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BIODIVERSITY RESEARCH

National-scale conservation assessments at an appropriate resolution

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Abstract. Most national-scale conservation assessments are carried out at a resolution that is different from the actual size of protected areas in the study region. Coincidence between nature reserves and both hotspots (areas of high species richness) and complementary areas (sets of sites within which all species are represented) have been reported. However, the resolution (size of grid cells) of the species' distribution data upon which many of these studies are based is often close to an order of magnitude larger than the size of the reserves. Presumably, only a proportion of the species recorded in the coarse grid cells will actually be present on reserves. We use fine (2×2-km square grid cells) and coarse (10×10-km square grid cells) resolution data of national distributions for breeding birds throughout Great Britain, and presence data for avian species on Royal Society for the Protection of Birds (RSPB) nature reserves, to investigate the proportion of species in the local area (100 km^2) that are actually present on reserves. RSPB reserves contain between 50% and 70% of species from the local area. These proportions are significantly higher than for randomly selected, non-reserve areas, indicating that RSPB reserves contain higher concentrations of bird species than the wider countryside. Furthermore, on RSPB reserves these proportions of threatened and non-threatened species are equal, whereas in non-reserve areas the proportions of non-threatened species are significantly higher than threatened species. Thus, reserves hold a higher proportion of threatened species than occurs in the wider countryside.

Key words. Birds, protected areas, representative index, reserve evaluation.

INTRODUCTION

The major aim of conservation is the preservation of species. Designating protected areas — nature reserves — is carried out around the world as one method of achieving this goal (Wiens, 1996; Flather *et al.*, 1997). A considerable body of work has been carried out at academic institutions into how to select the best sites for nature reserves. In recent years this research has centred on two main quantitative concepts, hotspots and complementary areas, which are used to derive 'ideal' reserve networks using national distribution data for a variety of taxonomic groups (Prendergast *et al.*, 1993; Lombard *et al.*, 1995; Lombard, 1995a). Hotspots are defined as an arbitrary (often 5%) proportion of the most speciose grid cells in a region (Prendergast *et al.*, 1993), while within a set of complementary areas all species from a taxonomic group are represented. Conservation organizations have developed independently

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their own, mainly habitat-based (Ratcliffe, 1977; Housden *et al.*, 1991; Thomas, 1991; Sheail, 1998), approaches to reserve acquisition which, in practice, have little overlap with the theoretical work. Because of competition with other land uses, land availability, threat, opportunity and cost, along with a detailed knowledge of species' abundance and distribution, all play a role in practical reserve acquisition (Thomas, 1991; Pressey, 1994).

Constraints on reserve acquisition mean that reserve networks may be unlikely to coincide with 'ideal' networks. However, national-scale conservation assessments, that evaluate the success of the siting of existing nature reserves compared to 'ideal' networks, show that existing reserve networks do coincide with both hotspots and sets of complementary areas for many taxonomic groups (Branch *et al.*, 1995; Drinkrow & Cherry, 1995; Gelderblom *et al.*, 1995; Lombard *et al.*, 1995; Lombard, 1995a; Mugo *et al.*, 1995; Skelton *et al.*, 1995; Hopkinson, 1999). Since the 'complementary areas' approach includes all species from a taxonomic group in the minimum number of sites, it is also used in national-scale conservation assessments to prioritize areas for future site acquisition (Sætersdal *et al.*, 1993; Pressey *et al.*, 1996; Williams *et al.*, 1996; van Jaarsveld *et al.*, 1998; Araújo, 1999; Williams, 1999).

One problem with national-scale conservation assessments is the species' distribution data upon which the analyses are based (Freitag & van Jaarsveld, 1995). These data are often only available at a much coarser grid resolution than the actual (or potential) size of nature reserves in the study regions (Lombard, 1995a; Pressey & Logan, 1995). For example, in Great Britain and Ireland, reliable national distribution data (Harding & Sheail, 1990; Gibbons *et al.*, 1993; Prendergast, 1994) are available at the 10-km square ($10\ \mathrm{km} \times 10\ \mathrm{km} = 100\ \mathrm{km}^2$) resolution, but few nature reserves in Great Britain are within an order of magnitude of the size of a 10-km square (Hopkinson, 1999). In South Africa, national-scale conservation assessments are conducted at the quarter degree square (QDS $\approx 625\ \mathrm{km}^2$ (Lombard *et al.*, 1995)) resolution, but more than 70% of reserves in South Africa are smaller than $50\ \mathrm{km}^2$ (Lombard, 1995b). Therefore, although a high degree of coincidence between existing nature reserves and both hotspots and sets of complementary areas has been reported, species recorded within the coarse grid cells may not be present, and thus afforded some degree of protection on the actual reserves. This fact has been recognized (Lombard *et al.*, 1995; Mugo *et al.*, 1995; Pressey & Logan, 1995), but to date no study has attempted to quantify the proportion of species recorded in the coarse grid cells used in national-scale conservation assessments that are actually present on nature reserves.

In addition to the coarse resolution assessments of the type described above, reserve evaluations can be based on population abundance or species richnesses on reserves (e.g. Hirons & Lambton, 1991). Although valuable, such studies do not compare the performance of reserves in relation to non-reserve areas — the wider countryside. Since reserves are often actively managed (Sutherland, 1995), or at least designated as areas where human impacts are minimized, it would be expected that species perform better (i.e. are more abundant, have higher rates of increase or are more likely to be present) on reserves compared to non-reserve areas. As far as we are aware, only four studies (Warren, 1993; Virkkala *et al.*, 1994; Caro *et al.*, 1998; Sánchez-Azofeifa *et al.*, 1999) have evaluated the performance of nature reserves compared to the wider countryside.

Warren (1993) reviewed butterfly conservation in central southern Britain on sites selected for their richness in butterfly species or because they contained populations of nationally rare butterflies. He showed that the rate of loss for Red Data Book and Scarce butterfly populations was just as great on protected as on unprotected land, and concluded that 'many of Britain's rarer [butterfly] species are not being conserved effectively under the present system of site protection'. Virkkala *et al.* (1994) evaluated population densities of forest birds in southern Finland. The densities of woodpecker and flycatcher species were significantly higher in nature reserves than in unprotected areas. Caro *et al.* (1998) censused populations of 21 ungulate species inside and outside African game reserves. Densities on reserves were significantly higher for nine of the larger ungulate species. These nine species tended to be those favoured by poachers, thus the exclusion of hunters from reserves appears to be beneficial. Sánchez-Azofeifa *et al.* (1999) compared deforestation rates and the extent of forest fragmentation inside and outside protected areas in the Sarapiqui region of Costa Rica. Deforestation rates were lower in protected areas, and fragmentation (number of patches) outside protected areas increased between 1976 and 1996, while average patch size decreased. These four examples show that whilst reserves can offer more protection than the wider countryside (as common sense would suggest), Warren's study shows that this is not necessarily always the case. More attention needs to be paid to reserve evaluations that compare protected and non-protected areas.

Here, we use unpublished data for bird species on Royal Society for the Protection of Birds (RSPB) nature reserves in Great Britain, and coarse (10-km × 10-km square grid cells) and fine (2-km × 2-km square grid cells) resolution data on the national distribution of birds (Gibbons *et al.*, 1993), to address two questions. First, what proportion of the species present in areas surrounding reserves are actually present on the reserves? Using proportions in this way corrects for regional differences in total species richness. Secondly, we ask, do RSPB nature reserves, a reserve network designated almost exclusively for birds, contain higher proportions of birds than randomly selected sites?

METHODS

For this study, the avifauna of Great Britain was taken to comprise 219 native and established introduced species with wild breeding populations (vagrants, casuals and exotics were excluded). These species were categorized as either threatened or non-threatened using three different threat status designations: (1) Red Data Birds (RDB) and candidate Red Data Birds (cRDB) ($n = 123$). Species qualified for inclusion as Red Data Birds based mainly on criteria referring to rarity, localized distribution, decline in population, and international importance (Batten *et al.*, 1990); (2) Red and Amber birds ($n = 130$). Red-listed species are globally threatened or in rapid decline in the United Kingdom, while Amber-listed species are in moderate decline, rare, localized, internationally important, or of an unfavourable conservation status in Europe (Gibbons *et al.*, 1996); and (3) Schedule 1 birds ($n = 64$). Species listed in Schedule 1 (Part I) of the Wildlife & Countryside Act 1981 (HMSO, 1981). It is important to note that Schedule 1 species are the only species afforded full legal protection in the United Kingdom. The other threat status designations were compiled by non-governmental organizations and indicate those species of greatest conservation concern. For each threatened set of birds, all remaining species were classified as non-threatened.

A representative index (RI) was derived as a measure of the proportion of species on reserves found in the local area (100 km^2). A reserve RI and a tetrad RI (see below) were calculated for all bird species combined, and for the threatened and non-threatened species. When calculating species richness scores (number of species), records for subspecies were amalgamated.

Reserve Representative Index (reserve RI)

The reserve RI was a measure of the proportion of species from the area surrounding a reserve that were present on the reserve itself (Fig. 1a). The reserve RI (equation 1) was calculated for each of 73 (≈50%) RSPB reserves for which data were available.

$$\text{Reserve RI} = \frac{\text{Species richness of reserve}}{\text{Species richness of 10 km square}} \qquad \text{Eqn 1}$$

These data were unpublished records of species breeding on reserves from 1990 to 1993. To standardize recorder effort, reserves were only included in the analyses if data were available in each of the 4 years. The species richness of a reserve was defined as the total number of species recorded breeding on the reserve during this period. The species richness of the 10-km square was for the square in which the centre of the reserve lies. These latter species richness scores were calculated from the national distribution data of species' presence in the 10-km squares of the British National Grid collected during the breeding seasons of 1988–91. These data were published in *The New Atlas of Breeding Birds in Britain & Ireland: 1988–91* (Gibbons *et al.*, 1993) (hereafter referred to as the *New Atlas*). As for the reserve species richness, only records where species showed evidence of

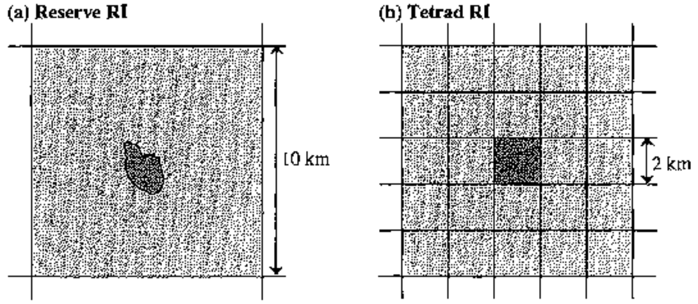

Fig. 1 Diagrammatic illustration of the reserve and tetrad RIs. (a) The reserve RI was calculated as the species richness of the reserve (dark-shaded area) divided by the species richness of the 10-km square in which the reserve is found (light-shaded area). (b) The tetrad RI was calculated as the species richness of the focal tetrad (dark-shaded square) divided by the species richness of the tetrad window (the focal tetrad plus all the light-shaded squares).

breeding were included in the 10-km square species richness score (see *New Atlas* for breeding criteria). Note that data used to compile the *New Atlas* will typically include data from the reserves. Thus, the reserve RI has a maximum value of 1. Interestingly, because of the slightly different time frames for the two datasets, in a small number of cases species were recorded as breeding on a reserve but not breeding in the 10-km square, potentially giving a reserve RI greater than 1. However, in practice all reserve RIs were less than 1.

Tetrad Representative Index (Tetrad RI)

The reserve RI provided a measure of the proportion of species from the surrounding area that were present on reserves. Unfortunately, the performance of RSPB reserves in relation to the wider countryside cannot be evaluated using the reserve RI because data for similar-sized and intensively surveyed non-reserve areas are not available. To enable such an evaluation, the tetrad RI was derived. The tetrad RI used tetrads (2-km × 2-km squares) that contained RSPB reserves to represent the actual reserves. The representative index for each reserve was calculated as the species richness of the tetrad containing the reserve divided by the species richness of the 10-km square surrounding this tetrad (Fig. 1b). Because extensive tetrad resolution data are available for birds throughout Great Britain and Ireland, it was possible to compare the tetrad RIs calculated for tetrads containing reserves with randomly selected, non-reserve tetrads. This allowed an assessment of the success of reserves compared to the wider countryside which was not possible using the reserve RI.

Tetrad resolution data were collected throughout the British and Irish National Grids as the basis for the species' distributions mapped in the *New Atlas*. Fieldworkers spent a total of 2 h in a tetrad in one season (1 April–31 July) during which the presence of all bird species observed were recorded. The 2-h survey period was either split into two separate 1-h visits (one in April or May and the second in June or July), or a single 2-h visit (after mid-May) (see *New Atlas* for full details). Different fieldworkers resurveyed some tetrads in subsequent seasons. A total of 34 510 tetrads in the British National Grid (excluding the Channel Islands) were surveyed over the 4 years (1988–91) of *New Atlas* fieldwork. Of these, 25 158 (73%) were surveyed for two separate 1-h visits. Where individual tetrads were surveyed for both two separate 1-h visits and a single 2-h visit, significantly more species were seen during the two separate 1-h visits ($t = 5.800$, d.f. = 476, $P < 0.001$). Therefore, only records collected during surveys of two separate 1-h visits were used in calculating the tetrad RI. Species were only included if they showed evidence of breeding.

The tetrad RI (equation 2; Fig. 1b) was calculated for each of the 25 158 tetrads which were surveyed by two separate 1-h visits.

$$\text{Tetrad RI} = \frac{\text{Species richness of focal tetrad}}{\text{Species richness of tetrad window}} \quad \text{Eqn 2}$$

The focal tetrad was the central tetrad in a 5 × 5 square block of tetrads (Fig. 1b). The species richness of the focal tetrad was determined from records collected in a single season. Where a focal tetrad was surveyed in more than one season, its tetrad RI was calculated as the mean of the tetrad RIs for each season. The species richness of the tetrad window included all species recorded during surveys of two separate 1-h visits in any season to the focal tetrad and the 24 surrounding tetrads (i.e. all the tetrads in the 5 × 5 square block of tetrads) — a total area of 100 km², equal in area to a 10-km square (Fig. 1b). The inclusion of the focal tetrad in the tetrad window ensured the maximum tetrad RI value was 1. Data were not available for all 24 tetrads surrounding every focal tetrad, because not every tetrad was surveyed during the course of the *New Atlas* fieldwork. The tetrad windows for 23 (0.1%) focal tetrads contained only one tetrad — the focal tetrad itself. These focal tetrads were excluded from further analyses. For the remaining 25 135 focal tetrads, the number of tetrads surveyed in the tetrad window ranged from 2 to 25. Only a small proportion (R^2 = 0.035) of variation in the tetrad RI for these focal tetrads was attributed to variations in the number of tetrads surveyed in the tetrad window.

Seventy-five of the 25 135 focal tetrads contained the centre of an RSPB reserve. These reserve tetrads were taken to represent the RSPB reserve network. This group of RSPB reserves is different from the 73 reserves for which the reserve RI was calculated, because the required data were available for different reserves. Forty-five reserves are common to both RIs. The mean of the tetrad RIs for the reserve tetrads was calculated for all species combined, and for each of the threatened and non-threatened sets of species. To provide a measure of the value of RSPB nature reserves for those species of greatest conservation concern, the difference between the tetrad RIs for each pair of threatened and non-threatened species in each reserve tetrad was also calculated (as tetrad RI for non-threatened species minus tetrad RI for threatened species). Then, the mean of these differences (mean difference) for each pair of threatened and non-threatened species was determined for the reserve tetrads.

To assess the performance of the nature reserves compared to the wider countryside, random networks of tetrads were generated and the tetrad RIs calculated and compared with the values for the real reserve tetrads. Each random network comprised 75 tetrads selected randomly from the 25 135 tetrads. For each comparison, 10 000 random networks were generated. The RIs (proportions) were arcsine-transformed for *t*-tests (Sokal & Rohlf, 1995).

RESULTS

Tetrads that contained an RSPB reserve contained significantly higher proportions of species from the local area (100 km²) than randomly selected, non-reserve tetrads (Table 1). The mean tetrad RIs for the reserve tetrads for all species combined, and each threatened and non-threatened set of species, were significantly greater than expected by chance. Furthermore, for all threatened species, none of 10 000 random networks had a greater mean tetrad RI than the reserve tetrads. These results were not a consequence of variations in the number of tetrads surveyed in the tetrad window. The mean (± standard error) number of tetrads surveyed in the tetrad window for the reserve tetrads, 13.23 (± 0.56), fell well within the range of values (11.28–16.84) for the random networks, mean = 14.04 (± 0.01). Furthermore, since all tetrads were surveyed using a standardized methodology (see Methods), these results were not a consequence of variation in recorder effort.

Seventy-one per cent (± 3) of bird species from a 10-km square that contained the centre of an RSPB reserve were recorded on the actual reserve (reserve RI). In comparison, the mean tetrad RI for reserve tetrads was 0.50 (± 0.02). These figures may appear low but they represent considerable concentrations of bird species on reserves. The reserve RIs for threatened and non-threatened species were consistently around 0.20 higher than the equivalent tetrad RIs for

Table 1 Comparisons between the mean tetrad RIs for reserve tetrads and 10 000 random networks of tetrads. Mean (± standard error) tetrad RIs are given. Significance levels are the two-tailed probabilities given by the number of means greater in 10 000 random networks. Reserve tetrads and random networks are each comprised of 75 tetrads

	Mean reserve tetrad RI	Number of means greater in 10 000 random networks	Significance
All species	0.50 (± 0.02)	8	$P = 0.0016$
RDB + cRDB	0.45 (± 0.02)	0	$P < 0.0002$
Non-RDB + cRDB	0.51 (± 0.02)	62	$P = 0.0124$
Red + Amber	0.50 (± 0.02)	0	$P < 0.0002$
Non-Red + Amber	0.50 (± 0.02)	38	$P = 0.0076$
Schedule 1	0.29 (± 0.04)	0	$P < 0.0002$
Non-Schedule 1	0.50 (± 0.02)	6	$P = 0.0012$

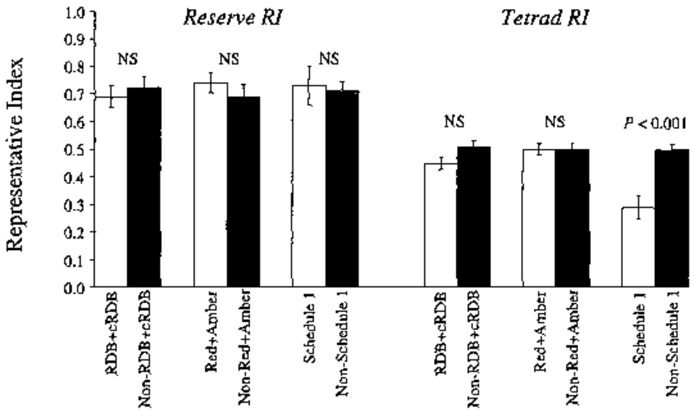

Fig. 2 Mean (± standard error bars) RIs for threatened (white bars) and non-threatened (black bars) species on RSPB reserves calculated using the reserve RI and tetrad RI for the reserve tetrads. NS: not significantly different at $P > 0.05$.

the reserve tetrads (Fig. 2). We consider the difference between the two RIs in the discussion. It does not, however, affect the relative differences shown in Fig. 2 between threatened and non-threatened species. The only significant difference between RIs for pairs of threatened and non-threatened species on reserves was that the tetrad RI for non-Schedule 1 species was greater than for Schedule 1 species ($t = 8.832$, d.f. = 148, $P < 0.001$). However, Schedule 1 species were recorded in 40% (33 of 75) of reserve tetrads. This was significantly higher ($P < 0.0002$) than in 10 000 random networks — no random network contained as many tetrads in which Schedule 1 species were recorded as the reserve tetrads. Red + Amber and RDB + cRDB species were recorded in 75 and 74 (of 75) reserve tetrads, respectively, which were not significantly different than in 10 000 random networks.

In randomly selected, non-reserve tetrads, the mean difference between the RIs for non-threatened and threatened species was significantly higher than in the reserve tetrads for RDB + cRDB and Schedule 1 species (Table 2). For these two threat status designations, reserves increased the proportions of threatened species in the tetrad more than the proportions of non-threatened species. The mean difference between the proportions of non-Schedule 1 and Schedule 1 species was, on average, 0.16 higher in random networks, while between the non-RDB + cRDB and RDB + cRDB species the average difference was

Table 2 Comparisons in the mean (± standard error) differences in tetrad RIs between reserve tetrads and 10 000 random networks of tetrads. The mean differences were calculated by first determining the difference between the tetrad RIs for each pair of threatened and non-threatened species in each reserve tetrad (calculated as the tetrad RI for non-threatened species minus tetrad RI for threatened species). Then, the mean of these differences (mean difference) for each pair of threatened and non-threatened species was determined for all reserve tetrads. Significance levels are the two-tailed probabilities given by the number of mean differences smaller in 10 000 random networks. Reserve tetrads and random networks were each comprised of 75 tetrads. NS: not significant at $P > 0.05$

	Mean difference between reserve tetrad RIs	Mean of differences between tetrad RIs in 10 000 random networks	Number of mean differences smaller in 10 000 random networks	Significance
RDB + cRDB	0.06 (± 0.02)	0.13 (± 0.0002)	6	$P = 0.0012$
Red + Amber	0.00 (± 0.02)	0.03 (± 0.0002)	419	$P = 0.0838^{NS}$
Schedule 1	0.21 (± 0.05)	0.37 (± 0.0003)	0	$P < 0.0002$

0.07 higher. Compared to the wider countryside, tetrads that contained RSPB reserves had a higher ratio of the proportions of threatened to non-threatened species.

DISCUSSION

Tetrads that contained RSPB reserves contained significantly higher proportions of species from the local area (100 km^2) than tetrads drawn at random from the wider countryside (Table 1). For reserves, there was no significant difference in the proportions of threatened and non-threatened species in five of six comparisons (Fig. 2). However, compared to random networks of tetrads, reserve tetrads not only contained higher concentrations of bird species from the local area, but also increased the proportions of threatened species to a greater degree than the proportions of non-threatened species (Table 2). This suggests RSPB nature reserves not only contain higher proportions of birds from the surrounding area than expected by chance, but also contain higher concentrations of threatened compared to non-threatened species.

There are three possible reasons why the mean reserve RIs for the threatened and non-threatened sets of species were higher than the equivalent tetrad RIs for the reserve tetrads (Fig. 2). First, there were differences in the surveying techniques used to collect data for the two RIs. The species richness of reserves, used in the reserve RI (equation 1), was an accumulation of species recorded over 4 years by wardens who were intimately acquainted with their reserves. In contrast, the species richness of focal tetrads, used in the tetrad RI (equation 2), was derived from a survey where observers were restricted to two separate 1-h visits in a single breeding season. The species richness of a reserve was therefore based on a more comprehensive survey than the species richness of a focal tetrad. This is likely to have a pronounced effect on records for rarer or cryptic species as they are less likely to be recorded during a restricted survey period. This may also explain why the mean tetrad RIs for reserve tetrads were significantly different between Schedule 1 and non-Schedule 1 species, while the mean reserve RIs between these two sets of species were not significantly different (Fig. 2).

Secondly, reserve tetrads were used to represent the RSPB reserve network. Reserve tetrads were those tetrads which contained the centre of an RSPB reserve. Actual reserves may occupy portions of adjacent tetrads. Therefore, using data for a single tetrad may underestimate the number of species actually on the reserve and thus lower the tetrad RI. However, this is balanced by variations in the size of reserves. Although the mean (± standard error) area of reserves in the 75 reserve tetrads was 499 ha (± 146), only 16 (21%) were larger than 400 ha (the size of a tetrad) and 31 (41%) were smaller than 100 ha. All the species recorded in a tetrad may not therefore be present on the actual reserve, and this may lead to an overestimate in the tetrad RI.

The final reason why the mean reserve RIs were higher than the equivalent tetrad RIs for reserve tetrads is that the tetrad RI used a 100-km² tetrad window around the focal tetrad to represent the local area. The reserve RI used the 10-km square in which the centre of the reserve lies. However, reserves may also occupy parts of adjacent 10-km squares. This means that some species may be recorded on the reserve but not be present in the 10-km square used to represent the local area. In this instance, the reserve RI will become inflated.

On average, 70% of the breeding bird species found in a 10-km square containing an RSPB reserve are also present on the reserve itself (Fig. 2: reserve RI). This result has important implications for national-scale conservation assessments that are made at an inappropriate resolution compared to the size of nature reserves in the region (for example Lombard *et al.*, 1995; Mugo *et al.*, 1995). At the resolution of a 10-km square, existing nature reserves in Great Britain show a high degree of coincidence with both hotspots and complementary areas for several taxonomic groups, and a significant proportion of species have been recorded within the 10-km squares of the reserve networks (Hopkinson, 1999). However, the present analyses show that not all species recorded in a 10-km square will actually be present, and potentially receiving some protection, on the reserves within the square. Finer-scale surveys are required to determine whether all species of conservation concern recorded within the coarse grid cells used in national-scale conservation assessments are, in fact, adequately protected.

Comparisons between the mean tetrad RIs for the reserve tetrads and random networks allowed an evaluation of the success of RSPB reserves in relation to the wider countryside. A comparison of this type was not possible for the reserve RI because data for similarly sized and intensively surveyed non-reserve areas are not available. RSPB reserves contained a higher proportion of species from the local area than expected by chance (Table 1). In the wider countryside (random tetrads) the proportion of threatened species was less than the proportion of non-threatened species, as might be expected. On reserves, the differences between these proportions was significantly reduced between the threatened and corresponding non-threatened Schedule 1 and RDB + cRDB species (Table 2). Thus, RSPB reserves increased the concentrations of threatened species more than that of non-threatened species. This is unsurprising, given the RSPB's reserve acquisition strategy which is based upon 'the need to conserve important bird species and assemblages that occur in association with scarce or priority ecosystems that themselves are in urgent need of conservation action' (Thomas, 1991). In Great Britain's patchy and highly modified landscape, reserves provide important refuges for threatened species.

There was no significant reduction in the mean difference between the tetrad RIs for the non-Red + Amber and Red + Amber species in reserve tetrads and random tetrads (Table 2). This may be linked to the fact that the Red + Amber threat status designation includes a number of widespread and abundant species which, in a limited survey period, are more likely to be observed than less abundant species. These species (such as skylark *Alauda arvensis* and song thrush *Turdus philomelos*) are included on the Red + Amber list because of their rapid (≥50%) or moderate (25–49%) declines in population and/or range sizes during the last 25 years (Gibbons *et al.*, 1996). These species are, however, still relatively widespread compared to RDB + cRDB and Schedule 1 species. The differences in mean range sizes between the different pairs of non-threatened and threatened species are all significant but is lower between non-Red + Amber and Red + Amber species (Table 3).

National-scale analyses at the tetrad resolution are possible in Great Britain for breeding birds because species' distribution data are available at this relatively fine resolution. In this country, county atlases, which frequently use tetrads as the basic recording unit, in conjunction with national distribution datasets, may provide a means for regional assessments of this type for other taxonomic groups. These results show that between 50% and 70% of bird species recorded in the coarse grid cells used in national-scale conservation assessments were actually present in tetrads containing an RSPB reserve, or in the reserve itself. RSPB reserves contained significantly higher proportions of species from the surrounding area than randomly selected, non-reserve areas. Previous conservation assessments

Table 3 Mean range sizes for breeding birds in Great Britain. Mean (± standard error) range sizes are given with comparisons between threatened and non-threatened species. Range sizes for each species were the number of occupied 10-km squares in the British National Grid. U_{adj}: Mann–Whitney U-statistic adjusted for ties

	n	Mean range size	Difference	U_{adj}	Significance
RDB + cRDB	123	532.5 (± 60.1)			
Non-RDB + cRDB	96	1584.8 (± 83.9)	1052.3	8.610	$P < 0.001$
Red + Amber	130	641.1 (± 69.0)			
Non-Red + Amber	89	1508.9 (± 86.5)	867.8	7.046	$P < 0.001$
Schedule 1	64	176.2 (± 35.2)			
Non-Schedule 1	155	1331.4 (± 68.7)	1155.2	9.356	$P < 0.001$

carried out at coarse resolutions may have been too optimistic in their assessment of nature reserve networks (Lombard *et al.*, 1995; Mugo *et al.*, 1995; Hopkinson, 1999). Coarse resolution assessments are important in determining areas to prioritize for reserve acquisition, especially where data are not available at finer resolutions. However, fine-resolution analyses must also be undertaken to ensure all species of conservation concern are adequately protected on actual reserves.

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