0.07	0.11	ns
Pied Wagtail	0.08	0.35	0.05	0.08	ns
Ring-necked Parakeet	0.08	0.44	0.01	0.02	**
Goldcrest	0.05	0.21	0.04	0.10	ns
Song Thrush	0.05	0.25	0.28	0.39	***
Coal Tit	0.02	0.11	0.04	0.11	ns
Rook	0.02	0.22	0.02	0.12	ns
Bullfinch	0.01	0.08	0.01	0.04	ns
Jackdaw	0.01	0.09	0.04	0.19	ns
Nuthatch	0.01	0.10	0.03	0.12	ns
Sparrowhawk	0.01	0.04	0.00	0.01	ns
Marsh/Willow Tit	0.00	0.06	0.00	0.01	ns
Reed Bunting	0.00	0.03	0.01	0.05	ns
Treecreeper	0.00	0.02	0.00	0.01	ns
Yellowhammer	0.00	0.02	0.03	0.13	ns

Table 6.3 Estimated densities (birds/ha) of selected species in London's green spaces determined from the London bird project (LBP) and from BBS squares in Greater London in 2002-03. Species are listed in order of density according to LBP. Unpaired t-tests were used to compare densities within species. LBP n = 280, BBS n = 83. * $P < 0.05$, ** $P < 0.01$, *** $P < 0.001$, ns not significant.

Species	LBP	RPRK	SPRK	UPRK	OTHER
Feral Pigeon	11.52	0	1.217	3.084	1.434
Woodpigeon	2.30	1.058	1.252	7.556	0.716
Wren	0.75	1.118	4.094	3.003	0.488
Robin	0.81	2.096	3.561	3.746	0.551
Blackbird	3.99	1.123	1.055	1.256	0.787
Blue Tit	1.61	2.111	1.098	0.950	0.861
Carrion Crow	1.31	0.555	0.227	0.932	0.294
Starling	3..39	1.230	3.035	7.364	1.786
House Sparrow	0.79	0.404	0.940	0.986	0.877

Table 6.4 A comparison of bird densities (birds/ha) from LBP data and from BBS data estimated from Distance sampling (Buckland *et al*. 1994). Estimates have been derived for four separate BBS habitat types, Rural parks (RPRK), suburban parks (SPRK), urban parks (UPRK) and all other habitats (OTHER).

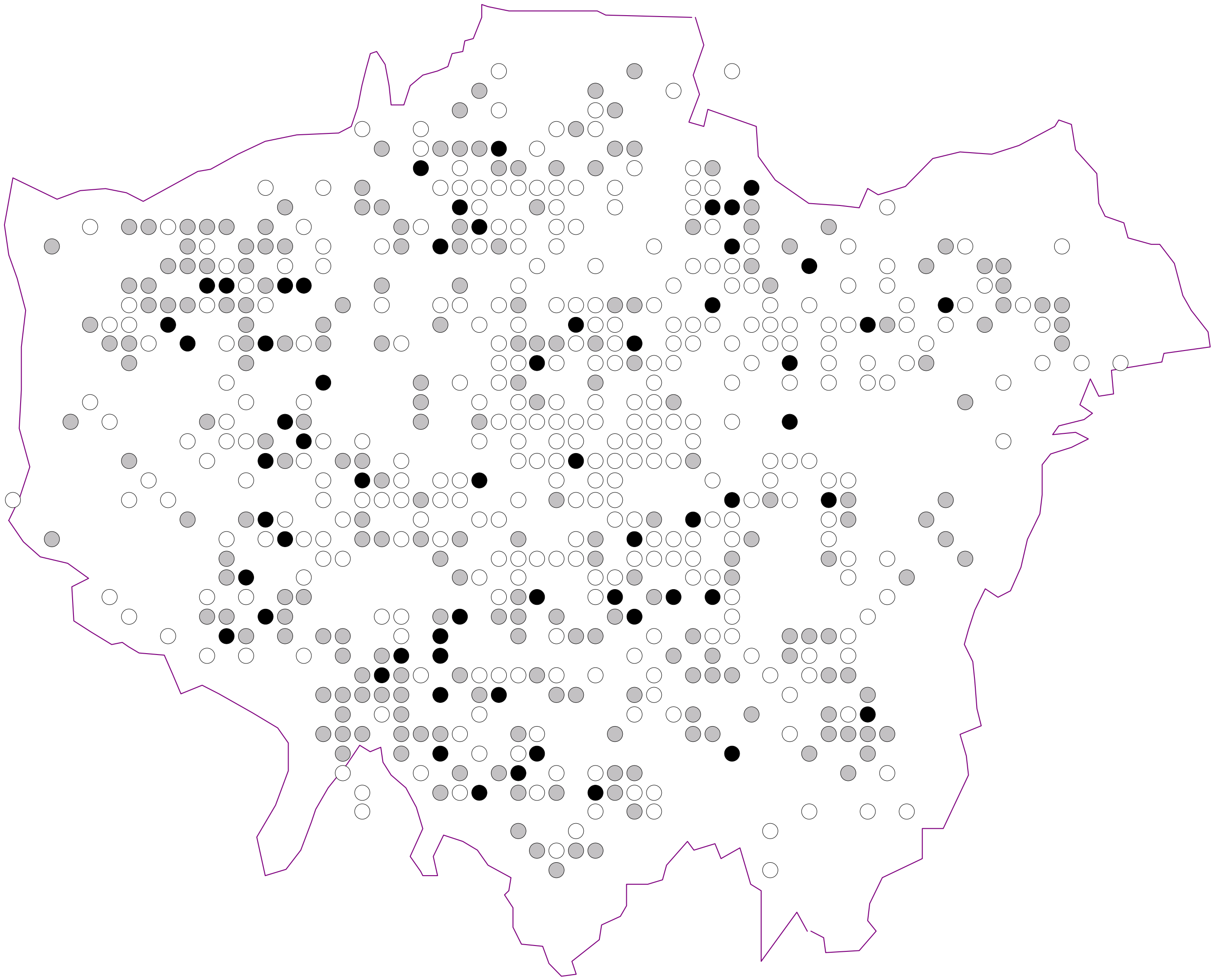


Figure 6.1 The distribution of bird survey data from either London Bird Project, BBS or GBW in Greater London. White = 1 survey, grey = 2 surveys, black = 3 surveys. N = 617.

CHAPTER 7 AN OVERVIEW OF THE LONDON BIRD PROJECT

Dan Chamberlain, Graham Appleton

What we know – This project has covered a range of public green spaces, many of which will not have been surveyed for birds previously and would not have been considered to be of importance to local biodiversity. We now have knowledge of the range of species that occur in London's green spaces in winter and summer, including those that are likely to be breeding. Parallel analyses of pre-existing data sets have enabled green spaces to be placed in a wider context of a range of habitats (Breeding Bird Survey (BBS) - Chapter 3) and compared to a specific habitat, gardens (Garden BirdWatch (GBW) – Chapter 4). Knowledge of the latter is clearly very important – BBS data suggest that gardens is the most species-rich habitat in Greater London and the only habitat to hold more species than parks. The value of gardens as wildlife habitat has been widely acknowledged elsewhere (e.g. Cannon 1999, Ansell *et al*. 2001, Gaston *et al*. 2004, Bland *et al*. 2004) and as such they should be treated as an important biodiversity resource. Yet the garden habitat is not without threats. 'Infilling' of gardens with new properties is becoming more common within Greater London (D. Dawson pers. comm.). This is a particular problem where gardens of large Victorian terraced houses are being built upon. Usage of gardens is also important, for instance where off-road parking replaces hedges, lawns and shrub borders or where lawns are abandoned in favour of paving or decking. Gardens provide relatively large areas of high quality habitat for many species and conservationists should monitor these kinds of developments.

One of the key aspects of the London Bird Project is the linkage of bird data to habitat data within green spaces and the production of management recommendations that reflect the relationships which have been detected. The habitats that were identified consistently as of value to birds were deciduous trees, mown grass and deciduous bushes. GBW data also suggested that trees and 'hedges' (essentially bushes at the garden boundary) were linked to the occurrence of many species in gardens. For green spaces, clearance of bushes is potentially a major issue given the (perceived) safety risk in parks with a high cover of shrubs. Small passerines, and House Sparrows in particular, favour deciduous bushes and it is possible that bush clearance may have been in part responsible for the fall in House Sparrow numbers and for local extinctions. It is suggested that, if bushes really need to be cleared from green spaces, the removal should be offset by additional planting in less contentious locations (e.g. away from paths, at the site boundary). There was little evidence that coniferous bushes were much used by birds (with two exceptions, Goldcrest and Coal Tit) and deciduous bushes should be planted in preference.

What we don't know – The range of sites covered in the green spaces survey was restricted to relatively small sites. Sites not covered (or covered in a very limited way) included larger parks, non-free access sites (e.g. London Zoo, Hampton Court), nature reserves and farmland and private gardens, all of which may hold important numbers of birds. Farmland and gardens were included in BBS and GBW data sets, so we have some knowledge of these habitats within Greater London. For several of the other sites, however, the bird communities are reasonably well described (Hewlett 2002 and references therein). Less is known about how these different habitats interact. An indication can be obtained by looking at seasonal changes over time in different habitats. For example, comparing seasonal shifts in parks and in gardens for House Sparrow (a red-listed species of conservation concern) reveals a decrease in parks in the winter but an increase in gardens, suggesting a habitat shift. However, we cannot know how individual birds use habitats over time. In the House Sparrow example, are these the same birds moving from one habitat to another, or are birds dispersing into gardens from the wider countryside? How connectivity of green spaces (e.g. effects of isolation and the role of habitat corridors) affects bird movements is also an important issue. Ringing and radio-tracking studies may be the only way to study such questions in more detail.

This project has identified a number of key research areas that would further enhance our knowledge of the biodiversity value of green spaces and of the urban environment in general. Bushes are clearly important to a wide range of species, but to maximise their value more knowledge is needed of precise management (such as cutting regimes) and also of the structural features that are of most importance to birds looking for nesting habitat, foraging habitat and general cover. A study of the placement of bushes (adjacent habitats, value as understorey or without trees) would also be of value. Furthermore, an assessment of the resources available to birds in native versus non-native trees and bushes would add value to any planned planting initiatives. This study has considered counts and presence of bird species. Very little is known about the ecology of birds in green spaces or, for that matter, gardens for most species. Intensive studies of reproductive performance, feeding ecology and movements would greatly enhance our ability to make recommendations that would have significant impacts on urban bird populations.

It is important to note that the recommendations made in this report are primarily concerned with increasing bird numbers (of individuals and species) and that no direct information has been collected for other taxa (although there were some data on invertebrates collected from a small number of sites as part of an undergraduate project, the results of which are presented in Appendix V). The position of birds at the top of the urban food chain means that they are often good indicators of the health of the wider environment (Furness & Greenwood 1993), but it may be possible to enhance bird diversity without equivalent benefits to wider biodiversity. It would therefore be extremely valuable to see green spaces surveyed for a wide range of other taxa, as has been attempted for urban gardens in Sheffield (Gaston *et al*. 2004).

Density as an indicator of habitat quality **-** Throughout this study, there has been an implicit assumption that higher bird density reflects better habitat quality. For non-breeding birds, density is likely to be a good indicator of habitat quality for the majority of species that aren't territorial in the autumn and winter. However, in the breeding season, density may be less indicative of habitat quality if there is unequal exploitation of resources within species (Fretwell 1972). In such circumstances, social dominance factors may affect the distribution of animals, and a large proportion of a given population may be non-breeders or unsuccessful breeders occupying less favoured habitats (Van Horne 1983, Vickery *et al*. 1992), especially where territorial behaviour is affecting breeding density. However, even within species which show marked differences in dominance between individuals, higher nesting density is typically observed in the better quality habitat, e.g. Great Tit (Krebs 1971) and Blackbird (Hatchwell *et al*. 1996). We can thoroughly test this assumption only with intensive studies that consider breeding densities, territorial behaviour, reproductive success and annual survival.

Wider benefits – In addition to furthering our knowledge of birds in Greater London, a secondary goal was to engage and educate the wider public. Promotion of the green spaces survey has resulted in over 225 people participating (an exact figure is difficult to arrive at as many people did surveys in pairs or through groups such as the East London Birders Forum), many of whom were new to volunteer-based bird survey work. Furthermore, promotion of GBW alongside the green spaces survey has seen membership increase from around 400 to 1031 participants in Greater London. It is hoped that these people have experienced an increased awareness of bird conservation issues, a better understanding of ecology, and an appreciation of the value of local green spaces and the threats they may face. We were delighted when 150 people joined BTO staff in November 2004 to look at the results of the survey work and we hope that an appreciation of the value of green spaces will reach a wider audience through the publication of a management guidelines leaflet and attendant publicity.

The data collected in this project represent a resource for London. Summary species lists will be available for interested parties, including the borough ecologists and site managers who hopefully will be able to use the data to enhance or reinforce their current management strategies. The data also represent a good baseline for any repeat surveys in the future. This could be very important for monitoring populations of species that appear to be rapidly changing in Greater London (e.g. Ring-necked Parakeet, Spotted Flycatcher, House Sparrow), but also to assess any impacts that novel management techniques or wider environmental changes may have made to bird populations. It should also be remembered that people are a resource too. Hopefully this project has increased the number of people willing to be involved in conserving and enhancing biodiversity within Greater London in the future.

Acknowledgements

First and foremost, we are extremely grateful to the many volunteers who devoted a great deal of their free time and effort to contribute to this project. A full list of participants is included below. We would also like to extend our gratitude to the volunteer observers who contribute to the Breeding Bird Survey and the BTO/CJ Garden BirdWatch within London.

The London Bird Project would not have been possible without funding from the Bridge House Trust, the grant-giving arm of the Corporation of London. We are extremely grateful for their support and would in particular like to acknowledge Stuart Goshawk's input and encouragement. Further funding was provided by the Nuffield Foundation, through a student bursary to Michelle Stuckey of the University of East Anglia (project report is included as an Appendix), and by the BTO/CJ Garden BirdWatch Scheme. Dan Chamberlain received a grant to study bird use of supplementary food in gardens from the Leverhulme Foundation for the period 2001-2003, which enabled him to develop many of the ideas used in the project.

Staff from the Greater London Authority, particularly Dave Dawson, Jan Hewlett and John Archer were extremely supportive throughout the project and gave advice on site selection, survey methods, analysis and interpretation. We are most grateful for their input. Derek Coleman, the BTO's voluntary Regional Representative for London, was very helpful in the initial stages of the project, particularly in terms of volunteer recruitment, and his continued advice was much appreciated. Rob Fuller (BTO) provided the inspiration for this project and we are grateful for his enthusiasm and input at the project planning stage. Andrew Cannon (then of BTO) also provided advice and support on GBW in the early stages of the project. Thanks also to Nicki Read, Rebecca Cranston and Clare Clark who provided much needed support in production of data files and reports. Some extremely useful feedback was received from groups visited by Su Gough of the BTO during the course of this project and we would like to thank The East London Birders Forum, The London Natural History Society and RSPB London members' groups for their enthusiastic support.

Remotely-sensed land cover data was derived from the CS2000 data base provided by the Centre for Ecology and Hydrology.

The volunteer observers who took part in the green spaces survey were: Alan Gray, Albert Watson, Alex Rafinski, Amanda Harper, Andrew Fleming, Andrew Henderson, Andrew Lewis, Andrew Osborne, Andy Barnes, Andy Pepper, Angela Dougall, Angela Linell, Ann Feltham, Anne Ackroyd, Anne Charvet, Annette Warrick, Ashley Irons, Barbara Read, Barry Cheal, Ben Mullen, Beverley Wilson, Bill Garrett, Bob Dawson, Bob Fry, Brendan Spurner, Brian Dawton, Brian Leach, Carey Turner, Carol Rawlings, Caroline Benwell, Caroline Lee, Catherine Schade, Catherine Schmitt, Chris Chapman, Chris White, Christine Hall, Christopher Langsdon, Cindy Blaney, Claire Trocme, Colin Conroy, Colin Hedderwick, Colm O'Flynn, D S Broadley, David Bland, David Hamilton, David Harrison, David Kreeger, David Montier, David Morrison, Derek Coleman, Derek Robinson, Derek Warren, Diana Housley, Diane Nair, Dr Ruth Boff, Duncan Stubbs, Durlabh Singh, Dusty Gedge, Members of ELBF, Eleanor Edwards, Elizabeth Jerden, Elizabeth Stainthorpe, Ellen Martin, Ernie Thomasom, Faith Keymer, Fay Broom, Flo Harrison, G Davies, Gary Macguire, Gavin Hawgood, Gay Carr, Geoffrey Munns, George Dunnion, George Greaves, Gill Beadell, Grant Hazlehurst, Heather Gerard, Helen Baker, Helen Bantock, Henriette Monteiro, Hilary King, Hilary Whittle, Howard Davies, Howard Vaughan, Iain Green, Ian Baker, Ian Shaw, Ian Woodward, Irene Welton, Jan Sellars, Jane D'Souza, Jane Graydon, Jane Shemilt, Jane Sugarman, Janette Bowden, Jean Clarke, Jean Fanning, Jean Gundersen, Jean Lukins, Jennifer Partridge, Jenny Fulford, Jenny Watson, Jill Aldred, Jill Tritton, Jim Flanagan, Jo Sergeant, Jo Underhill, Joan & Bob Anderson, Joanna Watts, John Abbott, John Birkett, John Eason, John Flynn, John Jervis, John Picton, John T Turner, John Turner, John White, Joyce Cobb, Joyce Staines, Julia Honeybone, Julian Allen, Karen Byers, Karolena & Iain Fenn, Kathryn Bull, Kay Ellam, Keir Mottram, Keith Chambers, Keith Chaperlin, Keith Martin, Keith Pell, Ken Palmer, Klara Hegedus, Lee Killington, Lee Wilton, Lorraine Kirk, Lynne Batchelor, Lynne Eva, Malcolm Ridler, Marc Carlton, Mariana Lindberg, Mark Russell,

Martin Grounds, Martin Kirbey, Michael J Spicer (Mike), Michael McCarthy, Michael Rank, Mike Lawrence, Miss G J Smith, Mr & Mrs Franklin, Mr D E Roberts, Mr Dobbs, Mr Francis Lane, Mr Irvine, Mr Trott, Mrs J Robertson, Mrs P Isaaks, Mrs Ruth Bray, Nardine Montague-Gibson, Natalia Semmes, Natalie Schorbon, Neil Andersen, Neil Rowland, Nicholas Meade, Nick Hobbs, Nicola Overington, Nigel Willits, Owen Richards, P J Casselton, Paddy Muramura, Patrick Keatley, Paul Taylor, Paul Tregenza, Paul Tublin, Paul White, Pete Mantle, Peter Burrows-Smith, Peter Hopkins, Peter Jones, Peter Whitcomb, Phil Davies, Piers Lawson, Rachel Patterson, Ralph Sellers, Raymond Garrett, Richard John, Richard Middleton, Rob Bacon, Rob Humphreys, Rob Tomalin, Robin Blades, Robin Morden, Robin Young, Roger Taylor, Ron Kettle, Rosalind Sim, Roslyn Knight, Roy Lockett, Sally Ingils, Sarah Barker, Sarah Bland, Sean Huggins, Sheila Mason, Simon Braidman, Stephen Harris, Stephen Lowe, Stuart Forbes, Sue Penny, Susanne Lawrence, Sybil Chick, Terry Bates, Timothy Tomkins, Tina Gray, Tom Burchell, Tom Cahalane, Tom Stone, Tony Banks, Tony Blake, Tony Gard, Tony Marland, Tony Parsons, Tricia Shaw, Victoria Hands, Wendy Simons, William Mathieson and William Whitbread.

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APPENDIX I Scientific names of all species mentioned in the text.

1. Birds

Bar-headed Goose	*Anser indicus*
Barnacle Goose	*Branta leucopsis*
Black Redstart	*Phoenicurus ochrurus*
Blackbird	*Turdus merula*
Blackcap	*Sylvia atricapilla*
Black-headed Gull	*Larus ridibundus*
Black-necked Grebe	*Podiceps nigricollis*
Blue Tit	*Parus caeruleus*
Brambling	*Fringilla montifringilla*
Bullfinch	*Pyrrhula pyrrhula*
Canada Goose	*Branta Canadensis*
Carrion Crow	*Corvus corone*
Cetti’s Warbler	*Cettia cetti*
Chaffinch	*Fringilla coelebs*
Chiffchaff	*Phylloscopus collybita*
Coal Tit	*Parus ater*
Collared Dove	*Streptopelia decaocto*
Common Gull	*Larus canus*
Common Sandpiper	*Actitis hypoleucos*
Common Tern	*Sterna hirundo*
Coot	*Fulica atra*
Cormorant	*Phalacrocorax carbo*
Corn Bunting	*Miliaria calandra*
Cuckoo	*Cuculus canorus*
Dunnock	*Prunella modularis*
Feral pigeon	*Columba livia*
Feral/hybrid Goose	*Anser sp*
Feral/hybrid mallard type	*Anas sp.*
Fieldfare	*Turdus pilaris*
Firecrest	*Regulus ignicappillus*
Gadwall	*Anas strepera*
Garden Warbler	*Sylvia borin*
Goldcrest	*Regulus regulus*
Goldfinch	*Carduelis carduelis*
Goshawk	*Accipiter gentilis*
Grasshopper Warbler	*Locustella naevia*
Great Black-backed Gull	*Larus marinus*
Great Crested Grebe	*Podiceps cristatus*
Great Spotted Woodpecker	*Dendrocopos major*
Great Tit	*Parus major*
Green Woodpecker	*Picus viridis*
Greenfinch	*Carduelis chloris*
Grey Heron	*Ardea cinerea*
Grey Wagtail	*Motacilla cinerea*
Greylag Goose	*Anser anser*
Hawaian Goose	*Branta sandvicensis*
Herring Gull	*Larus argentatus*
House Martin	*Delichon urbica*
House Sparrow	*Passer domesticus*
Jackdaw	*Corvus monedula*
Jay	*Garrulus glandarius*

Kestrel	*Falco tinnunculus*
Kingfisher	*Alcedo atthis*
Lapwing	*Vanellus vanellus*
Lesser Black-backed Gull	*Larus fuscus*
Lesser Redpoll	*Carduelis cabaret*
Lesser Spotted Woodpecker	*Dendrocopos minor*
Lesser Whitethroat	*Sylvia curruca*
Linnet	*Carduelis cannabina*
Little Grebe	*Tachybaptus ruficollis*
Little Owl	*Athene noctua*
Long-tailed Tit	*Aegithalos caudatus*
Magpie	*Pica pica*
Mallard	*Anas platyrhynchos*
Mandarin	*Aix galericulata*
Marbled Duck	*Marmaronetta angustirostris*
Marsh Tit	*Parus palustris*
Meadow Pipit	*Anthus pratensis*
Mistle Thrush	*Turdus viscivorus*
Moorhen	*Gallinula chloropus*
Mute Swan	*Cygnus olor*
Nightingale	*Luscinia megarhynchos*
Nuthatch	*Sitta europaea*
Peregrine	*Falco peregrinus*
Pheasant	*Phasianus colchicus*
Pied Wagtail	*Motacilla alba*
Pochard	*Aythya ferina*
Red-legged Partridge	*Alectoris rufa*
Redshank	*Tringa totanus*
Redwing	*Turdus iliacus*
Reed Bunting	*Emberiza schoeniclus*
Reed Warbler	*Acrocephalus scirpaceus*
Ringed Plover	*Charadrius hiaticula*
Ring-necked Parakeet	*Psittacula krameri*
Robin	*Erithacus rubecula*
Rook	*Corvus frugilegus*
Ruddy Duck	*Oxyura jamaicensis*
Sand Martin	*Riparia riparia*
Sedge Warbler	*Acrocephalus schoenobaenus*
Shelduck	*Tadorna tadorna*
Shoveler	*Anas clypeata*
Siskin	*Carduelis spinus*
Skylark	*Alauda arvensis*
Song Thrush	*Turdus philomelos*
Sparrowhawk	*Accipiter nisus*
Spotted Flycatcher	*Muscicapa striata*
Snipe	*Gallinago gallinago*
Starling	*Sturnus vulgaris*
Stock Dove	*Columba oenas*
Stonechat	*Saxicola torquata*
Swallow	*Hirundo rustica*
Swift	*Apus apus*
Tawny Owl	*Strix aluco*
Teal	*Anas crecca*
Tree Pipit	*Anthus trivialis*
Tree Sparrow	*Passer montanus*

Treecreeper	*Certhia familiaris*
Tufted Duck	*Aythya fuligula*
Turtle Dove	*Streptopelia turtur*
Waxwing	*Bombycilla garrulus*
Wheatear	*Oenanthe oenanthe*
Whitethroat	*Sylvia communis*
Whooper Swan	*Cygnus cygnus*
Wigeon	*Anas penelope*
Willow Tit	*Parus montanus*
Willow Warbler	*Phylloscopus trochilus*
Woodlark	*Lullula arborea*
Woodpigeon	*Columba palumbus*
Wren	*Troglodytes troglodytes*
Yellow Wagtail	*Motacilla flava*
Yellowhammer	*Emberiza citrinella*

2. Mammals

Grey Squirrel	Sciurus carolinensis
Red Squirrel	S. vulgaris

APPENDIX II Proportion of sites in which all species recorded were observed and the mean density (and confidence limits) of those species per site in summer and winter (means were taken over the two years of the survey). N (for occurrence) = 299 sites in summer and 285 in winter. N (for density) = 279 sites in summer and 269 in winter (site area was missing in some cases).

(a) Summer

Spp	Occurrence	Density	LCL	UCL
Blackbird	0.98	1.10	1.00	1.20
Woodpigeon	0.95	0.89	0.81	0.98
Blue Tit	0.92	0.74	0.66	0.81
Carrion Crow	0.89	0.62	0.54	0.69
Robin	0.87	0.52	0.46	0.58
Magpie	0.84	0.46	0.40	0.52
Starling	0.83	1.05	0.94	1.16
Wren	0.80	0.45	0.40	0.51
Great Tit	0.76	0.39	0.33	0.45
Feral Pigeon	0.68	1.02	0.86	1.18
Dunnock	0.56	0.18	0.14	0.21
Greenfinch	0.56	0.25	0.20	0.29
Song Thrush	0.55	0.16	0.13	0.19
Chaffinch	0.54	0.17	0.14	0.21
House Sparrow	0.50	0.34	0.27	0.42
Blackcap	0.46	0.14	0.11	0.18
Jay	0.45	0.11	0.08	0.13
Mistle Thrush	0.45	0.15	0.11	0.19
Great Spotted Woodpecker	0.43	0.09	0.07	0.12
Long-tailed Tit	0.43	0.17	0.13	0.21
Green Woodpecker	0.40	0.07	0.05	0.09
Collared Dove	0.34	0.12	0.08	0.15
Chiffchaff	0.30	0.07	0.04	0.10
Goldfinch	0.28	0.09	0.06	0.13
Pied Wagtail	0.28	0.09	0.06	0.12
Goldcrest	0.26	0.07	0.05	0.10
Swift	0.25	0.12	0.08	0.16
Moorhen	0.17	0.07	0.04	0.10
Coal Tit	0.16	0.04	0.02	0.05
Ring-necked Parakeet	0.16	0.06	0.03	0.08
Stock Dove	0.14	0.02	0.01	0.04
Coot	0.13	0.08	0.05	0.11
Canada Goose	0.12	0.09	0.06	0.13
Whitethroat	0.12	0.03	0.01	0.04
Kestrel	0.11	0.03	0.01	0.04
Nuthatch	0.11	0.03	0.01	0.04
Willow Warbler	0.10	0.01	0.01	0.02
Sparrowhawk	0.10	0.01	0.01	0.02
Grey Heron	0.10	0.02	0.01	0.03
Gull spp	0.09	0.04	0.01	0.06
Grey Wagtail	0.08	0.02	0.01	0.04
House Martin	0.08	0.02	0.01	0.04
Jackdaw	0.07	0.02	0.01	0.03
Linnet	0.07	0.02	0.01	0.03

Spp	Occurrence	Density	LCL	UCL
Mute Swan	0.07	0.03	0.01	0.06
Bullfinch	0.05	0.01	0.00	0.02
Cormorant	0.05	0.01	0.00	0.01
Great Crested Grebe	0.04	0.01	0.00	0.01
Garden Warbler	0.04	0.01	0.00	0.01
Kingfisher	0.04	0.00	0.00	0.01
Lesser Spotted Woodpecker	0.04	0.00	0.00	0.00
Lesser Whitethroat	0.04	0.01	0.00	0.01
Swallow	0.04	0.01	0.00	0.02
Treecreeper	0.04	0.01	0.00	0.01
Common Tern	0.03	0.01	0.00	0.02
Marsh Tit	0.03	0.01	0.00	0.01
Sedge Warbler	0.03	<0.01	0.00	0.01
Mandarin	0.02	<0.01	0.00	0.01
Pheasant	0.02	0.00	0.00	0.01
Rook	0.02	0.02	0.00	0.04
Skylark	0.02	0.01	0.00	0.03
Cuckoo	0.01	<0.01	0.00	0.01
Fieldfare	0.01	<0.01	0.00	0.01
Hybrid duck	0.01	<0.01	0.00	0.01
Little Owl	0.01	<0.01	0.00	0.01
Lesser Redpoll	0.01	0.01	0.00	0.01
Meadow Pipit	0.01	<0.01	0.00	0.01
Reed Bunting	0.01	<0.01	0.00	0.01
Redwing	0.01	<0.01	0.00	0.01
Reed Warbler	0.01	<0.01	0.00	0.01
Spotted Flycatcher	0.01	<0.01	0.00	0.01
Tawny Owl	0.01	<0.01	0.00	0.01
Tufted Duck	0.01	<0.01	0.00	0.01
Willow Tit	<0.01	<0.01	-0.01	0.02
Yellow Wagtail	<0.01	<0.01	0.00	0.01
Yellowhammer	<0.01	<0.01	0.00	0.01
Cetti's Warbler	<0.01	<0.01	0.00	0.01
Goshawk	<0.01	<0.01	0.00	0.01
Little Grebe	<0.01	<0.01	0.00	0.01
Nightingale	<0.01	<0.01	0.00	0.01
Stonechat	<0.01	<0.01	0.00	0.01
Sand Martin	<0.01	<0.01	0.00	0.01
Turtle Dove	<0.01	<0.01	0.00	0.01
Wheatear	<0.01	<0.01	0.00	0.01
Woodlark	<0.01	<0.01	0.00	0.01
Black Redstart	<0.01	<0.01	0.00	0.01

(b) Winter

Spp	Occurrence	Density	LCL	UCL
Blackbird	0.98	0.96	0.86	1.07
Woodpigeon	0.96	1.13	1.03	1.23
Carrion Crow	0.96	0.74	0.66	0.82
Blue Tit	0.93	0.84	0.75	0.92
Magpie	0.92	0.53	0.47	0.59
Robin	0.91	0.52	0.46	0.58
Great Tit	0.85	0.52	0.45	0.58
Starling	0.75	0.71	0.61	0.82
Chaffinch	0.70	0.30	0.25	0.36
Wren	0.68	0.29	0.24	0.34
Feral Pigeon	0.67	1.01	0.85	1.17
Long-tailed Tit	0.59	0.35	0.28	0.41
Greenfinch	0.58	0.30	0.24	0.36
Jay	0.57	0.15	0.12	0.18
Song Thrush	0.53	0.16	0.12	0.19
Dunnock	0.51	0.14	0.11	0.17
Redwing	0.50	0.43	0.35	0.51
Gull spp	0.50	0.48	0.38	0.58
Great Spotted Woodpecker	0.48	0.09	0.07	0.12
Mistle Thrush	0.48	0.13	0.10	0.16
Green Woodpecker	0.40	0.07	0.05	0.09
Pied Wagtail	0.39	0.14	0.10	0.18
Goldcrest	0.36	0.12	0.08	0.15
House Sparrow	0.36	0.28	0.21	0.34
Collared Dove	0.33	0.15	0.10	0.19
Goldfinch	0.33	0.13	0.09	0.17
Moorhen	0.21	0.09	0.05	0.13
Ring-necked Parakeet	0.19	0.10	0.05	0.14
Coal Tit	0.18	0.05	0.03	0.07
Fieldfare	0.16	0.05	0.03	0.08
Kestrel	0.14	0.02	0.01	0.03
Canada Goose	0.14	0.11	0.06	0.15
Grey Heron	0.13	0.01	0.01	0.02
Grey Wagtail	0.13	0.02	0.01	0.03
Nuthatch	0.12	0.03	0.01	0.05
Sparrowhawk	0.12	0.01	0.01	0.02
Coot	0.12	0.07	0.03	0.10
Bullfinch	0.09	0.02	0.01	0.03
Mute Swan	0.09	0.03	0.00	0.06
Cormorant	0.08	0.02	0.00	0.03
Siskin	0.08	0.04	0.02	0.07
Treecreeper	0.06	0.01	0.00	0.02
Kingfisher	0.06	0.00	0.00	0.01
Jackdaw	0.05	0.02	0.00	0.03
Stock Dove	0.05	0.01	0.00	0.01
Blackcap	0.05	0.01	0.00	0.01
Great Crested Grebe	0.04	0.00	0.00	0.01
Chiffchaff	0.03	0.00	0.00	0.01
Lesser Spotted Woodpecker	0.03	0.01	0.00	0.02
Meadow Pipit	0.03	0.01	0.00	0.02

Spp	Occurrence	Density	LCL	UCL
Marsh Tit	0.03	0.01	0.00	0.01
Linnet	0.02	0.02	0.00	0.03
Lesser Redpoll	0.02	0.01	0.00	0.03
Pheasant	0.02	0.01	0.00	0.02
Rook	0.01	0.01	0.00	0.03
Firecrest	0.01	<0.01	0.00	0.00
Goshawk	<0.01	<0.01	0.00	0.00
Little Owl	<0.01	<0.01	0.00	0.00
Reed Bunting	<0.01	<0.01	0.00	0.01
Snipe	<0.01	<0.01	0.00	0.00
Skylark	<0.01	<0.01	0.00	0.01
Yellowhammer	<0.01	<0.01	0.00	0.01
Brambling	<0.01	<0.01	0.00	0.01
Black-necked Grebe	<0.01	<0.01	0.00	0.01
Grasshopper Warbler	<0.01	<0.01	0.00	0.01
Lapwing	<0.01	<0.01	0.00	0.01
Mandarin	<0.01	<0.01	0.00	0.01
Peregrine	<0.01	<0.01	0.00	0.01
Stonechat	<0.01	<0.01	0.00	0.01
Spotted Flycatcher	<0.01	<0.01	0.00	0.01
Tawny Owl	<0.01	<0.01	0.00	0.01
Woodlark	<0.01	<0.01	0.00	0.01
Waxwing	<0.01	<0.01	0.00	0.01
Black Redstart	<0.01	<0.01	0.00	0.01
Yellow Wagtail	<0.01	<0.01	0.00	0.01

APPENDIX III Species recorded in BBS squares in Greater London 1994-2002 in order of reporting rate. The total number of different squares surveyed altogether was 83 (34-62 per year). Reporting rate was determined in two ways. Rate1 was determined over all squares in each year (n = 463). Rate2 was the number of times each square was occupied in any year out of the total (n = 83). Mean is the maximum count between early and late visits within 100m of the transect, summed over all squares in each year and divided by the total (n = 463).

Species	Rate1	Rate2	Mean
Blackbird	0.994	1.000	19.825
House Sparrow	0.933	0.990	43.914
Starling	0.987	0.988	50.421
Woodpigeon	0.978	0.988	19.309
Robin	0.899	0.976	7.534
Blue Tit	0.959	0.964	12.190
Carrion Crow	0.972	0.964	10.231
Magpie	0.937	0.952	7.940
Feral pigeon	0.812	0.940	32.737
Great Tit	0.834	0.940	4.825
Wren	0.862	0.928	7.721
Dunnock	0.698	0.880	3.192
Greenfinch	0.596	0.831	3.099
Song Thrush	0.596	0.783	1.994
Collared Dove	0.616	0.735	4.497
Jay	0.395	0.711	0.769
Mistle Thrush	0.397	0.699	1.056
Chaffinch	0.428	0.675	2.369
Blackcap	0.395	0.627	1.240
Goldfinch	0.285	0.614	0.942
Mallard	0.391	0.566	3.940
Great Spotted Woodpecker	0.289	0.530	0.499
Green Woodpecker	0.251	0.530	0.426
Long-tailed Tit	0.270	0.518	0.806
Chiffchaff	0.253	0.470	0.598
Moorhen	0.298	0.470	0.819
Pied Wagtail	0.184	0.458	0.367
Whitethroat	0.261	0.434	0.680
Willow Warbler	0.171	0.373	0.339
Linnet	0.149	0.325	0.676
Kestrel	0.084	0.313	0.095
Goldcrest	0.134	0.301	0.302
Canada Goose	0.160	0.277	1.741
Grey Heron	0.093	0.277	0.147
Stock Dove	0.097	0.277	0.281
Swift	0.076	0.277	0.248
Coot	0.192	0.265	1.162
Coal Tit	0.132	0.265	0.279
House Martin	0.069	0.229	0.259
Skylark	0.123	0.217	0.622
Tufted Duck	0.104	0.205	0.512
Jackdaw	0.065	0.181	0.266
Black-headed Gull	0.039	0.169	0.229
Nuthatch	0.097	0.169	0.212
Mute Swan	0.073	0.157	0.343
Pheasant	0.076	0.157	0.143
Herring Gull	0.032	0.145	0.063
Swallow	0.052	0.145	0.184
Garden Warbler	0.050	0.133	0.073

Species	Rate1	Rate2	Mean
Lesser Black-backed Gull	0.045	0.133	0.177
Lesser Whitethroat	0.035	0.133	0.039
Ring-necked Parakeet	0.028	0.133	0.048
Bullfinch	0.045	0.120	0.086
Cormorant	0.041	0.120	0.106
Grey Wagtail	0.041	0.120	0.063
Cuckoo	0.030	0.108	0.054
Sparrowhawk	0.022	0.108	0.026
Treecreeper	0.022	0.096	0.026
Yellowhammer	0.065	0.096	0.184
Great Crested Grebe	0.041	0.084	0.084
Mandarin	0.024	0.084	0.067
Reed Bunting	0.030	0.084	0.071
Spotted Flycatcher	0.028	0.084	0.039
Little Grebe	0.022	0.072	0.037
Rook	0.013	0.072	0.112
Reed Warbler	0.037	0.072	0.264
Sedge Warbler	0.028	0.072	0.065
Common Gull	0.011	0.060	0.013
Common Tern	0.015	0.060	0.052
Kingfisher	0.011	0.060	0.015
Lapwing	0.013	0.060	0.045
Marsh Tit	0.009	0.048	0.017
Wheatear	0.011	0.048	0.063
Corn Bunting	0.007	0.036	0.007
Greylag Goose	0.009	0.036	0.017
Lesser Redpoll	0.007	0.036	0.030
Meadow Pipit	0.009	0.036	0.073
Pochard	0.022	0.036	0.076
Red-legged Partridge	0.009	0.036	0.015
Lesser Spotted Woodpecker	0.004	0.024	0.004
Ruddy Duck	0.013	0.024	0.078
Tawny Owl	0.004	0.024	0.004
Tree Pipit	0.004	0.024	0.007
Barnacle Goose	0.002	0.012	0.002
Common Sandpiper	0.002	0.012	0.002
Fieldfare	0.002	0.012	0.002
Gadwall	0.002	0.012	0.004
Great Black-backed Gull	0.002	0.012	0.009
Grasshopper Warbler	0.002	0.012	0.002
Bar-headed Goose	0.002	0.012	0.002
Hawaian Goose	0.002	0.012	0.009
Redwing	0.002	0.012	0.002
Redshank	0.002	0.012	0.013
Ringed Plover	0.002	0.012	0.067
Marbled Duck	0.002	0.012	0.004
Shelduck	0.002	0.012	0.004
Shoveler	0.002	0.012	0.002
Turtle Dove	0.002	0.012	0.002
Tree Sparrow	0.002	0.012	0.004
Teal	0.002	0.012	0.002
Wigeon	0.002	0.012	0.015
Whooper Swan	0.002	0.012	0.043
Yellow Wagtail	0.004	0.012	0.004
Feral/hybrid mallard type	0.002	0.012	0.007

APPENDIX IV Ordinal regression and garden bird abundance.

The general ordinal regression model can be written:

$$g\{C_j\} = \theta_j - \beta' x$$

where C_j is the cumulative probability up to and including category *j*, **x** is a vector of covariates and $g\{.\}$, in the manner of a standard generalized linear model, is an appropriate link function. Here a logit link function is employed (the 'proportional-odds model'; McCullagh and Nelder, 1989)

In the first week of 2001, the numbers of Robins recorded in small, medium and large London gardens are as shown in Table A1. The cumulative percentages of small gardens recording birds in these five categories are shown in Fig. A1(a). Now for gardens in a single size category such as this, the ordinal regression model reduces to:

(1) $$\log\left\{C_j / \left(1 - C_j\right)\right\} = \theta_j$$

So that the parameter estimates θ , transformed back to the probability scale, provide a perfect fit to the data. For the robin data in small gardens the estimates are: θ_1 = -1.7707, θ_2 = 0.8910, θ_3 = 3.0445 and θ_4 = 3.9890. Hence, for example C_1 = 0.1455, which corresponds exactly to the observation that 16 of 110 small gardens (14.55%) recorded no robins. As shown in Fig. A1(a), the model provides a perfect fit at all abundance categories.

It is then straightforward to bring in garden size via covariates x1-x3 for each garden, where x1=1 for a small garden (and 0 otherwise), x2=1 for a medium garden and x3=1 for a large garden. The model can now be employed to estimate both intercepts θ and size 'effects' β1, β2 and β3 relating to small, medium and large gardens in turn. The constraint $\beta_3 = 0$ is applied, and then parameter estimates are obtained $\hat{\beta}_1$ = 0.9261 and $\hat{\beta}_2$ = 0.5466 associated with garden size, along with revised estimates for

the intercepts $\hat{\theta}_j$ = {-2.7779, -0.0063, 2.3092, 3.1798 }

After transformation, the probability of a small garden having one bird or less is:

(2) $$\hat{C}_2 = \frac{\exp(\theta_2 + \beta_1\ )}{1 + \exp(\theta_2 + \beta_1)} \quad = 0.7150$$

This is the probability of a garden falling into either of the lowest two categories. The full set of estimated cumulative probabilities from the model are compared with the observed cumulative frequencies in Fig. A1(b), where a good fit is observed for all three garden sizes. The fit is no longer perfect, however, as the additive nature of the model constrains the fitted cumulative probabilities to be proportional on the appropriate scale. The proportional odds model has the property that the ratio corresponding to the odds of an observation from a garden in one size category falling in or below the *j*th category, divided by the same odds for a garden of a different size, is the same for any *j*.

Note from these figures that the more the circumstances correspond to increased numbers of the bird, the lower the line representing cumulative probability lies on the graph; for example, the probability of a small garden having one bird or less (0.715) is greater than the corresponding probability for a large garden (0.498), and in general numbers of robins increase with garden size. Extension of the model, to include additional covariates representing habitat, is straightforward.

Abundance Category	size		
	Large	**Medium**	**Small**
1 (no birds)	5	21	16
2 (one bird)	28	131	62
3 (two birds)	24	78	27
4 (three birds)	4	6	3
5 (four birds or more)	4	4	2

Table A1 Breakdown of numbers of Robins recorded in five abundance categories in small, medium and large gardens in the first week of 2001.

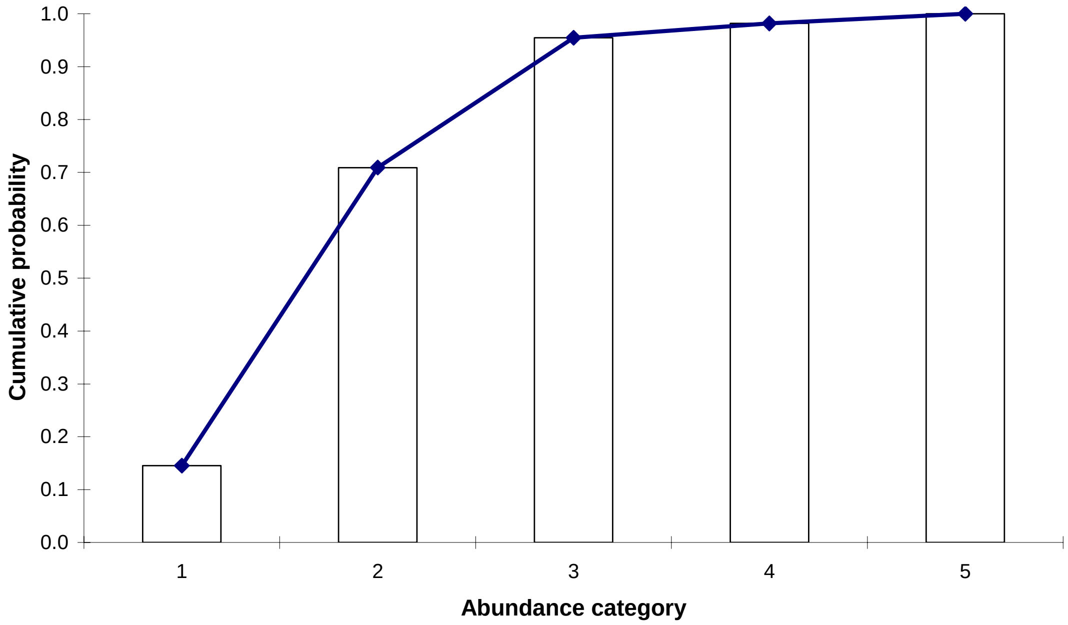

Figure A1a Cumulative proportions of small gardens in which the numbers of Robins are in, or below, the abundance category specified (bars) and estimated proportions under a simple ordinal regression model. As this model is parameter-saturated, the fit is perfect (see text).

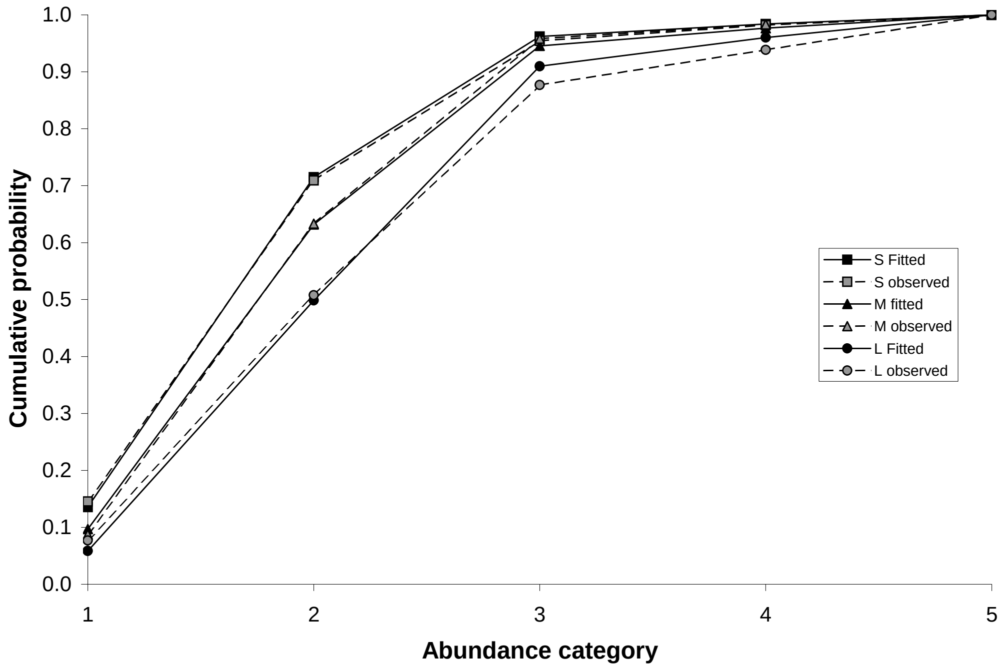

Figure A1b Cumulative proportions of small (S) medium (M) and large (L) gardens in which the numbers of Robins are in, or below, the abundance category specified and estimated proportions under an ordinal regression model with additive size effects. This constraint means the fit is no longer perfect, but it remains good (see text).