Analysis of Moorland Breeding Bird Distribution and Change in the Peak District

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SUMMARY

The 2004 Moors for the Future Partnership breeding bird survey of the Peak District moorlands covered a total area of 503 km^2 and highlighted declines for Dunlin, Twite and Wheatear, and increases in Curlew, Lapwing, Snipe and Whinchat populations from the 1990 English Nature baseline survey of the breeding birds of the South Pennines. This report presents analyses of these data to assess the underlying factors affecting bird distribution and changes in bird populations in the Peak District.

Having accounted for strong topographical effects, vegetation height was generally the most important predictor of fine-scale distribution. Golden Plovers were associated with short swards, whilst Reed Buntings, Snipe and Wren were most frequently recorded from tall vegetation. Curlew, Lapwing and Wheatear each favoured intermediate heights. Seven species were associated with cotton grass, whilst three, Curlew, Wren and Reed Buntings associated with heather, and intermediate levels of grass. Waders generally avoided areas of high disturbance. Most models had good predictive ability.

Complementary analysis of species abundance was conducted at a 1 km resolution. Topographical effects were again the most important determinants of abundance. Golden Plover and Dunlin were most common on cotton grass, whilst Curlew, Snipe and Reed Buntings were associated with grass cover, which Red Grouse avoided. Curlews also associated with heather, whilst Skylarks and Meadow Pipit abundances were negatively correlated with heather cover. Management effects on species abundance were variable and weak, and varied between species. Few models had good predictive power.

Models describing changes in abundance also had little generality. Reasons for this are discussed, but mean these results should be interpreted with caution. Declines in four species were most apparent on squares with high cover of non-heather heath, whilst two declined most on heather moorland. In contrast, the greatest increases in Curlew abundance (the species with the most robust change model) were on areas of heather moorland, whilst Golden Plovers increased on cotton grass habitats, but declined in other areas. There were few significant effects of management (burning or grazing) on bird population change at either the 1 km level, or at a larger, management unit scale. Thus, ESA agri-environment prescriptions or a measure of grouse-moor management failed to correlate strongly with changes in the abundance of any species.

Maps outlining the variation in changes in bird populations across the Peak District are produced to aid the interpretation of results. These have been used to produce a number of ‘hotspot’ maps, highlighting areas of particular conservation importance. A map combining data on red-list and SPA species, is a useful summary of these, indicating certain northern, central and eastern moors as being of greatest conservation interest.

An extensive discussion presents species-specific summaries of the results, in the context of other research, and outlines general principles for moorland management. It also presents some caveats associated with the work, largely as a result of error associated with summarising extensive survey data at a 1 km resolution, and some of the habitat and management measures. These should be borne in mind when interpreting the results.

1. INTRODUCTION

UK moorlands hold an important breeding bird assemblage, including eight species listed in Annex 1 of EU Directive 79/409 on the Conservation of Wild Birds and 28 others of conservation concern in the UK (Thompson *et al.* 1995, Gregory *et al.* 2002). Whilst national survey data provides good population information for some species, such as evidence for national declines in Black Grouse *Tetrao tetrix* and Ring Ouzel *Turdus torquatus* (Hancock *et al.* 1999, Wotton *et al.* 2002), accurate population and trend data for more widespread species and for specific regions are generally lacking. A recent analysis of changes in British upland breeding birds during the last 10-20 years has reported declines in many wader populations, and these were particularly widespread for Lapwing *Vanellus vanellus*, Dunlin *Calidris alpina* and Curlew *Numenius arquata* (Sim *et al.* 2005). Although passerines appeared to fair better, notable declines were apparent for Twite *Carduelis flavirostris* and Ring Ouzel, whilst Whinchat *Saxicola rubetra* and Stonechat *Saxicola torquata* populations have increased.

Major land-use and management changes across the UK uplands have been implicated in such putative declines, including large-scale afforestation, declines in grouse-moor management, and habitat change resulting from changes in grazing practices (e.g. Avery & Leslie 1990, Hudson 1992, Fuller & Gough 1999). It is important that sound science underpins our knowledge of how such management changes impact upon moorland bird populations, to guide future conservation management practices. To date, such studies have been largely correlative, relating current bird distribution to habitat and management (e.g. Avery *et al.* 1989, Stroud *et al.* 1990, Tharme *et al.* 2001, Pearce-Higgins & Grant, 2006), whilst data on population change in relation to such factors provides for more powerful analyses (e.g. Buchanan *et al.* 2003).

The Peak District National Park, located in the southern part of the South Pennines, England, supports important populations of Curlew, Golden Plover and Merlin *Falco columbarius* (Stillman & Brown 1994, Carr & Middleton 2004). The South Pennines is designated as a Special Protection Area (SPA) for Golden Plover, Short-eared Owl *Asio flammeus*, Peregrine *Falco peregrinus* and Merlin *Falco columbarius,* SAC and includes five SSSI, of which four are located within the Peak District. Complete bird surveys of the Peak District moorlands were carried out in 1990 (Stillman & Brown 1994) and 2004 (Carr & Middleton 2004) over more than 500 km² contiguous area. Analyses of the 1990 data indicate that Dunlin and Golden Plover were strongly associated with blanket bog high altitude plateaux, whilst other wader species were more abundant at lower altitudes. Merlin, Red Grouse and Short-eared Owl *Asio flammeus* were each associated with heather moorland, in particular tall heather for Merlin, with which Curlew also showed an association. Whinchat numbers were positively correlated with bracken cover (Stillman & Brown 1994).

Analysis of regional moorland bird population trends in UK uplands by Sim *et al.* (2005) indicates that the South Pennines tends to show more favourable changes in species abundance than other UK mainland regions, although significant declines in Dunlin, Meadow Pipit, Wheatear *Oenanthe oenanthe*, Ring Ouzel and Twite populations were recorded from 1990 – 2002. Curlew numbers doubled in the same time period, with the South Pennines the only mainland UK upland region covered where this species increased (Sim *et al.* 2005). The recent resurvey of the Peak District moorlands in 2004 (Carr & Middleton 2004) broadly supports the findings of Sim *et al.*

(2005) for the South Pennines with large (>25%) declines apparent for Dunlin, Twite and Wheatear, and large (>25 %) increases in Curlew, Lapwing, Snipe and Whinchat populations since 1990, although these large increases in Lapwing and Snipe abundance, contrast with broadly stable populations across the South Pennines as a whole (Sim *et al.* 2005). These changes have occurred over a time when the intensity of grouse-moor management in the Peak District has remained stable or increased, whilst there have been considerable reductions in sheep stocking levels, as large areas of moorland have been entered into agri-environment schemes such as ESA agreements.

These 2004 data therefore provide an important opportunity to examine the relative importance of habitat and management in driving such bird population changes. The principal aim of this study is to improve the understanding of patterns of moorland bird distribution in the Peak District. Changes in distribution and abundance over the past 14 years have been assessed and related to habitat variables. This information can form an evidence base for rational decision-making about the management of resources and priority setting for targeting areas for conservation (and/or recreation) in the National Park. The objectives are therefore to:

a) establish bird-habitat associations with respect to vegetation condition, land use and recreational patterns.
b) assess changes of distribution and abundance of birds between 1990-2004.
c) create 'hotspots' map and identify priority areas for bird conservation.

The report is written along the lines of a scientific report. If as a reader you wish to skip much of the technical detail, you should probably concentrate on the species accounts in the discussion (Pp 93-108) and general principles of moorland management (Pp 108-111), although the results section includes many graphs and maps which may be of interest. However, please be aware that within the remaining text in the report are some important caveats relating to the limitations of the data on which we based the analysis, and some of the difficulties interpreting this kind of analysis, that may also be of interest.

2. METHODS

2.1. Study area

The study encompasses the unenclosed upland area of the Peak District National Park. Large Millstone Grit plateaux of 500 – 600 m altitude dominate the study area, which are overlain by deep (3-4 m) peat and covered by species poor *Eriophorum vaginatum* dominated bog vegetation, with *Empetrum nigrum* and *Vaccinium myrtillus* occurring on drier areas and slopes. Areas of *Calluna vulgaris* dominate lower altitudes, and are generally actively managed for Red Grouse, although this vegetation has been replaced in some areas by acid grassland of *Molinia caerulea*, *Nardus stricta* and *Deschampsia flexuosa,* largely through high levels of grazing (e.g. Anderson & Yalden 1981). The lower boundary of the unenclosed land is generally at 250 – 350 m altitude, and comprises either enclosed pasture, meadow or woodland. Areas of moorland within the Peak District are used for sheep grazing and sport shooting of Red Grouse, whilst the National Park receives large numbers of recreational visitors (see Pearce-Higgins & Yalden 1997).

Surveys were conducted across the entire unenclosed uplands, which in 2004 were delineated by a map of the moorland boundary (DEFRA unpubl.) of the Peak District. A total survey coverage of 459 km^2 was achieved in both years (Figure 2.1). For a small number of estates (27 km^2), access could not be agreed in 2004, but they were surveyed in 2005 (Shepherd 2005). To avoid potential biases, these data were not included in the formal analysis of this report. However, we present gross changes in species abundance within the appropriate squares in Appendix 1.

2.2. Bird surveys

The Brown & Shepherd (1993) methodology was used to census bird populations. In short, this involves two visits to each 1 km square; the first between 1 April – 15 May, and the second from 16 May – 30 June. During each visit, areas of moorland were covered to within 100 m, and the locations of behaviour of breeding birds mapped on 1:25,000 maps. Surveys were conducted between 09:00 – 18:00 during suitable weather conditions. Although a methodology primarily designed to survey breeding wader populations, sightings of most other bird species were recorded in the same way. Red Grouse, Skylark A*lauda arvensis* and Meadow Pipit numbers were, however too frequent to map individual sightings, but instead abundances were tallied per 1 km square during the first visit only.

This technique was originally devised to survey large areas of extensive moorland efficiently, to identify areas of high abundance of breeding waders for future site designation (Brown & Shepherd 1993). In this analysis, the data are being used to estimate bird abundance at a 1 km level. An assessment of the efficacy of this survey method to estimate abundance at the 1 km scale, is produced by correlating maximum counts recorded from the two visits within each 1 km from 2000 (Sim *et al.* 2005) with the 2004 counts (Appendix 2). In general, there appears to be considerable inter-annual variability for many species as indicated by the relatively low correlation coefficients for most (7 of 12 showed a significant correlation between the two counts when excluding squares with null records). This variation may be due to a number of different effects. For some species, such as Skylark and Meadow Pipits, this method is not the preferred technique for assessing abundance (Thirgood *et al.* 1995, Buchanan *et* *al.* 2006), whilst others, such as Ring Ouzel and Snipe, are difficult to detect. Some species are likely to vary in their site specificity between years, for example Lapwing distribution on fields can vary considerably from one year to another (Mark Bolton, pers. comm.), or have large territories, such as Curlew, and therefore may be allocated to different 1 km squares in different years. Finally, counts may be heavily dependent upon the timing of any survey visit in relation to the timing of breeding, such as with Golden Plovers (Pearce-Higgins & Yalden, 2005).

Prior to analysis, we excluded the few records where observers considered the birds to be migrating through the area (mostly records from the first visit where birds were seen only in fast, direct flight at high altitude), and records that were outside of unenclosed moorland areas (DEFRA unpubl.). Detailed analyses were conducted for 14 species that were sufficiently abundant for analysis of habitat associations and change. A cut-off of >75 sightings was used, with Dunlin the rarest species whose data were analysed (76 sightings), whilst the next most frequently observed individual being Mistle Thrush with 54 individuals sighted. Given the problems which we encountered producing some of the models for Dunlin (see results), it is likely that detailed analysis would not have been possible for these rarer species. We also excluded data on Kestrel because observations were largely of hunting birds and therefore not closely tied to particular localities, and Willow Warbler, whose abundance will be linked to the occurrence of non-moorland vegetation, i.e. bushes and trees, which were poorly mapped. The list of species whose data were analysed in detail in this report is therefore as follows; Curlew, Dunlin, Golden Plover, Lapwing, Meadow Pipit, Stonechat, Snipe, Red Grouse, Reed Bunting, Ring Ouzel, Skylark, Wheatear, Whinchat, Wren, although data for Meadow Pipit, Skylark and Red Grouse were only considered at a 1 km resolution.

2.3. Moorland habitat data

Vegetation data

Data on vegetation were obtained from two sources. The first was a map of broad-habitat classes produced from various sources, mainly ESA habitat maps for the Dark Peak and South West Peak (DEFRA), derived from 1988 air photo imagery, amended with data from recent with NVC and phase 1 surveys (PDNPA, EN, MFF). These data classified vegetation communities into fourteen categories (Table 2.1), and hence provided a detailed breakdown of the habitats available within the Peak District, although were dated from a period c. 15 years earlier than the 2004 bird data. Further, the use of such vegetation, which categorise what in reality are complex mosaics and gradations of habitat into discrete areas of single habitat types can introduce additional bias (Seoane *et al.* 2004). Bare ground and eroding moorland were combined into one category due to their apparent similarity.

The second was derived from manipulation of a Landsat 7 image captured on 7/4/2003 (Appendix 3). This was used to predict the cover of four vegetation types; Heather (*Calluna vulgaris*), Grass (consisting of *Molinia*, *Nardus* and fine-leaved Grasses), Cotton Grass (*Eriophorum* spp.) and Non-heather dwarf shrub (*Vaccinium myrtillus* and *Empetrum nigrum*), together with vegetation height. The accuracy of predictions using this method averaged 74 %. To test which data were the most appropriate for analysis, we produced fine-scale models using vegetation data from both sources, and assessed their relative predictive ability from their ability to predict bird distribution across a subset of the data excluded from the initial analysis (see below). The accuracy of the model predictions from both outputs are compared in Appendix 4, and indicate that the satellite-derived data produced more accurate predictions in each case. Most estimated models using the ESA habitat groups did not differ from random estimates, as indicated by AUC values (Swets 1988) of equal or lower than 0.5 (i.e. only 50% or less of all occurrences and non-occurrences were predicted correctly). As a consequence, the satellite derived habitat data are used for the majority of the modelling undertaken, with the exception of the preference index (see below), which provides for a crude analysis of the habitat associations of a wide-range of species with the ESA habitat categories.

Topographical data

Data on elevation and slope were obtained from a digital terrain model with a 50 m resolution, derived from 10 m spot height data (data supplied by Infoterra). We compared using mean slope in the analysis with the proportion of an area with either shallow (< 5°) or steep (> 10°) slopes, which we initially thought may offer a better habitat description for particular species. However, such measures were always very strongly ($r > 0.7$) correlated with mean slope, and so for parsimony we simply used mean slope in the analysis. A map of watercourses (Ordnance Survey data, PDNPA licence number LA 076015 2004) was used to create a gridded data set with a 50 m resolution of distance (m) to the nearest watercourse, whilst a polygon outlining peat cover was derived from a digital soil map (BGS 1:50,000 soil map, BGS digital licence 2004/007A).

Moorland fragmentation data

Two variables providing an assessment of the degree of moorland fragmentation were derived from CS2000 (CEH, 2000), the first was the proportion of a 1 km buffer surrounding each 1 km square that was covered by woodland habitat classes, and the second the proportion of the buffer covered by enclosed farmland habitat classes.

2.4. Moorland management data

Disturbance

A map of footpath locations was produced from 2002 GetMapping aerial photographs by the Moors for the Future team. This includes many more footpaths and tracks than Public Rights of Ways (PRoWs) at more precise locations than delineated on OS maps. In addition, the PDNPA ranger service collated a ‘level of access’ map across the Peak District Moorlands, which graded the intensity of recreational use on a 1-5 scale from low to high pre Countryside Rights of Way (CRoW) Open Access commencement in September 2004. Each footpath was graded by National Park rangers on a linear scale of 1 (low) to 5 (high) to provide an index of visitor use. These data were used to produce a disturbance map of the Peak District with a 30m resolution. This was calculated for each square (j) as the average of the footpath grades across squares with a 1 km radius (*ki*), selected as the likely maximum distance over which disturbance effects were likely to operate. These averages were weighted by the reciprocal of distance (D) between squares ij as follows:

$$Y = \sum_{j=1}^{ki} Disturbance_i \left(\frac{1}{D_{ik}} \right)$$

Heather burning and grouse-moor management

Polygons outlining broad areas of heather burning were digitised from the 2002 GetMapping aerial photographs by the Moors for the Future team. These did not delineate individual heather burns, but outlined areas of moorland subject to regular patterns of heather burning. For analysis of habitat associations, we used the % cover of burn polygons within the specified buffer.

These data were also used to provide an index of the intensity of grouse-moor management within larger land management unit polygons (see below). This was assessed from the proportion of heather cover, as derived from the satellite image (see above), which was located within heather burning polygons. Thus, management units subject to a high intensity of grouse-moor management are likely to exhibit a high proportion of heather cover within the burn polygons. This measure was used in preference to a simple measure of the cover of burn polygons, to eliminate any bias in the latter, due to the inability of estate managers to burn on blanket bog that will still be subject to predator control regimes.

Grazing

Larger-scale data on the distribution of ESA agri-environmental scheme agreements were available (data supplied by DEFRA). These were polygons identifying grazing management units, and separated by Tier classification (Table 2.2). Because many of the non-moorland tiers were small and contained few birds, classes were lumped based on maximum stocking levels to increase the number of species for which statistical analysis was possible.

2.5. Statistical analysis

Habitat preference Index

Although the ESA habitat data were inappropriate for the detailed modelling of habitat associations due to their relatively weak predictive performance at the fine-scale (see above), it was possible to conduct broad habitat associations using these vegetation data. This meant that we were able to take advantage of the finer division of these habitat data into 14 classes, allowing an examination of habitat associations in relation to some habitats not mapped by the satellite-derived data. However, this analysis is simplistic in nature, and does not take into account other factors that may influence bird distribution, being based on a comparison of the observed frequencies of sightings of each species in the different habitat categories (oc), with what would be expected from the area of each habitat (ec). The significance of any habitat selection was then tested using a simple Chi-squared test. To aid the interpretation of these results, each species was allocated a simple preference score (P) from –1 to +1 for each habitat as follows, where –1 is absolute avoidance, 0 indicates usage is as expected from the area of that habitat, and +1 is absolute selection.

$$P = \left(\frac{oc - ec}{oc + ec} \right)$$

Analysis of fine-scale habitat associations

Bird locations were mapped to an accuracy of 100 m. For analysis, each of these was buffered with an additional 100 m radius, and average or proportional cover values of habitat and management variables calculated for each point from within the buffer (i.e. radius of 150m of point location), using MapInfo Professional v 6.0 (MapInfo Corporation 2000). For comparison with locations where birds were not sighted, we generated random points within the surveyed area and selected at random a sample of these points equal to the number of bird locations for each species, excluding any random locations that replicated known locations for that species. Habitat and management data were extracted for these random points in the same way as for the bird locations. All proportions were arcsine-squareroot transformed before analysis.

Spatial autocorrelation is a common problem with such analysis, as points located close together, which are likely to be subject to similar environmental variables, are not truly independent. To minimise the risk of spatial autocorrelation resulting in spurious correlations, we used an autocovariate term (A; Augustin *et al.* 1996). This is the weighted average of the number of occupied points amongst a set of K_i points within a species specific buffer of point *I*. The weight given to a point *j* is $Wij = 1 / h_{ij}$, where h_{ij} is the Euclidean distance between the squares *i* and *j*. The response variable *Y* was 1 if a bird was recorded from point *j*, or 0 if absent.

$$A = \sum_{j=1}^{Ki} WijYi$$

The maximum distance for the buffer over which each autocovariate term was calculated was determined separately for each species by calculation of a semi-variogram (Cressie 1991) based on bird locations and random points. Semi-variograms were produced in Idrisi Kilmanjaro (Idrisi 2003), and the range over which semi-variance increased with distance used as the maximum distance over which the autocovariate was calculated for that species. Semi-variograms are presented in Table 2.3. Where models failed to converge, due to a lack of strong spatial trends in abundance, the autocovariate term was calculated across the minimum buffer distance (1 km) for those analyses. Being a nuisance variable to correct for spatial autocorrelation, values relating to the autocovariate term are not presented in the results.

Thus, for each species we produced a dataset with an equal number of bird and random locations, and the habitat variables summarised in Table 2.3. Because the distribution of cotton grass cover was completely nested within the outline of the peat polygon, and the focus was not on associations with peat, this variable was excluded from the fine-scale analysis. For investigating the effects of management (i.e. heather burning and disturbance) we originally anticipated using a residual analysis having taken account of the potentially confounding effects of habitat (cf. Tharme *et al.* 2001). However, this approach has recently been criticised, even when the variables of interest are correlated with other, nuisance, variables (Garcia-Berthou 2001, Freckleton 2002). Therefore, we adopt the suggestion of Freckleton (2002) and analysed all terms together.

To assess the ability of each model to predict distribution, we removed a certain number of data points to use as a test dataset. For each species with over 400 locations, we randomly excluded 50 bird and 50 random locations, and for species with less than 400 locations we selected a test dataset with 25 of both bird and random locations. The remaining data were used for model building. All data analysis was carried out in the software package R v 2.1 (R Core Development Team 2005).

We used a model averaging approach to model selection and derivation of parameter estimates (Gibson *et al.* 2004, Rushton *et al.* 2004), to minimise the potential problems that collinearity (Table 2.4**)** between predictor variables causes for stepwise model selection techniques (i.e. Graham 2003). All possible models involving the nine main effects (four habitat, four topographic and the autoregressive variable) and each of their quadratic terms (first scaling each variable to a mean and standard deviation of zero and one to improve model convergence) were built. Using the BMA package v 3.0.1 we used Baysian model averaging (based on the Baysian Information Criterion and Occam's window of 0.99) to produce parameter estimates for generalized linear model with binomial errors predicting presence or absence on the training dataset of each species. To aid subsequent interpretation of the results, we repeated this process 100 times including a randomly generated parameter with a uniform distribution between zero and one. For each of the 100 repeated models, we calculated the weighting of the random parameter and its quadratic term and estimated the value below which 95% of the random estimates fell. This value is presented for each species, along with the overall weighting given to each term in the models, to assess the approximate significance (hereafter significance) of individual parameters, by comparing the selection probability of any given parameter with the 95% confidence limit of a randomly generated parameter (per Whittingham *et al*. 2005). For three models, the 95 % confidence limits of the random parameter overlaps with a 100% selection probability, suggesting that in these cases it is possible for a random variable to occur in the most supported models by chance. In these cases, only models terms with a 100 % weighting are regarded as significant. Models were constructed with a binomial error structure and logit link function.

Analysis of 1 km habitat associations

The abundance of each species within each 1 km square was derived from the maximum count of the number of individuals of a species recorded from the two visits to that square, excluding sightings from non-moorland habitats (largely enclosed fields). This figure minimises the error associated with the interpretation of behavioural codes, and the amalgamation of bird registrations from separate visits which are regarded to be of the same pair (cf. Sim *et al.* 2005, Pearce-Higgins & Yalden 2005).

Habitat and management variables were averaged across each 1 km square, and incorporated into the analysis, including the proportion of a 1 km buffer surrounding each square that was covered by either woodland, or enclosed farmland habitat categories (Table 2.3). An autocovariate term was calculated as described above, but with a 1 km resolution, rather than the 100 m resolution of the fine-scale association data (Table 2.4). Thus, for each species, analysis was conducted using the composition of heather, grass, cotton grass and non-heather dwarf shrub, vegetation height, altitude, slope, cover of peat, distance to the nearest watercourse, disturbance score, cover of burn polygon, cover of woodland, cover of enclosed farmland, and the autocovariate term as explanatory variables, in conjunction with their quadratic terms (see above). Correlations between these predictor variables are given in Table 2.6. Prior to model building, data from 50 squares were randomly excluded for each species, to provide an independent test data set against which to test the accuracy of our model predictions. As in the fine-scale analysis, model averaging was used to identify terms with a high selection probability and for the estimation of model coefficients. As the analysis was of count data, models were constructed with a Poisson error structure and log link function.

Analysis of change at 1 km resolution

Analysis of correlates of change was also conducted at the 1 km level using the same habitat and management variables as the 1 km analysis of habitat associations. A measure of change was produced by modelling the 2004 count as a proportion of the total number of birds recorded from each square in 1990 and 2004, using a binomial error structure and logit link function as follows:

$$Change = \operatorname{logit}\left(\frac{Count04}{Count04 + Count90}\right)$$

Previous analyses of correlates of change in bird populations have modelled count in the later year with a Poisson error structure, with the log of the first count as an offset (i.e. Peach *et al.* 2001, Buchanan *et al.* 2003), but this results in the elimination of data within initial count of zero (which cannot be logged). This new approach is equivalent to modelling log (count04/count90), but allows zero counts to be modelled, and automatically weights appropriately for large and small counts (Rhys Green, *in litt.*). Thus, change was modelled at this scale as a function of heather, grass, cotton grass and non-heather dwarf shrub, vegetation height, altitude, slope, cover of peat, distance to the nearest watercourse, disturbance score, cover of burn polygon, cover of woodland, cover of enclosed farmland, and the autocovariate term. The autocovariate term was calculated again at the 1 km level, but from an index of change in each square rather than the 2004 count, as follows:

$$Change = \frac{Count04 - Count90}{Count04 + Count90}$$

Analysis of change across land management units

Analysis of change in relation to larger-scale management processes (agri-environment schemes and the intensity of grouse-moor management) was conducted at the scale of individual ESA agreement boundaries. Thus, the change in the number of birds was calculated for each polygon as for each 1 km square above, and analysed firstly in relation to all ESA tiers. Due to the problems of variability in bird count data, particularly for species such as Lapwing or Curlew, where territories can be large, or individuals change location between years, polygons with an area of less than 1 km^2 were excluded from the analysis, which resulted in all of the non-moorland ESA agreements being excluded from the analysis. To account for differences between the North Peak and South West Peak ESA areas (the former being dominated by large moorland tier units, the latter by smaller units), site was included as a two-level factor. Also included in the analysis were any significant habitat or management variables from the previous analysis of change at the 1 km level, averaged across the land management unit, to account for potentially confounding habitat effects, particularly as the ESA agreements were not randomly distributed in relation to habitat. Data at this larger spatial scale were also used to assess the impact of grouse-moor management intensity upon changes in moorland bird populations. Thus, the ESA polygons were used as a surrogate for shooting estate boundaries or game-keeper beats (which MFF were unable to supply), given that on moorland, it is likely that keepers will vary their activity between different parts of an estate, depending upon grazing management. When interpreting the results, it is worth noting that this may have resulted in a degree of pseudoreplication with respect to the intensity of grouse-moor management, due to the inclusion of repeat data from shooting estates containing multiple ESA agreements.

Further, the use of this measure, which is based on the distribution of muirburn, is dependent on the assumption that the digitized pattern of these muirburn polygons is a useful surrogate for spatial variation in the intensity of grouse-moor management. Independently obtained data on gamekeeper density (no. gamekeepers / area of beat) from six beats within the Peak District correlates significantly with our measure of grouse-moor management for those estates ($r_s = 0.94$, $n = 6$, $P = 0.014$), giving us confidence that our measure used in the analysis is a meaningful one. The change analysis was therefore conducted with the appropriate habitat and management variables, ESA, grouse-moor management and the interaction between the two, as potential predictor variables. Analysis was repeated excluding the habitat and management variables from the 1 km analysis, but as this made no difference to the results relating to ESA and grouse-moor management, it is the former analyses that are presented.

Maps of bird distribution

Maps showing the fine-scale distribution of each species within the Peak District were produced for the eleven species included within the fine-scale analysis. These were derived from the fine-scale bird location data, smoothed to show broad areas of greatest abundance by interpolation, and provide a better representation of bird distribution than simple counts at a 1 km resolution. Equivalent maps were produced for 1990 and 2004, allowing a comparison of change in distribution between the two, produced by simple subtraction of the two images. These provide a rapid assessment of the broad geographical areas within the Peak District that have maintained or lost particular bird populations.

Additionally, a map of predicted probability of occurrence is produced from the fine-scale habitat association models of Table 3.2, to visually assess the locations of apparently suitable habitat for a particular species. By comparing this map with the observed interpolated 2004 distribution of the same species, it is possible to indicate the areas of the Peak District where each model performs best or worst. This is achieved by ranking the discrepancies between the values of both maps and highlighting those areas with the highest and lowest discrepancies. A final assessment of the ability of the fine-scale model to predict bird abundance at a 1 km level, is produced by correlating mean predictive probability of occurrence from within a 1 km square, with observed count from the same square.

Conservation Priority Areas

Six illustrative thematic hotspot maps were produced to identify conservation priority areas for moorland birds within the Peak District. These maps were produced using a number of different conservation priority rankings and weightings to individuals of different species based upon Birds of Conservation Concern listings (see Gregory 2002). Maps were derived from the smoothed fine-scale bird distribution data (see above), in MapInfo using IDW interpolation with a cell size of 0.1 km, a search radius of 1 km, and an exponent value of 2 reducing the weighting of counts with increasing distance from individual cells. This means that individual 0.1 km cells contain a summary of all counts of relevant species occurring within 1 km, and that individuals occurring close to a 0.1 km cell will have a proportionately greater influence upon the cells conservation ranking that those further away.

For map colour coding, six ranked colour categories were produced for each map, with equal proportions of cells present within each category.

Species of Conservation Concern are:

Red list species (excluding Skylark which were only recorded with a 1 km resolution):
- Ring Ouzel
- Linnet
- Twite
- Reed Bunting

Amber list species:
- Meadow Pipit
- Peregrine
- Stonechat
- Tree Pipit
- Short-eared Owl
- Curlew
- Dunlin
- Lapwing
- Snipe

SPA designated species for Dark Peak:
- Golden Plover
- Short-eared Owl
- Peregrine
- Merlin
- Dunlin (qualifying species under article 4.2)

Maps were produced for :
a) total counts of all wader species
b) total counts of all wader species weighted by conservation status (green species weighted by 1, amber species by 3, no red listed wader species present),
c) total count of all red and amber listed birds of conservation concern species (red weighted by 3, amber by 1)
d) Total counts for species for which the South Pennines qualified for SPA status.
e) the total number of red list and SPA designation species
f) relative proportion of red list and SPA designation species (This method scales the influence of individuals of a particular species in proportion to population size, so that one individual of a rare species influences the final hotspot location more than an individual of a commoner species. This corrects the bias caused by more abundant species, such as Golden Plover, contributing more data than rare species, such as Twite).

Table 2.1: Habitat classes within the ESA habitat data. For the fine-scale analysis, bare peat / bare ground and eroding moorland were combined into one category.

Habitat class
Bare peat/ bare ground
Bracken
Cliff/Scree
Cotton grass moorland
Dry bog heather dominated
Dry bog non-heather dominant
Dry dwarf shrub heath, heather dominated
Dry dwarf shrub heath, non-heather dominated
Dry grassland
Eroding moorland
Marshy grassland
Other
Wet bog/wet heath/acid flush
Woodland/Scrub

Table 2.2. ESA prescription names, maximum stocking levels, and reduction in stocking rates. Tiers were combined into three classes for analysis, on the basis of maximum stocking levels (Livestock Grazing Units). Due to their small size, non-moorland ESA prescriptions were excluded (analysis was only conducted on agreements covering > 1km^2).

ESA Tier Name	Max stocking level in summer (LGU)	Stock reduction in winter (LGU)	**LGU class used in bird analysis**
IIb Moorland exclosure (Dark Peak)	0	no stock	**0**
IIa Moorland enhancement (Dark Peak)	0.10	25% removal of summer flock 1 Oct - 28 Feb	**1**
Moorland tier 2 (option 2) (SW Peak)	0.10	no stock 1 Oct - 28 Feb	**1**
Ic Moorland (Dark Peak)	0.15	25% removal of summer flock 1 Nov - 28 Feb	**2**
Moorland tier 1 (part 4) (SW Peak)	0.225	25% removal of summer flock 1 Nov - 28 Feb	**2**

Table 2.3. List of variables used in each analysis derived from satellite imagery, and summary variable names as presented in Tables and Figures in the results section.

Variable	Fine-scale analysis	1 km analyses	LMU analysis
Autocovariate term	X	X	
Elevation	Elev100	MELEVKM	*
Slope	Slope100	MSLOPEKM	*
Distance to nearest watercourse	Water100	STREAMKM	*
Cover of peat		PPEATKM	*
Composition of heather	Cv100	MPCV11KM	*
Composition of grass	AllGra100	GRASSKM	*
Compositon of cotton grass	CotGra100	COTTGRKM	*
Composition of non-heather dwarf shrub	NonHea100	NHEATHKM	*
Vegetation height	Hgt100	MVGHTKM	*
Disturbance score	Disturb	DISTKM	*
Cover of burn polygon	Burn	BURNKM	*
Proximity to in-bye		PINBYEKM	*
Proximity to woodland		PWOODKM	*
Index of grouse-moor management			GMM
ESA grazing Tier			LGUclass

* variables only included if significant in the 1km change analysis.

X Autocovariate term regarded as a nuisance variable, and not presented in the results.

Table 2.4. Results from semi-variograms describing the pattern of spatial autocorrelation in the bird data for each of the three analyses. Presented is the range from each semivariogram, which indicates the distance (m) over which semi-variance is spatially dependent. This is thus an approximation to the distance over which spatial autocorrelation may be a problem in the analysis.

		Range (m)	
Species	Point scale	1 km scale	1 km change scale
Curlew	8219	6372	4451
Dunlin	2496	2042	3345
Golden Plover	3848	4191	24967
Lapwing	2319	1664	2446
Reed Bunting	678	1597	2290
Ring Ouzel	2278	1598	9190
Snipe	2367	1525	3157
Stonechat	3879	1626	No convergence
Wheatear	2673	No convergence	4743
Whinchat	4771	1590	1248
Wren	1764	3183	No convergence
Meadow Pipit	-	6450	6686
Red Grouse	-	8426	5360
Skylark	-	4673	5865

Table 2.5. Pearson's correlation matrix (r) for variables used in the fine-scale analysis, based upon values across 1,200 random points. Correlations which are highly significant (*P* < 0.001) are in bold. For example, the cover of cotton grass is positively correlated with elevation, as it is predominantly found on high moorland plateaus.

	Water100	Slope100	NonHea100	Allgra100	Elev100	Dist100	Cv100	CotGra100	Burn100
Slope100	-0.021								
NonHea100	-0.062	**-0.147**							
Allgra100	0.061	**0.312**	**-0.117**						
Elev100	**-0.233**	**-0.246**	**0.242**	**-0.226**					
Dist100	**0.182**	**0.135**	**-0.141**	0.033	**-0.243**				
Cv100	0.003	**-0.129**	-0.009	**-0.499**	0.085	**-0.138**			
Cotgra100	**-0.093**	**-0.324**	**0.429**	0.041	**0.308**	**-0.128**	**-0.283**		
Burn100	0.088	-0.087	-0.068	**-0.285**	-0.003	**-0.099**	**0.508**	**-0.179**	
Hgt100	0.073	**0.150**	**0.242**	**0.592**	-0.059	-0.122	0.083	**0.222**	0.021

Table 2.6. Pearsons correlation matrix (r) for variables used in the 1 km analysis, based upon values across all squares. Correlations which are highly significant (*P* < 0.001) are in bold. For example the cover of peat is highly positively correlated with elevation (moorland plateaux) and cotton grass.

	MSLOPEKM	MELEVKM	MVGHTKM	GRASSKM	NHEATHKM	COTTGRKM	PINBYETKM	PPEATTKM	PWOODTKM	STREAMKM	MPCV11KM	BURNKM
MELEVKM	**-0.144**											
MVGHTKM	**0.411**	**-0.508**										
GRASSKM	**0.359**	-0.040	**0.300**									
NHEATHKM	-0.108	**0.562**	**-0.556**	0.013								
COTTGRKM	**-0.263**	**0.533**	**-0.381**	**0.243**	**0.618**							
PINBYETKM	**0.126**	**-0.518**	**0.545**	0.044	**-0.515**	**-0.453**						
PPEATTKM	**-0.327**	**0.759**	**-0.605**	-0.054	**0.561**	**0.625**	**-0.623**					
PWOODTKM	**0.252**	**-0.500**	**0.290**	-0.100	**-0.404**	**-0.384**	**0.335**	**-0.474**				
STREAMKM	-0.048	**-0.445**	**0.255**	0.053	**-0.265**	**-0.250**	**0.295**	**-0.434**	**0.187**			
MPCV11KM	**-0.213**	**0.210**	**-0.403**	**-0.517**	**0.202**	**-0.161**	**-0.320**	**0.283**	**-0.223**	-0.058		
BURNKM	**-0.149**	0.032	**-0.240**	**-0.403**	0.061	**-0.185**	**-0.209**	0.129	**-0.146**	0.014	**0.791**	
DISTKM	**0.159**	**-0.286**	**0.281**	0.052	**-0.192**	**-0.210**	**0.221**	**-0.345**	**0.265**	**0.272**	**-0.191**	**-0.162**

3. RESULTS

3.1. Fine-scale index of habitat preference using ESA habitat data

Using the ESA habitat data, in which the Peak District was divided into distinct polygons of 14 habitat types (Table 2.1), an index of habitat preference was determined for each species, based on the proportion of sightings of that species recorded in each habitat (Table 3.1). Because this analysis is simple, with few assumptions (see methods) we have been able to conduct the analysis for a wider range species with fewer data, excluding only those with 20 sightings or fewer. However, results are subject to limited interpretation, because we have not attempted to account for potentially confounding topographical and management effects. Thus, associations between bird abundance and habitat may not be causative.

Each of the wader species showed strong habitat associations. Both Dunlin and Golden Plover were associated with cotton grass moorland, eroding moorland and more weakly with non-heather dominated dry bog habitats. Interestingly Dunlin shows a strong negative association with bare ground. Of the four remaining waders, associations with dry grassland, marshy grassland (apart from Common Sandpiper) and wet bog were most notable, although the strongest determinants of abundance for Common Sandpiper were woodland and ‘other’ habitats (presumably riparian).

Raptors tended to show the weakest habitat preferences, because many of the sightings were of flying birds, and were therefore less readily associated with particular habitat classes. Kestrel appeared to hunt most over bracken, grassland and wet flush habitats, whilst peregrine strongly associated with cliffs.

The results from the passerines show a consistent strong association with bracken cover for each of the ten species, whilst many also show weaker associations with dry heather dominated heath, dry grassland and wet bog vegetation. Five species were also strongly associated with woodland or scrub; Linnet, Mistle Thrush, Stonechat, Tree Pipit and Willow Warbler.

3.2. Fine-scale habitat associations using satellite derived habitat data

Models comparing habitat and management variables between bird locations and random points were produced for 11 species (Table 3.2). Models for Dunlin failed to converge when including quadratic terms, probably due to the relatively small sample size (76 locations) compared with other species, and so analysis was conducted using linear terms only for this species. The final parameter estimates for five species, Golden Plover, Reed Bunting, Ring Ouzel, Snipe and Dunlin exhibited high predictive power (AUC > 0.8), whilst for Curlew and Stonechat model averaged parameter estimates produced weaker predictions (AUC < 0.7). Predicted probabilities at bird locations were significantly greater than at random points for each species (Table 3.6), indicating all models had some predictive power. The autocovariate term (not listed) received a high selection probability in each case, indicating that the distributions of all species were spatially aggregated. When averaged at the 1 km level, the mean probability of occurrence from the fine-scale models was positively correlated with observed abundance within each 1 km square six species (Curlew, Golden Plover, Reed Bunting, Snipe, Whinchat and Dunlin; Figs 3.5). However, the strongest relationship, for Golden Plover, only explained about one third of the variation in count between 1 km squares.

Topography parameters, particularly altitude and slope, explained a high percentage of variation in species distribution. Slope was a significant correlate of distribution for eight species, and the most consistent environmental component in this analysis. The distributions of Curlew, Golden Plover, Snipe and Reed Bunting were negatively correlated with slope, whilst Ring Ouzel, Stonechat, Whinchat and Wren were most frequently encountered on sloping ground. The locations of six species were significantly influenced by altitude. Golden Plover were associated with high altitudes, whilst Lapwing, Reed Bunting, Stonechat, Whinchat and Wren were each most frequent at lower altitudes. The third topographical term, distance to stream, was positively correlated with Golden Plover and Stonechat occurrence, but negatively with Reed Bunting and Wren; the latter two species being most closely associated with streams (Fig 3.1).

Vegetation height was the second most important predictor variable of species distribution, with a high selection probability for seven of the eleven species examined. Golden Plover distribution was strongly negatively correlated with height, i.e. closely associated with short swards, whilst Reed Bunting, Snipe and Wren were each associated with tall vegetation, such as found in bracken and marshy grassland (cf. Table 3.1). Quadratic correlations between the distribution of Curlew and Lapwing and vegetation height indicate that these species were most frequent in areas of moderately tall vegetation (40 cm height), whilst Whinchat showed a very strong association with vegetation between 25-30cm in height.

Of the measures of vegetation composition, the most commonly significant term was cotton grass cover, strongly associated with peat soils (see methods). In most cases the association between bird abundance and cotton grass was positive, although for Curlew it was the weakest association of the four vegetation covers suggesting that Curlew tend to avoid cotton grass, and for Ring Ouzel the form of the quadratic correlation suggests avoidance of very high cover (Table 3.2, Fig 3.1). Of the other vegetation types, heather tended to be associated with a greater probability of encounter of Curlew, Wren and Reed Buntings. The occurrences of Snipe and Wheatear in particular were positively correlated with grass cover (i.e. acid grassland habitats consisting of *Molinia*, *Nardus* and fine-leaved grasses). Curlew, Reed Bunting and Wren each showed strong associations for areas with intermediate levels of grass cover (Fig. 3.1), although due to the strong negative correlation between grass and heather cover (Table 2.5), and the association of each of these species with heather, and therefore should be interpreted cautiously as the form of these relationships may be affected by the heather correlations. Curlew, Whinchat and Wheatear were all positively associated with non-heather dwarf shrub vegetation, whilst Ring Ouzel and Stonechat exhibited quadratic correlations with the cover of this habitat indicative of a preference for 20 – 30 % cover. The positive parameter estimate of Wren with non-heather dwarf shrub cover is not indicative of a strong association, as this represents the smallest parameter estimate for this species with any of the four vegetation covers.

Of the management variables, correlations between distribution and disturbance by vicinity to footpaths had a high selection probability for five species. These were indicative of an avoidance of areas of high disturbance for four waders; Curlew, Golden

Plover, Lapwing and Snipe. In contrast, Wren were frequently encountered along footpaths. The presence of heather burning appeared a relatively weak determinant of bird occurrence, only strongly correlated with the distribution of Reed Bunting, which appeared to avoid the burn polygons.

3.3. Large-scale habitat associations using satellite derived habitat data

Models predicting the number of birds present in each square were produced for 14 species, including Meadow Pipit, Skylark and Red Grouse for which finer-scale analyses were not possible (Table 3.3). These models had much lower predictive power than the fine-scale models, with the correlation between observed and predicted abundance across the test squares being better than chance for six species only (Curlew, Golden Plover, Reed Bunting, Snipe, Red Grouse and Skylark; Table 3.5). As with the fine-scale associations, models predicting Dunlin abundance would only converge with the quadratic terms excluded. Similarly, the autocovariate term was strongly related to abundance in each case.

Topographical variables, particularly slope and altitude were the most consistent terms in the models across the 14 species, being significant in 9 and 6 models respectively. Curlew, Lapwing, Golden Plover, Snipe, Dunlin and Reed Bunting each showed negative correlations between abundance and mean slope i.e. associated with flatter ground, whilst Red Grouse, Wren and Ring Ouzel were associated with sloping ground. Golden Plover were again associated with high elevations, whilst Curlew, Lapwing, Reed Bunting, Whinchat and Wren were more confined to the moorland fringe at lower altitudes. The similarity in the relationships between bird distribution and slope and altitude across these two analysis scales increases our confidence in these effects. Of the other topographical terms analysed at this scale, distance to stream was negatively correlated with the abundance of Dunlin but weakly positively with that of Red Grouse, whilst both Whinchat and Snipe showed reduced abundance on peat habitats.

In contrast to the fine-scale analysis, vegetation height was an important predictor variable of species abundance for only two species with the abundance of both Whinchat and Curlew showing quadratic correlations with height, peaking at 25-30 cm, and 30-35 cm respectively.

For vegetation composition, as with the fine-scale analysis, Golden Plover and Dunlin showed strong associations with the cover of cotton grass, whilst Skylark and Meadow Pipits were more weakly positively correlated with this habitat. Curlew, Snipe and Reed Bunting were each associated with grass cover, reflecting to some degree the finer-scale associations, whilst Red Grouse show a quadratic negative correlation with grass cover. Both Skylark and Meadow Pipit abundances were negatively correlated with heather cover, whilst Curlew was positively related to heather. Wren and Stonechat abundance was greatest in squares with intermediate levels of non-heather dwarf shrub cover (c. 20-30 % cover for both).

Of the management variables, correlations between distribution and disturbance had a high selection probability for five species, with a negative association for Curlew, a quadratic relationship for Ring Ouzel and Red Grouse suggestive of some reduction of abundance at high disturbance levels, and a positive correlation between recreational disturbance and both Wren and Whinchat abundance. The cover of burn polygons within a square was negatively correlated with the abundance of Reed Bunting and Meadow Pipits, and positively correlated with both Ring Ouzel and Wren abundances. Red Grouse showed reduced abundance on squares in close proximity to enclosed farmland, whilst Skylark were most abundant in such areas. Wren abundance was strongly negatively associated with the cover of woodland in surrounding squares, but Meadow Pipits were more common in squares with woodland surrounding them.

3.4. Analysis of change using satellite derived habitat data

Analyses of change were not conducted for Wren and Stonechat, due to the preponderance of squares where abundance had increased from zero, making the analysis of large-scale associations (Table 3.4) equivalent to the analysis of change. An indication of the proportion of squares where the population increased or declined can be obtained from Figure 3.3, where squares which were colonized in 2004 (i.e. the species was not recorded from that square in 1990) and abandoned (i.e. the species was recorded from that square in 1990 but not 2004) are separated from other squares with increasing, stable or declining population sizes. For only one species, Curlew, did the model of change exhibit predictive power, producing a significant positive correlation between observed and predicted change across the test data (Table 3.5). Therefore, the results for most species appear to have little generality, and should therefore be regarded with considerable caution.

The most frequent correlates of change were in relation to vegetation cover. Four species, Reed Bunting, Snipe, Whinchat and Dunlin were more likely to decline in squares with a high cover of non-heather dwarf shrubs. Similarly, changes in Whinchat and Wheatear abundance were negatively correlated with heather cover, whilst Curlew appeared to decline the most on squares with an intermediate level of heather cover, with the largest increases above about 50 % heather cover (Fig. 3.3). Changes in the abundance of Dunlin were negatively correlated with grass cover, and that of Reed Bunting negatively correlated with cotton grass cover. Golden Plovers were most likely to increase on cotton grass dominated squares, but declined in areas of other habitat types. The only species for which change was correlated with vegetation height were Curlew and Dunlin, both of which showed the relatively weak increases in abundance on sites with tall vegetation.

In relation to topography, Dunlin and Wheatear each showed the greatest declines on steeply sloping terrain, whilst Ring Ouzel and Whinchat declines were correlated with elevation. The former declining most on low altitude squares, and the latter declining most at intermediate elevations around 350 m. Mean distance to stream and the cover of peat were not correlated with change for any species.

Of the management variables, the cover of burn polygons was associated with change for three species. Skylark were most likely to decline in squares with a high intensity of burning, whilst Dunlin and Ring Ouzel showed an increase in squares with a high cover of managed burns. Curlew increased in km squares with high levels of disturbance, mainly moorland fringe areas and the Derwent valley, which are popular for recreation, although this contrasts with their fine-scale avoidance of areas close to footpaths (Table 3.2). Finally, the cover of woodland, and enclosed grassland (inbye) only showed weak relationships with change in species abundance with Skylark and Dunlin, both being influenced only by a small number of points (Fig. 3.3).

3.5. Analysis of change across moorland management units

For most species, there were no significant differences between the three different ESA grazing tier classes and the changes in abundance of moorland birds, or apparent effects of the intensity of grouse-moor management in changes in abundance at this scale. The only exceptions were for Dunlin and Golden Plover, with a negative correlation between the change in Dunlin abundance and grouse-moor management (Dunlin, $F_{1,26}$ = 5.25, *P* = 0.030), and significant interactions between tier class and intensity of grouse-moor management for both species (Golden Plover, $F_{2,55}$ = 3.29, *P* = 0.045; Dunlin, $F_{2,26}$ = 5.25, *P* = 0.010). Although both are weak, upon examination of these relationships (Fig 3.4) they indicate similar patterns of change; weak negative associations between the change in abundance and grouse-moor management under LGU classes zero (exclosure) and two (moorland tier), whilst under moorland enhancement or Tier 2, populations were more likely to increase when the intensity of grouse-moor management was greatest. However, the only line that differed significantly from zero was the negative correlation between the change in Dunlin abundance and intensity of grouse-moor management under 2 LGU for Dunlin (coefficient = -4.6 ± 2.3, *P* = 0.046). These relationships therefore suggest that overall, there is no strong evidence for changes in both Dunlin and Golden Plover populations in relation to either grazing or grouse-moor management, apart from a tentative reduction in Dunlin on heavily managed (i.e. high grazing and intensity of grouse-moor management) ground.

It is possible that habitat effects already detected in the 1 km analysis may contribute to some of the variation in population change at this larger, moorland management unit scale. To assess these, we constructed some additional models of change, using mean values for the significant terms for each species, from Table 3.4. These indicate that effects of mean vegetation height on Curlew (0.17 ± 0.064, $F_{1,62}$ = 7.02, *P* = 0.010), cotton grass cover on Golden Plover (0.033 ± 0.010, $F_{1,60}$ = 10.80, *P* = 0.0017), altitude on Ring Ouzel (0.011 ± 0.0052, $F_{1,37}$ = 4.66, *P* = 0.038), and both altitude (0.019 ± 0.0099, $F_{1,28}$ = 4.71, *P* = 0.039), and heather cover (–0.043 ± 0.018, $F_{1,28}$ = 6.09, *P* = 0.020) on Whinchat, remained significant at this large-scale. However, with the exception of Golden Plover, levels of statistical significance were relatively low ($P < 0.05$).

3.6. Maps of distribution and the accuracy of model predictions

Visual depictions of species distributions in 1990 and 2004, and the change in abundance, are mapped separately for each species (Fig 3.5 a-k). We smooth occurrence data using interpolation (see methods) and thereby show areas of greatest abundance (red: high, yellow: low), and change (blue: decline, red: increase) for each species. In addition, these maps are combined with a map of predicted distribution based on the fine-scale models of Table 3.2, to provide a visual impression of the geographical locations where each model tends to perform best, or worst, whilst the scatter graph shows the strength of the correlation between mean probability of occurrence predicted across the 1 km level from the fine-scale models, and observed abundance.

Across the large plateaux of the western Peak District of Kinder, Bleaklow and Saddleworth Moors, Golden Plover, and in some areas, Snipe and Dunlin, appear to have increased in abundance. However, in these areas, populations of Wheatear have declined, and Ring Ouzels appear to have been almost lost, despite much apparently suitable habitat remaining. In contrast, the north-eastern moors of Langsett, Howden and Broomhead appear to have suffered some declines in the abundance of Golden Plover and Dunlin, although the survey of these moors in 2005 (Appendix 1) suggests that Golden Plover have increased in some of these areas, so some of these apparent changes in population may be due to annual variation. It is in these eastern moors where Curlew numbers in particular have increased, and Ring Ouzels maintained a foothold. The Eastern Moors south of Derwent and Burbage have experienced the greatest increases in a number of passerines that have shown widespread population increases within the Peak District, in particular Reed Bunting, Stonechat and Whinchat, although some of the wader species, particularly Golden Plover, Snipe and Lapwing have declined in parts of these areas. A similar pattern is apparent in the moors to the west and south of Buxton, with increasing populations of several passerine species, such Wheatear, Stonechat and Reed Bunting. However, Golden Plover and Ring Ouzel numbers in these areas appear to have fallen. In this area, Snipe numbers appear to have increased, particularly around The Roaches.

3.7. Conservation Priority Area Maps

To delineate and highlight the areas of greatest conservation importance for moorland birds within the Peak District, we produced hotspots maps for waders and species of conservation concern with different weightings (see Figs. 3.6a-f). A number of different criteria have been used, based either on the breeding wader assemblage, which are a particularly important component of the moorland bird community, national conservation priority listings, and qualifying species for the South Pennines SPA.

For wader conservation it is the northern and central moors around the Woodhead catchment, Derwent and Howden Moors that consistently exhibit high conservation priority irrespective of the different weightings given to different species (Fig 3.6a, b). These moorlands are particularly important for Golden Plover and Dunlin, although numbers of these species appear to have declined in some northern and central areas (see interpolation maps - Fig 3.5). Furthermore, the importance of the moorland fringe is highlighted by Fig. 3.6b which gives greater ranking to amber list species, such as Snipe, Lapwing and Curlew, that favour some of these lower altitude, grassy, areas (Tables 3.1, 3.3). Based on national conservation listings which give red-list species a high weighting, it is these lower altitude southerly moors east of The Roaches, and the Eastern Moors south west of Sheffield which appear to be the most important (Fig 3.6a), largely due to significant Linnet and Reed Bunting populations in these areas. These listings favour such species that were once nationally widespread but which have declined across lowland farmland, and therefore may not represent the conservation priorities within a typical area of moorland. A better representation of the likely conservation priority areas for the suite of moorland birds important to the Peak District can be obtained from Fig. 3.6d, which illustrates hotspots based on the qualifying species for the SPA, and highlights the importance of parts of the northern plateaux areas such as Saddleworth, and some of the central moors, such as Kinder, Derwent and Howden. A combination of these two species lists and weightings are presented in Figures 3.6e and f, based on different weightings. The latter is perhaps the most useful, weighting each sighting according to the proportion of the Peak District moorland population that it represents; thus, one Twite provides a greater weighting than one Linnet. Overall, this indicates the northern moors west of Huddersfield, the central moors around Kinder and Howden, and the Eastern moors to the west of Chesterfield as being of greatest overall significance. The northern and central areas appear to be largely for moorland birds important to the SPA, and the eastern moors for formerly widespread species that have now declined in lowlands and for which the uplands hold increasingly important populations. Thus, depending on how the different bird species are ranked, most areas are highlighted as hotspots for some species, although given that the entire moorland area is within a SSSI and SPA for breeding bird interest, this is perhaps unsurprising.

Table 3.1a. Summary of the analysis of habitat preferences for waders (top) and raptors (bottom). The left column for each species indicates the number of sightings in each caterogy, and the right column, the preference index for that habitat (see text). Species with significant selection are in bold.

Vegetation category	**Common Sandpiper**		**Curlew**		**Dunlin**		**Golden Plover**		**Lapwing**		**Snipe**	
Bare peat/ bare ground	**0**	**-1.00**	**3**	**-0.74**	**1**	**-0.33**	**37**	**0.29**	**0**	**-1.00**	**0**	**-1.00**
Bracken	**1**	**-0.09**	**25**	**-0.14**	**0**	**-1.00**	**6**	**-0.70**	**8**	**0.07**	**4**	**-0.20**
Cliff/Scree	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**
Cotton grass moorland	**3**	**-0.12**	**112**	**0.03**	**41**	**0.60**	**245**	**0.40**	**15**	**-0.19**	**18**	**-0.02**
Dry bog heather dominated	**0**	**-1.00**	**126**	**-0.06**	**10**	**-0.16**	**96**	**-0.20**	**15**	**-0.32**	**6**	**-0.62**
Dry bog non-heather dominant	**5**	**0.30**	**31**	**-0.41**	**12**	**0.24**	**153**	**0.34**	**3**	**-0.67**	**4**	**-0.54**
Dry dwarf shrub heath, heather dominated	**0**	**-1.00**	**163**	**0.17**	**0**	**-1.00**	**36**	**-0.53**	**25**	**0.02**	**17**	**-0.10**
Dry dwarf shrub heath, non-heather dominated	**0**	**-1.00**	**76**	**0.04**	**2**	**-0.55**	**27**	**-0.45**	**10**	**-0.19**	**12**	**-0.03**
Dry grassland	**8**	**0.42**	**107**	**0.08**	**1**	**-0.80**	**31**	**-0.49**	**45**	**0.41**	**37**	**0.39**
Eroding moorland	**1**	**-0.15**	**12**	**-0.51**	**9**	**0.42**	**121**	**0.53**	**2**	**-0.59**	**2**	**-0.54**
Marshy grassland	**1**	**-0.30**	**82**	**0.23**	**0**	**-1.00**	**18**	**-0.48**	**32**	**0.50**	**30**	**0.53**
Other	**7**	**0.94**	**3**	**-0.32**	**0**	**-1.00**	**4**	**-0.20**	**0**	**-1.00**	**1**	**-0.03**
Wet bog/wet heath/acid flush	**1**	**0.17**	**29**	**0.19**	**0**	**-1.00**	**5**	**-0.60**	**5**	**0.10**	**7**	**0.33**
Woodland/Scrub	**1**	**0.59**	**2**	**-0.56**	**0**	**-1.00**	**1**	**-0.75**	**1**	**-0.19**	**1**	**-0.12**
TOTAL sightings / significance	**28**	**_P_ < 0.001**	**771**	**_P_ < 0.001**	**76**	**_P_ < 0.001**	**780**	**_P_ < 0.001**	**161**	**_P_ < 0.001**	**139**	**_P_ < 0.001**

Vegetation category	**Kestrel**		Merlin		**Peregrine**		Raven		Short-eared Owl	
Bare peat/ bare ground	**0**	**-1.00**	0	-1.00	**0**	**-1.00**	2	0.00	0	-1.00
Bracken	**15**	**0.36**	3	0.24	**1**	**-0.23**	2	-0.25	1	-0.20
Cliff/Scree	**0**	**-1.00**	0	-1.00	**2**	**0.92**	1	0.69	0	-1.00
Cotton grass moorland	**19**	**-0.08**	3	-0.32	**5**	**0.00**	10	-0.02	3	-0.23
Dry bog heather dominated	**24**	**-0.11**	13	0.25	**3**	**-0.39**	9	-0.22	12	0.30
Dry bog non-heather dominant	**10**	**-0.22**	5	0.10	**5**	**0.17**	6	-0.10	2	-0.25
Dry dwarf shrub heath, heather dominated	**23**	**-0.03**	11	0.26	**6**	**0.04**	10	-0.07	2	-0.45
Dry dwarf shrub heath, non-heather dominated	**20**	**0.15**	2	-0.32	**6**	**0.28**	13	0.30	3	-0.03
Dry grassland	**33**	**0.26**	2	-0.43	**6**	**0.16**	13	0.18	6	0.19
Eroding moorland	**2**	**-0.59**	1	-0.35	**3**	**0.26**	6	0.24	0	-1.00
Marshy grassland	**9**	**-0.10**	2	-0.18	**0**	**-1.00**	2	-0.44	5	0.36
Other	**1**	**-0.11**	0	-1.00	**0**	**-1.00**	0	-1.00	0	-1.00
Wet bog/wet heath/acid flush	**7**	**0.25**	1	-0.04	**0**	**-1.00**	3	0.21	1	0.06
Woodland/Scrub	**1**	**-0.20**	0	-1.00	**0**	**-1.00**	0	-1.00	0	-1.00
TOTAL sightings / significance	**164**	**_P_ = 0.001**	43	_P_ = 0.343	**37**	**_P_ = 0.000**	77	_P_ = 0.140	35	_P_ = 0.279

Table 3.1b. Summary of the analysis of habitat preferences for passerines. The left column for each species indicates the number of sightings in each caterogy, and the right column, the preference index for that habitat (see text). Species with significant selection are in bold.

Vegetation category	**Linnet**		**Mistle Thrush**		**Reed Bunting**		**Ring Ouzel**		**Stonechat**		**Tree pipit**	
Bare peat/ bare ground	**0**	**-1.00**	**1**	**-0.14**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**
Bracken	**14**	**0.44**	**8**	**0.57**	**21**	**0.50**	**15**	**0.58**	**12**	**0.50**	**10**	**0.71**
Cliff/Scree	**0**	**-1.00**	**2**	**0.88**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**	**0**	**-1.00**
Cotton grass moorland	**3**	**-0.70**	**2**	**-0.55**	**5**	**-0.63**	**1**	**-0.85**	**2**	**-0.73**	**0**	**-1.00**
Dry bog heather dominated	**16**	**-0.18**	**4**	**-0.40**	**21**	**-0.18**	**7**	**-0.42**	**8**	**-0.36**	**2**	**-0.57**
Dry bog non-heather dominant	**1**	**-0.85**	**1**	**-0.66**	**2**	**-0.77**	**1**	**-0.80**	**2**	**-0.63**	**0**	**-1.00**
Dry dwarf shrub heath, heather dominated	**38**	**0.34**	**4**	**-0.31**	**32**	**0.13**	**27**	**0.32**	**23**	**0.25**	**4**	**-0.20**
Dry dwarf shrub heath, non-heather dominated	**13**	**0.06**	**5**	**0.04**	**8**	**-0.30**	**21**	**0.43**	**7**	**-0.09**	**3**	**-0.10**
Dry grassland	**11**	**-0.15**	**12**	**0.33**	**24**	**0.11**	**17**	**0.22**	**18**	**0.25**	**9**	**0.31**
Eroding moorland	**0**	**-1.00**	**1**	**-0.42**	**0**	**-1.00**	**0**	**-1.00**	**1**	**-0.63**	**0**	**-1.00**
Marshy grassland	**15**	**0.28**	**8**	**0.40**	**34**	**0.51**	**3**	**-0.35**	**12**	**0.32**	**7**	**0.45**
Other	**4**	**0.61**	**0**	**-1.00**	**3**	**0.41**	**1**	**0.17**	**0**	**-1.00**	**0**	**-1.00**
Wet bog/wet heath/acid flush	**8**	**0.43**	**1**	**-0.13**	**13**	**0.52**	**0**	**-1.00**	**4**	**0.26**	**2**	**0.33**
Woodland/Scrub	**3**	**0.44**	**2**	**0.62**	**1**	**-0.20**	**0**	**-1.00**	**4**	**0.65**	**3**	**0.78**
TOTAL sightings / significance	**126**	***P* = 0.000**	**51**	***P* = 0.000**	**164**	***P* = 0.000**	**93**	***P* = 0.000**	**93**	***P* = 0.000**	**40**	***P* = 0.000**

Vegetation category	**Wheatear**		**Whinchat**		**Wren**		**Willow Warbler**	
Bare peat/ bare ground	**2**	**-0.14**	**0**	**-1.00**	**20**	**-0.26**	**0**	**-1.00**
Bracken	**6**	**0.16**	**28**	**0.73**	**143**	**0.44**	**32**	**0.62**
Cliff/Scree	**3**	**0.85**	**0**	**-1.00**	**7**	**0.38**	**0**	**-1.00**
Cotton grass moorland	**6**	**-0.39**	**2**	**-0.75**	**36**	**-0.66**	**0**	**-1.00**
Dry bog heather dominated	**5**	**-0.57**	**4**	**-0.65**	**183**	**-0.13**	**12**	**-0.46**
Dry bog non-heather dominant	**6**	**-0.23**	**1**	**-0.82**	**29**	**-0.62**	**1**	**-0.89**
Dry dwarf shrub heath, heather dominated	**22**	**0.19**	**21**	**0.15**	**308**	**0.23**	**44**	**0.25**
Dry dwarf shrub heath, non-heather dominated	**9**	**-0.01**	**8**	**-0.08**	**213**	**0.29**	**15**	**-0.03**
Dry grassland	**18**	**0.21**	**18**	**0.20**	**192**	**0.11**	**28**	**0.15**
Eroding moorland	**11**	**0.39**	**0**	**-1.00**	**15**	**-0.61**	**0**	**-1.00**
Marshy grassland	**7**	**0.02**	**16**	**0.40**	**75**	**-0.07**	**19**	**0.24**
Other	**0**	**-1.00**	**1**	**0.12**	**9**	**-0.05**	**1**	**-0.15**
Wet bog/wet heath/acid flush	**6**	**0.40**	**4**	**0.21**	**51**	**0.21**	**13**	**0.49**
Woodland/Scrub	**0**	**-1.00**	**0**	**-1.00**	**19**	**0.23**	**11**	**0.74**
TOTAL sightings / significance	**101**	***P* = 0.000**	**103**	***P* = 0.000**	**1300**	***P* = 0.000**	**176**	***P* = 0.000**

Table 3.2a. Fine-scale model outputs produced using model averaging. Estimates (± SD) are produced for each species, along with the selection probability (0-100) that each term receives from model averaged outputs. Significant terms are those with a weight outside the 95 % range of a random variable (see text) are in bold. A positive estimate means that a species distribution is positively correlated with that parameter, such as the association between Golden Plovers and cotton grass. Estimate values can only be compared within one parameter between species, e.g. the association with cotton grass is stronger for Dunlin, than for Golden Plover within these models – however the strength of parameters within each model for one species are determined by the weights, e.g. for Golden Plover slope has a stronger predictive power than disturbance.

	Curlew		Dunlin		Golden Plover		Lapwing		Snipe	
Intercept	**-6.29± 0.69**	**100**	**-5.00± 2.11**	**100**	**-0.03± 0.85**	**100**	**-2.23± 1.91**	**100**	**-4.68± 1.72**	**100**
AllGra100	**5.17± 0.68**	**100**	-0.31± 1.20	12	0.00± 0.07	2	0.04± 0.25	5	**3.28± 0.89**	**100**
CotGra100	**2.67± 1.07**	**94**	**11.44± 4.72**	**96**	**3.75± 0.76**	**100**	0.09± 0.64	4	0.46± 1.37	14
Cv100	**4.65± 0.77**	**100**	0.21± 1.12	10	0.01± 0.10	2	-0.01± 0.16	2	0.04± 0.38	2
Slope100	**-0.05± 0.02**	**98**	-0.03± 0.07	18	**-0.07± 0.02**	**99**	-0.01± 0.02	9	**-0.08± 0.06**	**79**
NonHea100	**3.38± 0.83**	**100**	0.56± 2.17	12	0.01± 0.12	2	0.04± 0.41	3	0.19± 0.90	7
Hgt100	**0.07± 0.02**	**100**	0.00± 0.03	7	**-0.12± 0.02**	**100**	**0.18± 0.05**	**100**	**0.11± 0.06**	**87**
Elev100	0.00± 0.00	3	0.00± 0.00	11	**0.00± 0.00**	**100**	**-0.01± 0.00**	**98**	0.00± 0.00	2
Water100	0.00± 0.00	3	0.00± 0.00	8	**0.00± 0.00**	**58**	0.00± 0.00	2	0.00± 0.00	3
Burn	0.04± 0.32	3	-0.15± 1.13	8	0.00± 0.06	2	0.26± 1.65	6	0.00± 0.17	2
Disturb	**-0.90± 0.74**	**67**	0.49± 2.10	11	**-0.82± 1.09**	**40**	**-1.66± 2.16**	**45**	**-0.78± 1.56**	**25**
AllGra100^2	**-0.29± 0.07**	**100**			0.00± 0.02	4	0.02± 0.09	9	-0.01± 0.05	5
CotGra100^2	0.00± 0.01	2			0.00± 0.01	2	0.01± 0.06	7	0.00± 0.01	2
Cv100^2	0.00± 0.02	3			0.00± 0.01	2	-0.01± 0.04	5	0.00± 0.03	3
Slope100^2	0.00± 0.01	1			0.00± 0.01	2	0.00± 0.02	3	0.03± 0.09	12
NonHea100^2	0.00± 0.02	4			0.00± 0.01	2	0.00± 0.04	4	0.00± 0.02	2
Hgt100^2	**-0.06± 0.05**	**61**			-0.01± 0.03	10	**-0.23± 0.15**	**77**	0.00± 0.01	2
Elev100^2	-0.01± 0.02	6			**-0.22± 0.08**	**95**	**-0.47± 0.35**	**75**	-0.08± 0.16	26
Water100^2	0.00± 0.00	2			0.00± 0.01	3	0.00± 0.01	3	-0.01± 0.04	7
Burn^2	0.00± 0.03	3			0.00± 0.00	2	-0.01± 0.07	5	0.00± 0.01	1
Disturb^2	0.00± 0.01	3			**-0.07± 0.08**	**55**	0.00± 0.04	6	**-0.23± 0.24**	**57**
Random 95%		10		40		12		27		37

Table 3.2b. Fine-scale model outputs produced using model averaging. Estimates (± SD) are produced for each species, along with the selection probability (0-100) that each term receives from model averaged outputs. Significant terms are those with a weight outside the 95 % range of a random variable (see text) are in bold. A positive estimate means that a species distribution is positively correlated with that parameter, such as the association between Golden Plovers and cotton grass. Estimate values can only be compared within one parameter between species, e.g. the association with cotton grass is stronger for Dunlin, than for Golden Plover within these models – however the strength of parameters within each model for one species are determined by the weights, e.g. for Golden Plover slope has a stronger predictive power than disturbance.

	Reed Bunting		Ring Ouzel		Stonechat		Whinchat		Wheatear		Wren	
Intercept	**-1.19± 2.44**	**100**	**-1.65± 1.62**	**100**	**2.32± 2.06**	**100**	**6.74± 2.81**	**100**	**-4.14± 1.88**	**100**	**-3.67± 0.64**	**100**
AllGra100	**6.02± 2.92**	**87**	0.03± 0.41	4	0.00± 0.10	1	0.18± 0.95	7	**3.72± 2.41**	**88**	**4.69± 0.62**	**100**
CotGra100	**10.11± 3.43**	**99**	**0.49± 1.95**	**9**	**11.27± 3.20**	**100**	0.36± 1.57	8	-0.10± 0.71	6	**5.45± 0.82**	**100**
Cv100	**8.58± 3.97**	**88**	1.21± 2.73	22	0.05± 0.38	2	0.36± 1.59	8	1.37± 2.43	32	**6.65± 0.86**	**100**
Slope100	**-0.04± 0.05**	**44**	**0.17± 0.05**	**100**	**0.20± 0.05**	**100**	**0.07± 0.08**	**53**	0.02± 0.04	26	**0.10± 0.01**	**100**
NonHea100	0.01± 0.17	0	**0.16± 0.91**	**5**	0.02± 0.39	1	**3.88± 4.18**	**55**	**5.83± 4.09**	**79**	**3.64± 0.88**	**100**
Hgt100	**0.15± 0.05**	**100**	0.00± 0.02	4	0.00± 0.02	4	**0.00± 0.01**	**2**	0.00± 0.01	4	**0.01± 0.01**	**28**
Elev100	**-0.02± 0.00**	**100**	0.00± 0.00	24	**-0.02± 0.00**	**100**	**-0.02± 0.01**	**100**	0.00± 0.00	4	**-0.01± 0.00**	**100**
Water100	**0.00± 0.00**	**12**	0.00± 0.00	25	**0.00± 0.00**	**43**	0.00± 0.00	3	0.00± 0.00	4	**-0.01± 0.00**	**100**
Burn	**-0.38± 1.07**	**14**	0.00± 0.24	2	0.00± 0.14	1	0.09± 1.43	3	0.02± 0.37	4	-0.01± 0.07	2
Disturb	0.00± 0.10	0	0.63± 1.39	23	0.00± 0.12	1	0.21± 0.79	10	-0.02± 0.23	4	**1.11± 0.41**	**95**
AllGra100^2	**-0.78± 0.36**	**90**	0.00± 0.04	2	0.00± 0.03	1	0.00± 0.03	2	0.00± 0.04	5	**-0.24± 0.05**	**100**
CotGra100^2	**-0.10± 0.16**	**37**	**-0.25± 0.17**	**80**	0.00± 0.02	1	-0.01± 0.09	4	0.03± 0.10	13	0.00± 0.02	5
Cv100^2	**0.03± 0.11**	**9**	-0.10± 0.24	19	0.00± 0.03	1	**-0.06± 0.16**	**18**	0.01± 0.06	5	**-0.05± 0.08**	**32**
Slope100^2	0.00± 0.00	0	**-0.45± 0.32**	**77**	0.01± 0.06	3	0.00± 0.04	2	-0.01± 0.06	8	-0.01± 0.03	7
NonHea100^2	0.00± 0.01	0	**-0.21± 0.29**	**43**	**-0.04± 0.13**	**10**	**-0.55± 0.27**	**92**	0.00± 0.02	5	**-0.02± 0.05**	**21**
Hgt100^2	0.00± 0.00	0	0.00± 0.02	2	0.01± 0.04	3	**-0.26± 0.46**	**29**	-0.01± 0.05	6	0.00± 0.01	5
Elev100^2	**-0.78± 0.45**	**84**	0.00± 0.03	2	**-1.16± 0.33**	**100**	0.00± 0.02	3	0.00± 0.02	3	**-0.17± 0.05**	**99**
Water100^2	0.00± 0.00	0	0.01± 0.06	4	**0.02± 0.09**	**10**	0.00± 0.06	3	0.00± 0.02	4	**0.18± 0.03**	**100**
Burn^2	**-0.01± 0.03**	**7**	0.00± 0.02	3	0.00± 0.01	1	0.00± 0.03	3	0.00± 0.03	4	0.00± 0.01	3
Disturb^2	0.00± 0.00	0	0.01± 0.06	7	0.00± 0.01	1	0.00± 0.03	2	0.00± 0.04	5	0.00± 0.01	2
Random 95%		5		26		4		16		39		11

Table 3.3a. Model outputs for associations at the 1km scale (see Table 3.2 for legend).

	Curlew		Dunlin		Golden Plover		Lapwing		Snipe		Red Grouse	
Intercept	**-2.88± 0.03**	**100**	**-1.68± 0.06**	**100**	**-4.32± 0.02**	**100**	**5.24± 0.04**	**100**	**0.25± 0.04**	**100**	**1.41± 0.01**	**100**
MSLOPEKM	**-0.04± 0.00**	**89**	**-0.21± 0.00**	**100**	**-0.09± 0.00**	**100**	**-0.09± 0.00**	**100**	**-0.14± 0.00**	**100**	**0.02± 0.00**	**100**
MELEVKM	**0.00± 0.00**	**100**	0.00± 0.00	4	**0.01± 0.00**	**100**	**-0.01± 0.00**	**100**	0.00± 0.00	20	0.00± 0.00	25
MVGHTKM	**0.09± 0.00**	**100**	0.05± 0.00	46	0.00± 0.00	2	0.00± 0.00	1	0.00± 0.00	2	-0.01± 0.00	38
GRASSKM	**1.90± 0.02**	**100**	-0.26± 0.04	11	0.00± 0.00	1	0.55± 0.04	31	**3.46± 0.03**	**100**	**0.00± 0.00**	**2**
NHEATHKM	0.43± 0.03	28	0.04± 0.02	4	0.03± 0.01	4	-1.04± 0.07	31	0.00± 0.01	2	-0.01± 0.00	3
COTTGRKM	0.40± 0.03	24	**8.06± 0.06**	**100**	**6.98± 0.05**	**100**	-1.31± 0.09	30	-0.01± 0.01	2	-0.01± 0.00	2
PINBYETKM	-0.02± 0.01	4	-0.06± 0.03	4	-0.03± 0.01	4	-0.01± 0.01	1	-0.01± 0.01	2	**-0.87± 0.01**	**99**
PPEATTKM	0.00± 0.00	2	0.01± 0.01	4	0.06± 0.01	14	-0.08± 0.01	16	**-0.93± 0.02**	**84**	0.00± 0.00	2
PWOODTKM	-0.11± 0.01	22	0.00± 0.01	3	0.00± 0.01	4	-0.02± 0.01	3	-0.01± 0.00	3	0.00± 0.00	3
STREAMKM	0.00± 0.00	8	**-0.02± 0.00**	**100**	0.00± 0.00	9	0.00± 0.00	1	0.00± 0.00	29	**0.00± 0.00**	**8**
MPCV11KM	**1.85± 0.02**	**100**	0.130.02	9	-0.02± 0.01	4	-4.10± 0.07	93	0.03± 0.01	3	0.00± 0.00	2
BURNKM	0.00± 0.00	2	-0.010.01	3	-0.10± 0.01	19	1.14± 0.04	65	0.01± 0.01	3	0.00± 0.00	2
DISTKM	**-0.89± 0.03**	**74**	0.130.03	7	-0.01± 0.01	2	-0.45± 0.04	20	-0.01± 0.01	2	**0.01± 0.00**	**5**
MSLOPEKM^2	0.03± 0.00	33			0.02± 0.00	19	0.00± 0.00	1	0.00± 0.00	2	0.00± 0.00	2
MELEVKM^2	0.00± 0.00	5			**-0.27± 0.00**	**100**	**-0.89± 0.01**	**100**	-0.12± 0.01	36	0.00± 0.00	7
MVGHTKM^2	**-0.19± 0.00**	**100**			0.00± 0.00	2	0.00± 0.00	1	0.00± 0.00	4	0.00± 0.00	3
GRASSKM^2	0.00± 0.00	6			0.00± 0.00	1	0.05± 0.00	51	0.00± 0.00	2	**-0.07± 0.00**	**100**
NHEATHKM^2	0.00± 0.00	2			0.00± 0.00	4	0.02± 0.00	12	0.00± 0.00	2	0.00± 0.00	9
COTTGRKM^2	0.00± 0.00	9			**-0.15± 0.00**	**100**	0.12± 0.00	62	0.00± 0.00	2	0.00± 0.00	2
PINBYETKM^2	0.00± 0.00	2			-0.01± 0.00	8	0.00± 0.00	3	0.00± 0.00	2	0.00± 0.00	2
PPEATTKM^2	0.00± 0.00	3			0.00± 0.00	3	0.00± 0.00	1	**-0.42± 0.01**	**88**	0.00± 0.00	3
PWOODTKM^2	0.00± 0.00	3			0.04± 0.00	29	0.00± 0.00	4	0.00± 0.00	2	0.00± 0.00	2
STREAMKM^2	0.00± 0.00	4			-0.01± 0.00	10	0.00± 0.00	1	-0.04± 0.00	36	**-0.02± 0.00**	**94**
MPCV11KM^2	0.00± 0.00	2			0.00± 0.00	3	-0.01± 0.00	6	0.00± 0.00	2	0.00± 0.00	10
BURNKM^2	0.00± 0.00	2			0.00± 0.00	3	0.01± 0.00	10	0.00± 0.00	2	0.00± 0.00	1
DISTKM^2	-0.01± 0.00	22			-0.03± 0.00	25	-0.19± 0.01	82	0.00± 0.00	3	**-0.06± 0.00**	**100**
Random 95%		55		66		49		100		39		52

Table 3.3b. Model outputs for associations at the 1km scale (see Table 3.2 for legend).

	Reed Bunting		Ring Ouzel		Stonechat		Whinchat		Wheatear		Wren		Skylark		Meadow Pipit	
Intercept	**5.76**± **0.06**	**100**	**-3.81**± **0.04**	**100**	**-1.98**± **0.04**	**100**	**4.79**± **0.07**	**100**	**-2.59**± **0.03**	**100**	**-0.05**± **0.02**	**100**	**-0.46**± **0.01**	**100**	**1.55**± **0.00**	**100**
MSLOPEKM	**-0.18**± **0.00**	**100**	**0.16**± **0.00**	**100**	0.01± 0.00	12	0.00± 0.00	3	0.00± 0.00	1	**0.12**± **0.00**	**100**	0.00± 0.00	2	0.00± 0.00	2
MELEVKM	**-0.01**± **0.00**	**100**	0.00± 0.00	1	0.00± 0.00	30	**-0.01**± **0.00**	**100**	0.00± 0.00	4	**0.00**± **0.00**	1	**0.00± 0.00**	**2**	0.00± 0.00	2
MVGHTKM	0.00± 0.00	2	0.00± 0.00	0	0.00± 0.00	1	**-0.02± 0.00**	**11**	0.00± 0.00	2	0.00± 0.00	1	0.00± 0.00	3	0.00± 0.00	2
GRASSKM	**1.21**± **0.04**	**63**	0.02± 0.01	1	0.00± 0.00	1	1.06± 0.06	42	1.80± 0.04	89	0.00± 0.00	**100**	0.00± 0.00	3	0.00± 0.00	2
NHEATHKM	0.01± 0.01	1	1.30± 0.13	21	**10.17**± **0.12**	**100**	0.00± 0.01	1	0.00± 0.01	1	**5.14**± **0.04**	2	0.01± 0.00	2	0.00± 0.00	2
COTTGRKM	0.25± 0.04	9	0.00± 0.01	1	0.02± 0.02	2	-0.01± 0.01	1	-0.01± 0.01	2	0.01± 0.00	36	**0.05± 0.01**	**6**	**0.00**± **0.00**	**2**
PINBYETKM	0.02± 0.01	3	-0.74± 0.06	28	0.15± 0.02	9	-2.33± 0.12	50	-0.01± 0.01	2	-0.37± 0.02	2	**0.14**± **0.01**	**22**	0.00± 0.00	2
PPEATTKM	0.00± 0.00	2	-0.02± 0.01	4	**-0.94**± **0.02**	**85**	-0.28± 0.02	23	0.00± 0.00	1	0.00± 0.00	**21**	0.00± 0.00	2	**0.00**± **0.00**	**2**
PWOODTKM	0.00± 0.00	2	0.00± 0.00	0	-0.01± 0.00	2	-1.52± 0.05	65	-0.03± 0.01	4	**-0.18**± **0.02**	15	0.00± 0.00	2	**0.01**± **0.00**	**6**
STREAMKM	0.00± 0.00	3	0.00± 0.00	1	0.00± 0.00	11	0.00± 0.00	1	0.00± 0.00	88	0.00± 0.00	4	0.00± 0.00	2	0.00± 0.00	2
MPCV11KM	0.00± 0.01	2	0.00± 0.01	1	-0.01± 0.01	1	0.04± 0.01	2	-0.11± 0.02	7	0.03± 0.01	**98**	**-0.59**± **0.02**	**66**	**0.00**± **0.00**	**5**
BURNKM	**-0.87**± **0.04**	**51**	**3.15**± **0.04**	**100**	0.03± 0.01	4	0.03± 0.01	3	0.00± 0.00	2	**0.83**± **0.01**	31	-0.01± 0.00	4	**0.00**± **0.00**	**6**
DISTKM	0.07± 0.01	8	**5.37**± **0.10**	**98**	0.00± 0.00	1	**2.59**± **0.04**	**93**	0.32± 0.03	20	**0.23± 0.02**	**95**	0.00± 0.00	2	0.00± 0.00	2
MSLOPEKM^2	0.00± 0.00	1	-0.14± 0.01	51	0.00± 0.00	2	-0.08± 0.01	33	0.00± 0.00	3	-0.09± 0.00	17	0.00± 0.00	3	0.00± 0.00	3
MELEVKM^2	**-0.54**± **0.01**	**98**	-0.01± 0.00	5	0.00± 0.00	3	0.00± 0.00	1	-0.08± 0.01	28	-0.01± 0.00	12	**-0.02**± **0.00**	**21**	0.00± 0.00	2
MVGHTKM^2	0.00± 0.00	1	-0.19± 0.01	42	0.00± 0.00	1	**-0.51**± **0.01**	**90**	0.00± 0.00	1	-0.01± 0.00	2	0.00± 0.00	2	0.00± 0.00	2
GRASSKM^2	0.00± 0.00	3	0.00± 0.00	1	-0.01± 0.00	6	0.10± 0.00	59	-0.01± 0.00	9	0.00± 0.00	**100**	0.00± 0.00	2	0.00± 0.00	2
NHEATHKM^2	0.00± 0.00	2	-0.05± 0.01	11	**-0.48**± **0.01**	**91**	0.00± 0.00	1	0.00± 0.00	2	**-0.20**± **0.00**	2	0.00± 0.00	2	0.00± 0.00	2
COTTGRKM^2	0.00± 0.00	3	-0.41± 0.02	68	-0.35± 0.01	67	-0.01± 0.00	3	0.00± 0.00	2	0.00± 0.00	47	0.00± 0.00	4	**0.00**± **0.00**	**3**
PINBYETKM^2	0.00± 0.00	2	0.00± 0.00	2	0.00± 0.00	1	0.19± 0.01	40	0.00± 0.00	1	-0.06± 0.00	3	**0.00± 0.00**	**6**	0.00± 0.00	2
PPEATTKM^2	-0.09± 0.01	32	0.00± 0.00	2	-0.05± 0.00	19	-0.10± 0.01	23	-0.04± 0.00	17	0.00± 0.00	**81**	0.00± 0.00	2	**0.00**± **0.00**	**10**
PWOODTKM^2	-0.01± 0.00	4	0.01± 0.00	4	0.00± 0.00	1	-0.12± 0.01	31	-0.05± 0.00	19	**-0.14**± **0.00**	2	0.00± 0.00	2	**0.00± 0.00**	**3**
STREAMKM^2	0.00± 0.00	12	0.00± 0.00	0	-0.02± 0.00	20	0.00± 0.00	2	-0.01± 0.00	11	0.00± 0.00	2	0.00± 0.00	2	0.00± 0.00	2
MPCV11KM^2	0.00± 0.00	2	-0.18± 0.01	63	-0.01± 0.00	6	-0.29± 0.01	64	0.00± 0.00	3	0.00± 0.00	2	**-0.02**± **0.00**	**20**	0.00± 0.00	2
BURNKM^2	-0.03± 0.00	21	-0.03± 0.00	16	0.00± 0.00	1	0.00± 0.00	1	0.00± 0.00	2	0.00± 0.00	2	0.00± 0.00	2	0.00± 0.00	3
DISTKM^2	0.00± 0.00	4	**-0.17**± **0.01**	**76**	0.00± 0.00	1	0.01± 0.00	8	0.00± 0.00	3	0.00± 0.00	**100**	0.00± 0.00	2	0.00± 0.00	2
Random 95%		34		73		76		82		93		72		6		3

Table 3.4a. Model outputs for change at the 1km scale (see Table 3.2 for legend).

	Curlew		Dunlin		Golden Plover		Lapwing		Snipe		Red Grouse	
Intercept	**-0.8037 ± 1.391**	**100**	**-1.10 ± 4.42**	**100**	**-0.248 ± 0.439**	**100**	**-3.17 ± 3.31**	**100**	**0.92 ± 1.87**	**100**	**0.95 ± 0.52**	**100**
MSLOPEKM	0.00 ± 0.01	6	**-0.28 ± 0.09**	**100**	0.00 ± 0.00	2	0.02 ± 0.05	21	0.02 ± 0.05	21	0.04 ± 0.02	93
MELEVKM	0.00 ± 0.00	10	0.00 ± 0.00	9	0.00 ± 0.00	14	0.01 ± 0.00	96	0.00 ± 0.00	8	0.00 ± 0.00	1
MVGHTKM	**0.03 ± 0.04**	**46**	**0.14 ± 0.18**	**48**	0.00 ± 0.01	2	0.10 ± 0.10	58	0.01 ± 0.05	8	-0.05 ± 0.02	95
GRASSKM	0.02 ± 0.15	3	**-0.60 ± 1.91**	**16**	0.00 ± 0.08	2	0.48 ± 1.38	14	0.02 ± 0.23	2	-1.12 ± 0.92	63
NHEATHKM	-0.10 ± 0.51	6	**-0.38 ± 1.44**	**13**	-0.02 ± 0.18	3	-13.68 ± 5.33	93	**-9.16 ± 4.58**	**89**	-1.87 ± 0.94	86
COTTGRKM	-0.22 ± 0.73	11	0.07 ± 0.76	6	**2.01 ± 0.82**	**93**	-2.23 ± 6.27	13	-0.03 ± 0.60	2	2.44 ± 0.94	95
PINBYETKM	0.05 ± 0.26	5	**13.02 ± 6.29**	**92**	0.03 ± 0.28	3	-0.04 ± 0.33	3	0.48 ± 1.36	14	0.00 ± 0.04	1
PPEATTKM	-0.10 ± 0.18	25	-0.04 ± 0.51	7	0.00 ± 0.03	1	-0.01 ± 0.10	2	0.02 ± 0.16	4	0.00 ± 0.01	1
PWOODTKM	0.01 ± 0.09	3	0.02 ± 0.42	5	0.07 ± 0.23	11	0.06 ± 0.40	4	0.00 ± 0.12	1	0.54 ± 0.70	48
STREAMKM	0.00 ± 0.00	2	0.00 ± 0.00	11	0.00 ± 0.00	8	0.00 ± 0.00	2	0.00 ± 0.00	2	0.00 ± 0.00	0
MPCV11KM	**-1.34 ± 1.10**	**68**	0.08 ± 0.64	9	-0.05 ± 0.23	6	-0.18 ± 1.07	7	-0.01 ± 0.20	2	0.41 ± 0.67	30
BURNKM	0.00 ± 0.09	2	**0.71 ± 1.44**	**27**	0.00 ± 0.03	1	-0.01 ± 0.15	1	0.00 ± 0.11	1	-0.08 ± 0.19	17
DISTKM	**0.18 ± 0.70**	**8**	-0.25 ± 1.26	8	0.00 ± 0.07	1	0.10 ± 0.70	4	0.10 ± 0.64	5	0.00 ± 0.06	1
MSLOPEKM^2	0.00 ± 0.01	1			0.00 ± 0.01	3	0.00 ± 0.01	1	0.00 ± 0.02	2	0.00 ± 0.00	1
MELEVKM^2	0.00 ± 0.01	1			0.00 ± 0.01	3	-0.25 ± 0.24	61	0.00 ± 0.04	3	**-0.11 ± 0.03**	**100**
MVGHTKM^2	0.00 ± 0.01	2			-0.01 ± 0.02	11	0.00 ± 0.02	1	0.08 ± 0.17	23	0.01 ± 0.03	19
GRASSKM^2	0.00 ± 0.01	1			0.00 ± 0.01	2	0.03 ± 0.09	17	0.21 ± 0.32	38	0.08 ± 0.06	68
NHEATHKM^2	0.00 ± 0.01	2			0.00 ± 0.00	1	0.30 ± 0.17	84	0.00 ± 0.03	2	0.00 ± 0.00	1
COTTGRKM^2	0.00 ± 0.01	2			0.00 ± 0.01	3	0.07 ± 0.18	20	0.01 ± 0.04	6	0.00 ± 0.02	6
PINBYETKM^2	0.00 ± 0.02	2			0.00 ± 0.01	1	0.01 ± 0.07	1	0.01 ± 0.07	3	0.00 ± 0.00	1
PPEATTKM^2	0.00 ± 0.01	1			0.00 ± 0.01	2	-0.09 ± 0.20	20	0.00 ± 0.02	1	0.02 ± 0.04	20
PWOODTKM^2	0.00 ± 0.02	3			0.01 ± 0.02	9	0.00 ± 0.03	1	0.00 ± 0.02	1	-0.05 ± 0.09	31
STREAMKM^2	0.00 ± 0.01	2			0.00 ± 0.01	4	-0.01 ± 0.04	11	0.01 ± 0.05	6	0.00 ± 0.00	2
MPCV11KM^2	**0.32 ± 0.10**	**100**			0.00 ± 0.01	2	-0.38 ± 0.19	90	0.00 ± 0.02	1	0.01 ± 0.03	14
BURNKM^2	0.00 ± 0.01	1			0.00 ± 0.00	1	0.00 ± 0.01	1	0.00 ± 0.01	1	-0.03 ± 0.02	72
DISTKM^2	**0.20 ± 0.09**	**93**			0.00 ± 0.01	2	-0.21 ± 0.16	84	0.00 ± 0.02	2	0.00 ± 0.00	1
Random 95%		27		12		32		100		48		100

Table 3.4b. Model outputs for change at the 1km scale (see Table 3.2 for legend).

	Reed Bunting			Ring Ouzel			Whinchat			Wheatear			Skylark			Meadow Pipit		
Intercept	**-0.02**	± **1.05**	**100**	**2.25**	± **1.58**	**100**	**4.25**	± **3.33**	**100**	**1.91**	± **0.86**	**100**	**-0.05**	± **0.27**	**100**	**0.89**	**± 0.43**	**100**
MSLOPEKM	0.00	± 0.02	4	-0.01	± 0.03	14	0.00	± 0.01	1	**-0.15**	± **0.04**	**99**	0.00	± 0.00	1	0.02	± 0.01	90
MELEVKM	0.00	± 0.00	3	**-0.01**	± **0.00**	**95**	**0.00**	± **0.00**	6	0.00	± 0.00	2	0.00	± 0.00	1	0.00	± 0.00	92
MVGHTKM	0.00	± 0.02	5	0.00	± 0.01	2	0.00	± 0.01	0	0.00	± 0.01	2	0.00	± 0.00	2	-0.01	± 0.01	63
GRASSKM	-0.04	± 0.34	4	0.00	± 0.24	2	0.55	± 1.44	15	0.12	± 0.56	7	0.11	± 0.31	14	0.00	± 0.03	1
NHEATHKM	**-0.29**	± **1.60**	**7**	0.10	± 0.82	3	**-12.71**	± **11.70**	**65**	-0.12	± 0.78	4	-0.08	± 0.37	6	-0.10	± 0.34	11
COTTGRKM	**-0.31**	± **1.48**	**8**	0.00	± 0.44	2	-0.03	± 0.46	1	-0.07	± 0.64	3	-0.06	± 0.35	5	0.00	± 0.05	1
PINBYETKM	0.43	± 1.41	14	-0.01	± 0.21	2	-1.58	± 2.59	31	0.01	± 0.21	2	0.02	± 0.14	3	-0.02	± 0.09	5
PPEATTKM	0.01	± 0.14	4	0.01	± 0.09	2	0.01	± 0.09	1	0.03	± 0.15	5	0.00	± 0.03	2	0.00	± 0.01	1
PWOODTKM	-0.05	± 0.38	5	0.02	± 0.20	2	-1.91	± 1.95	55	-0.04	± 0.26	5	**1.10**	± **0.33**	**97**	0.60	± 0.26	92
STREAMKM	0.00	± 0.00	3	0.00	± 0.00	2	0.00	± 0.00	3	0.00	± 0.00	2	0.00	± 0.00	1	0.00	± 0.00	4
MPCV11KM	0.12	± 0.67	6	-0.01	± 0.45	4	**-0.16**	± **0.84**	**4**	**-3.05**	± **1.87**	**83**	-0.13	± 0.56	6	**-1.09**	**± 0.26**	**100**
BURNKM	0.06	± 0.46	5	**1.53**	± **1.13**	**74**	-0.02	± 0.27	2	0.15	± 0.58	10	**-1.43**	± **0.46**	**97**	0.00	± 0.04	3
DISTKM	0.09	± 0.57	6	-0.06	± 0.41	4	0.00	± 0.06	0	-0.05	± 0.31	4	0.00	± 0.05	1	-0.23	± 0.26	50
MSLOPEKM^2				0.00	± 0.02	2	0.08	± 0.19	18	0.00	± 0.02	2	0.00	± 0.01	2	0.00	± 0.00	3
MELEVKM^2				0.00	± 0.03	2	**0.33**	± **0.28**	**68**	-0.05	± 0.11	21	-0.09	± 0.07	70	-0.02	± 0.03	43
MVGHTKM^2				-0.01	± 0.05	3	0.00	± 0.03	1	0.00	± 0.01	2	0.00	± 0.00	1	0.00	± 0.00	2
GRASSKM^2				0.00	± 0.03	2	0.13	± 0.21	35	0.00	± 0.02	2	0.01	± 0.03	21	0.01	± 0.02	22
NHEATHKM^2				0.00	± 0.02	1	0.00	± 0.02	0	0.00	± 0.01	2	0.00	± 0.01	1	0.00	± 0.01	4
COTTGRKM^2				-0.02	± 0.07	7	0.00	± 0.01	0	0.00	± 0.01	2	0.00	± 0.01	2	0.00	± 0.00	1
PINBYETKM^2				0.00	± 0.04	3	0.00	± 0.06	1	0.01	± 0.04	4	0.02	± 0.06	17	0.00	± 0.01	3
PPEATTKM^2				0.05	± 0.13	18	0.00	± 0.02	0	0.00	± 0.03	3	0.00	± 0.02	3	0.00	± 0.01	4
PWOODTKM^2				0.01	± 0.03	4	0.00	± 0.03	1	-0.01	± 0.04	5	0.01	± 0.04	4	-0.13	± 0.05	94
STREAMKM^2				0.00	± 0.01	1	0.03	± 0.09	14	0.00	± 0.01	2	0.00	± 0.01	7	0.00	± 0.00	5
MPCV11KM^2				-0.03	± 0.10	13	**-0.81**	± **0.69**	**71**	-0.02	± 0.08	10	0.04	± 0.06	37	0.04	± 0.03	73
BURNKM^2				0.01	± 0.05	6	0.19	± 0.28	36	0.00	± 0.02	2	0.00	± 0.02	5	-0.01	± 0.01	30
DISTKM^2				-0.01	± 0.05	9	0.00	± 0.01	0	0.00	± 0.02	4	-0.08	± 0.02	**100**	0.00	± 0.00	5
Random 95%			7			27			62			59			77			100

Table 3.5. Summary of model performance for each of the three analyses presented in Tables 3.1-3.3. Models in bold outline significant positive associations between predicted and observed abundance / change across the test data, and therefore indicate those models with predictive power.

Species	Fine-scale association		1 km association		1 km change	
	AUC	*P*	Kendall's Tau	*P*	Kendall's Tau	*P*
Curlew	**0.659**	**0.003**	**4.135**	**0**	**2.2368**	**0.0253**
Golden Plover	**0.845**	**0**	**5.225**	**0**	-0.6375	0.5238
Lapwing	**0.723**	**0.003**	1.597	0.110	-1.7110	0.0871
Reed Bunting	**0.899**	**0**	**2.752**	**0.006**		
Ring Ouzel	**0.861**	**0**	-0.592	0.554	-0.8207	0.4118
Stonechat	**0.690**	**0.011**	-0.506	0.613		
Snipe	**0.867**	**0**	**3.492**	**0.001**	1.4118	0.158
Whinchat	**0.778**	**0**	0.990	0.322	0.6547	0.5127
Wheatear	**0.762**	**0.001**	-1.086	0.278	-0.1926	0.8473
Wren	**0.787**	**0**	1.759	0.079		
Dunlin	**0.944**	**0**	0.345	0.73		
Red Grouse			**2.936**	**0.003**	-3.0497	0.0023
Skylark			**2.315**	**0.021**	1.5268	0.1268
Meadow Pipit			-1.710	0.087	0.6518	0.5145

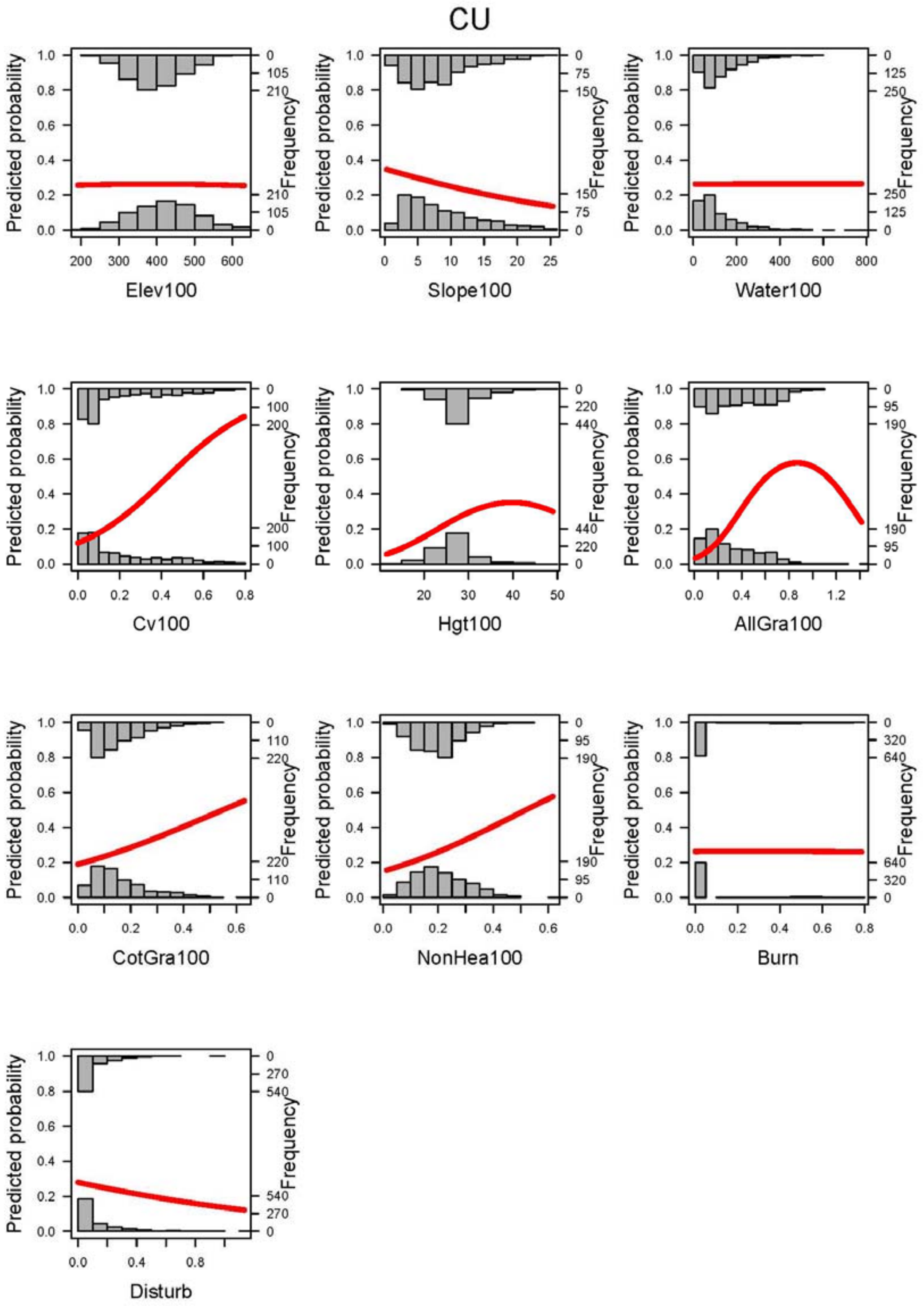


Figure 3.1a Results from the fine-scale analysis for Curlew. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

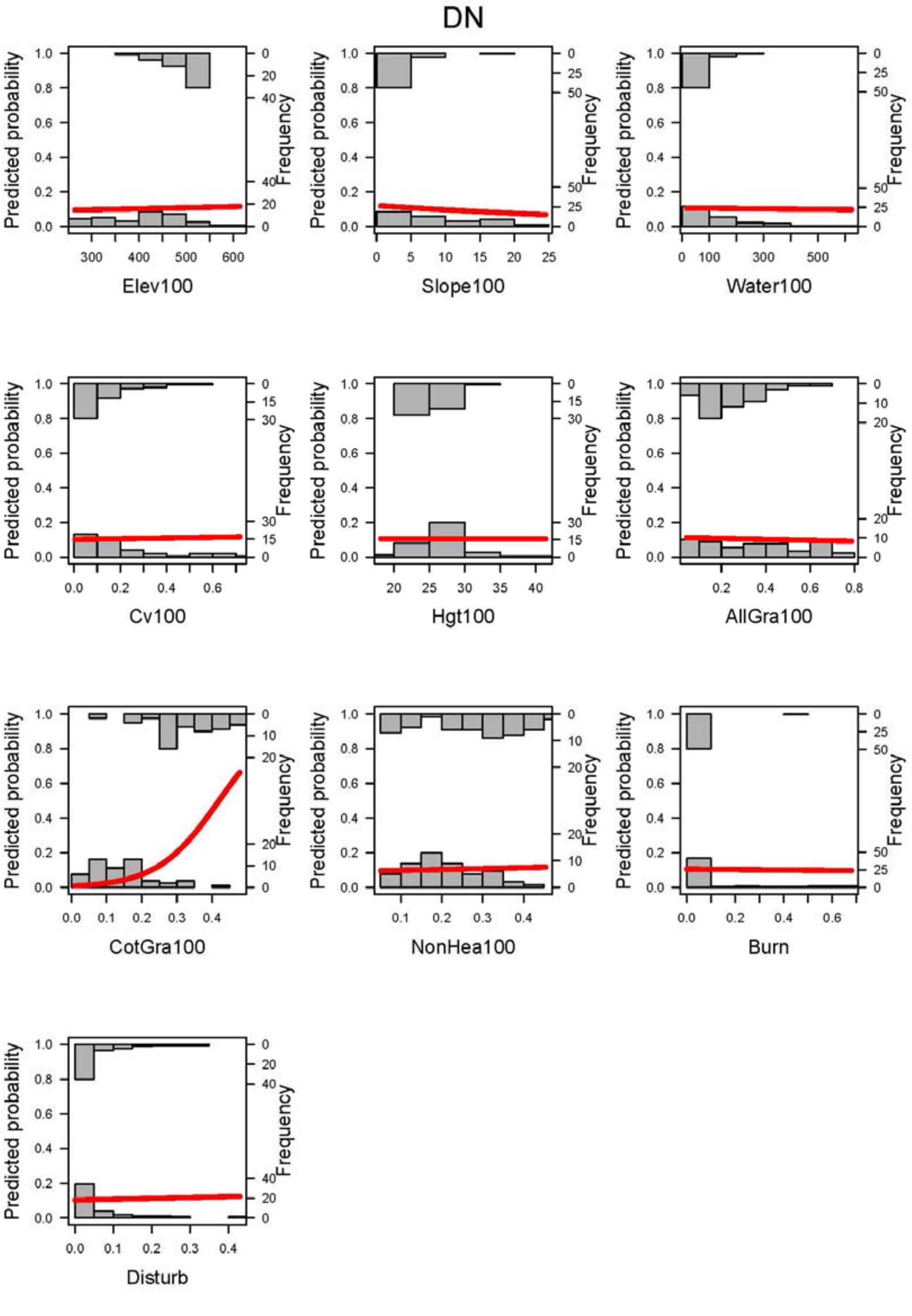


Figure 3.1b Results from the fine-scale analysis for Dunlin. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

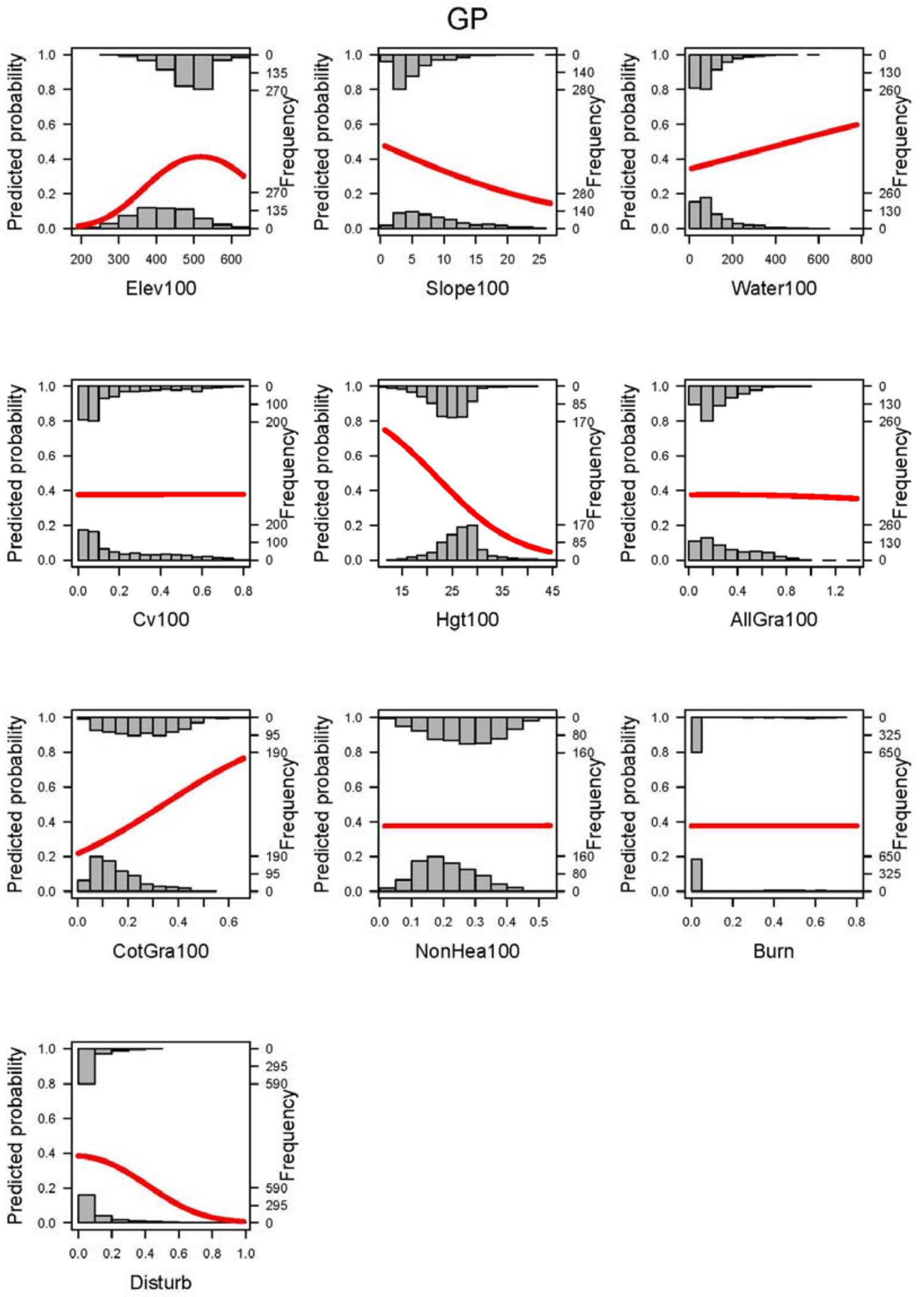


Figure 3.1c Results from the fine-scale analysis for Golden Plover. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

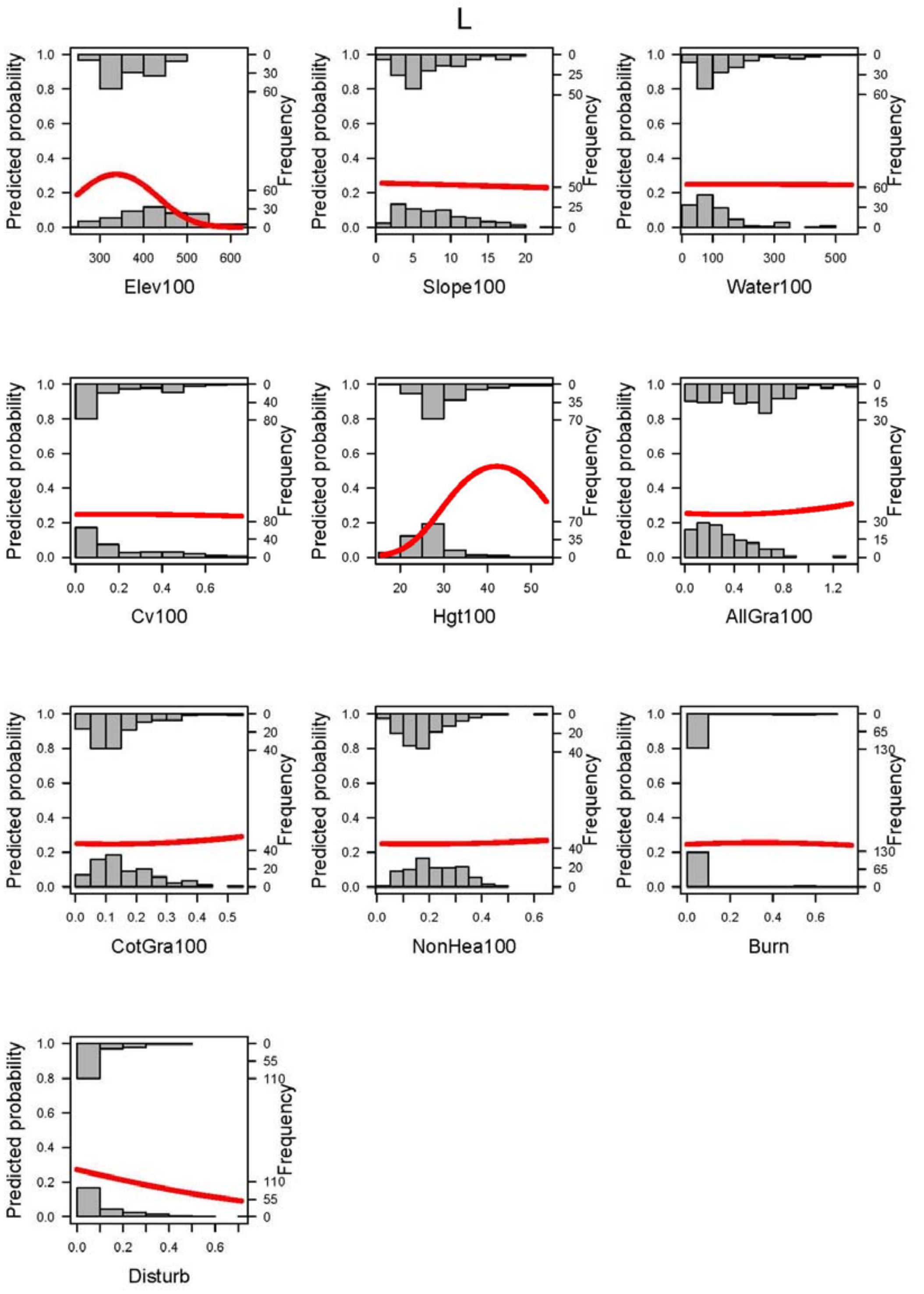


Figure 3.1d: Results from the fine-scale analysis for Lapwing. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

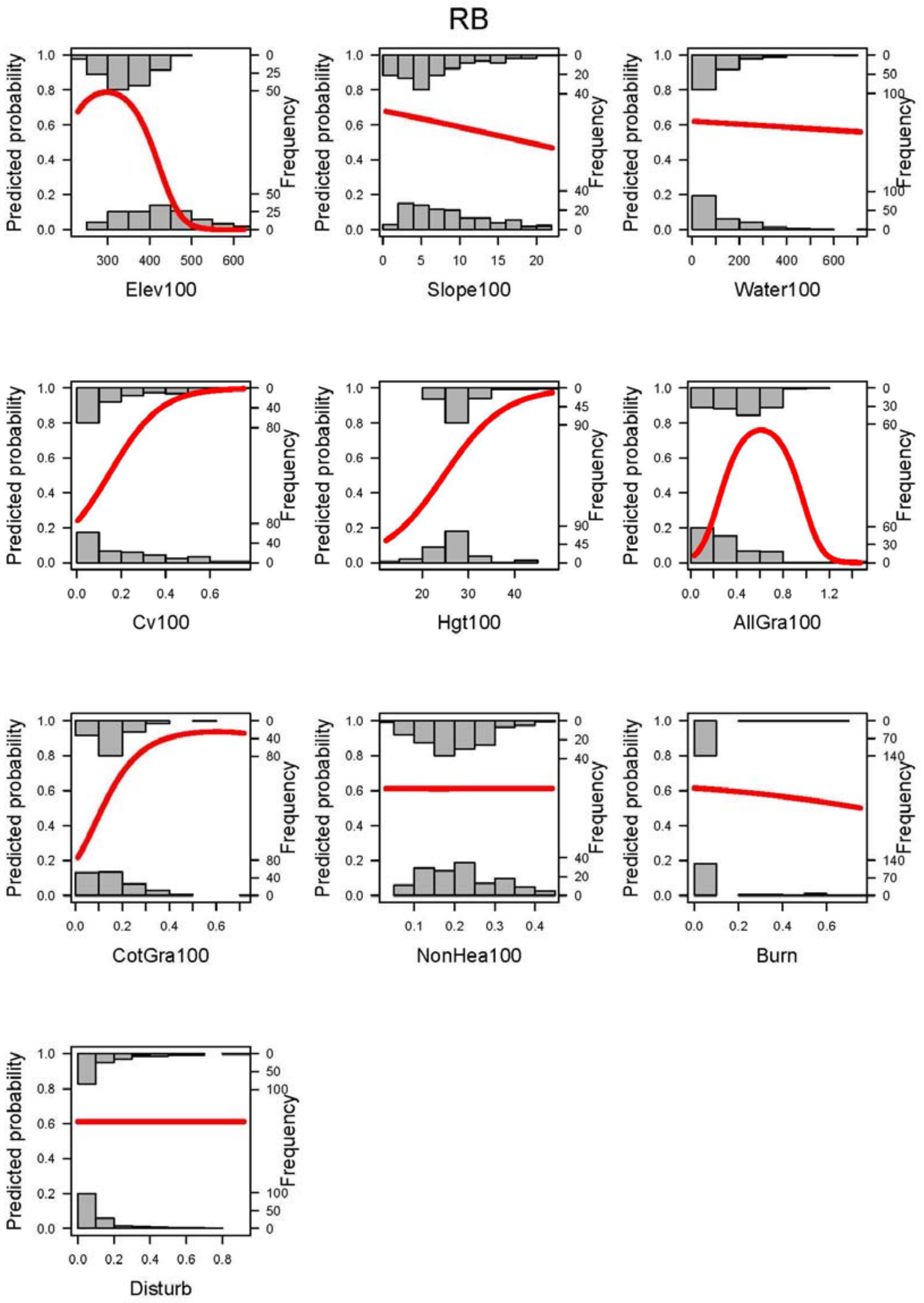


Figure 3.1e: Results from the fine-scale analysis for Reed Bunting. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

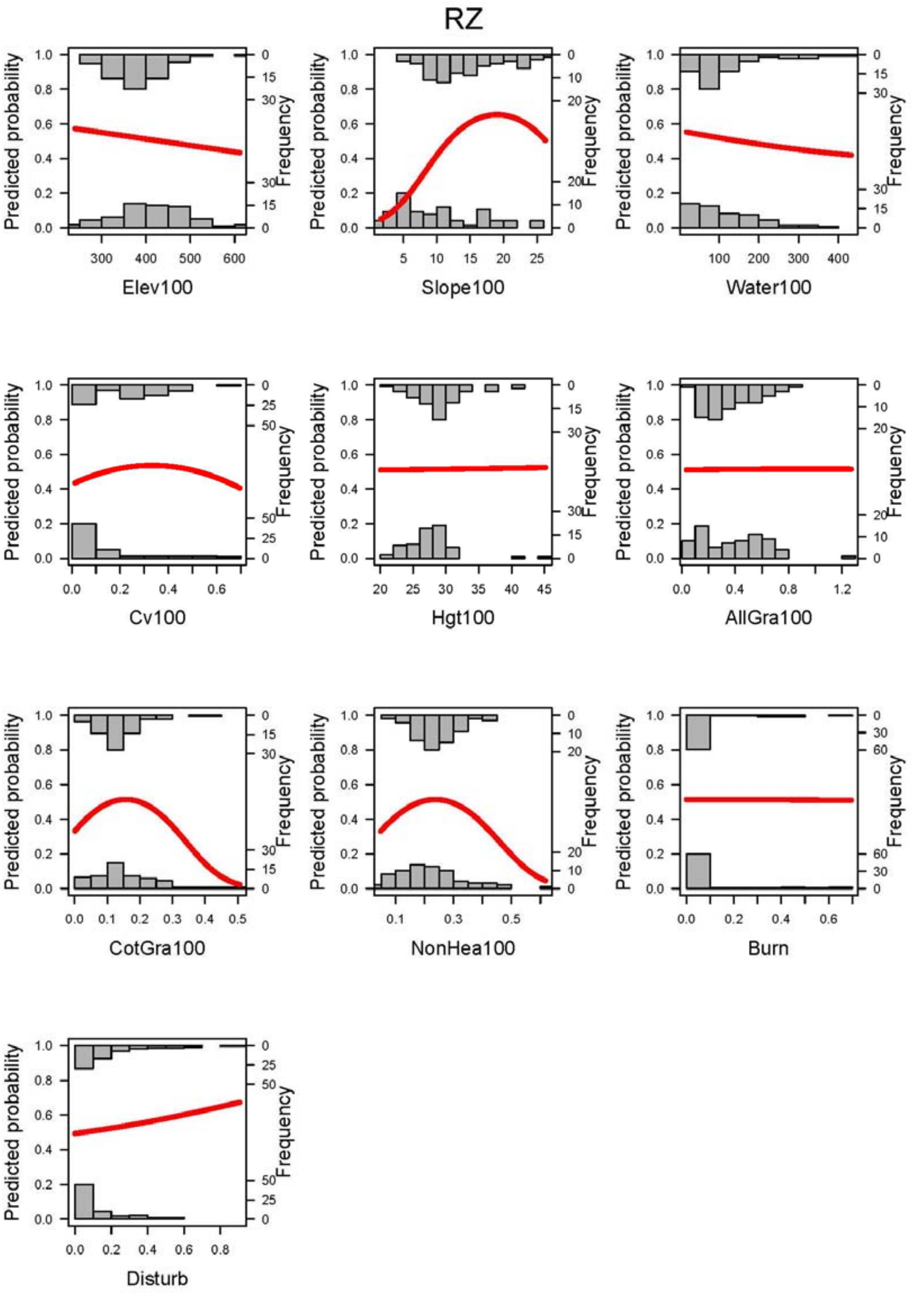


Figure 3.1f: Results from the fine-scale analysis for Ring Ouzel. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

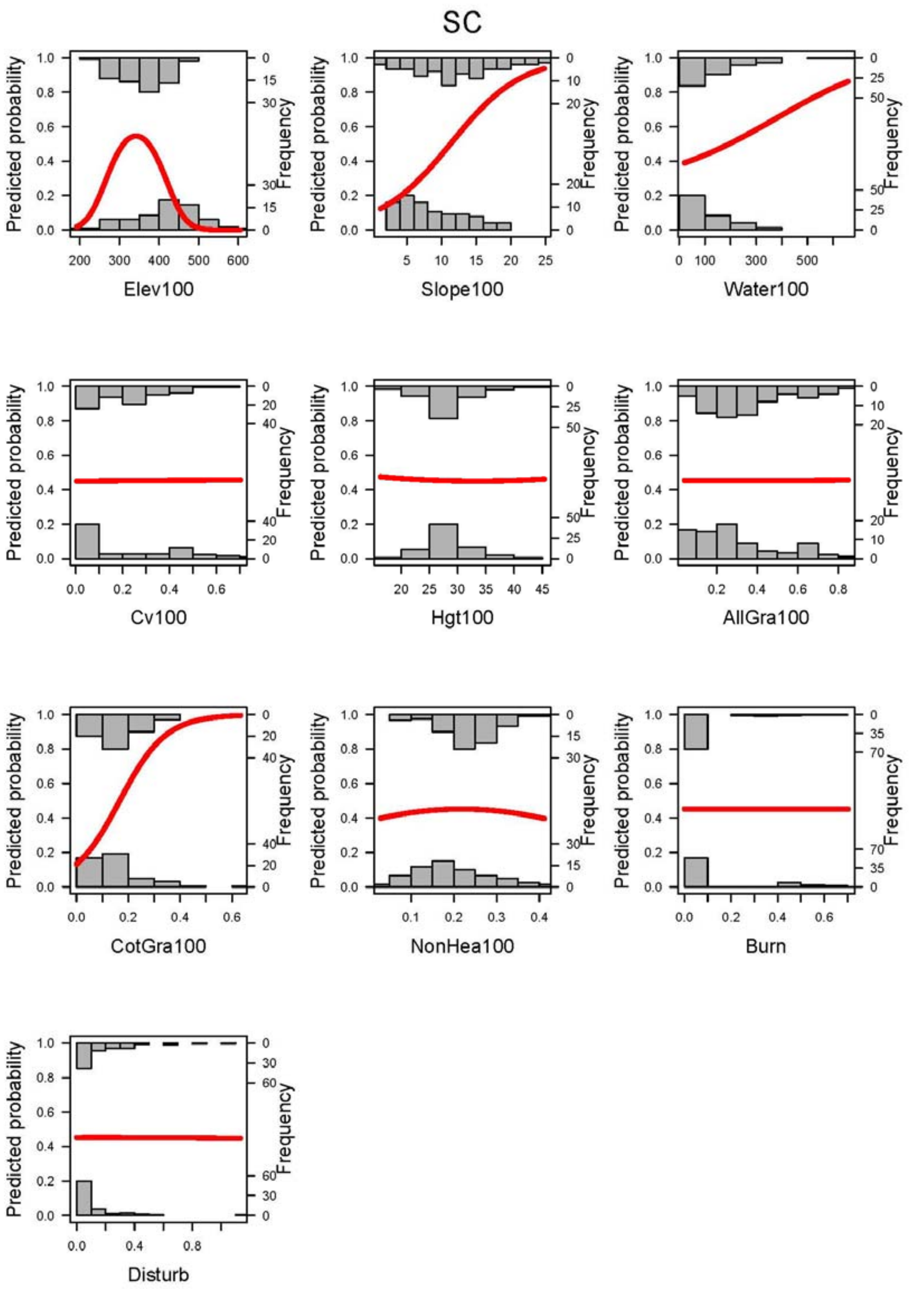


Figure 3.1g: Results from the fine-scale analysis for Stonechat. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

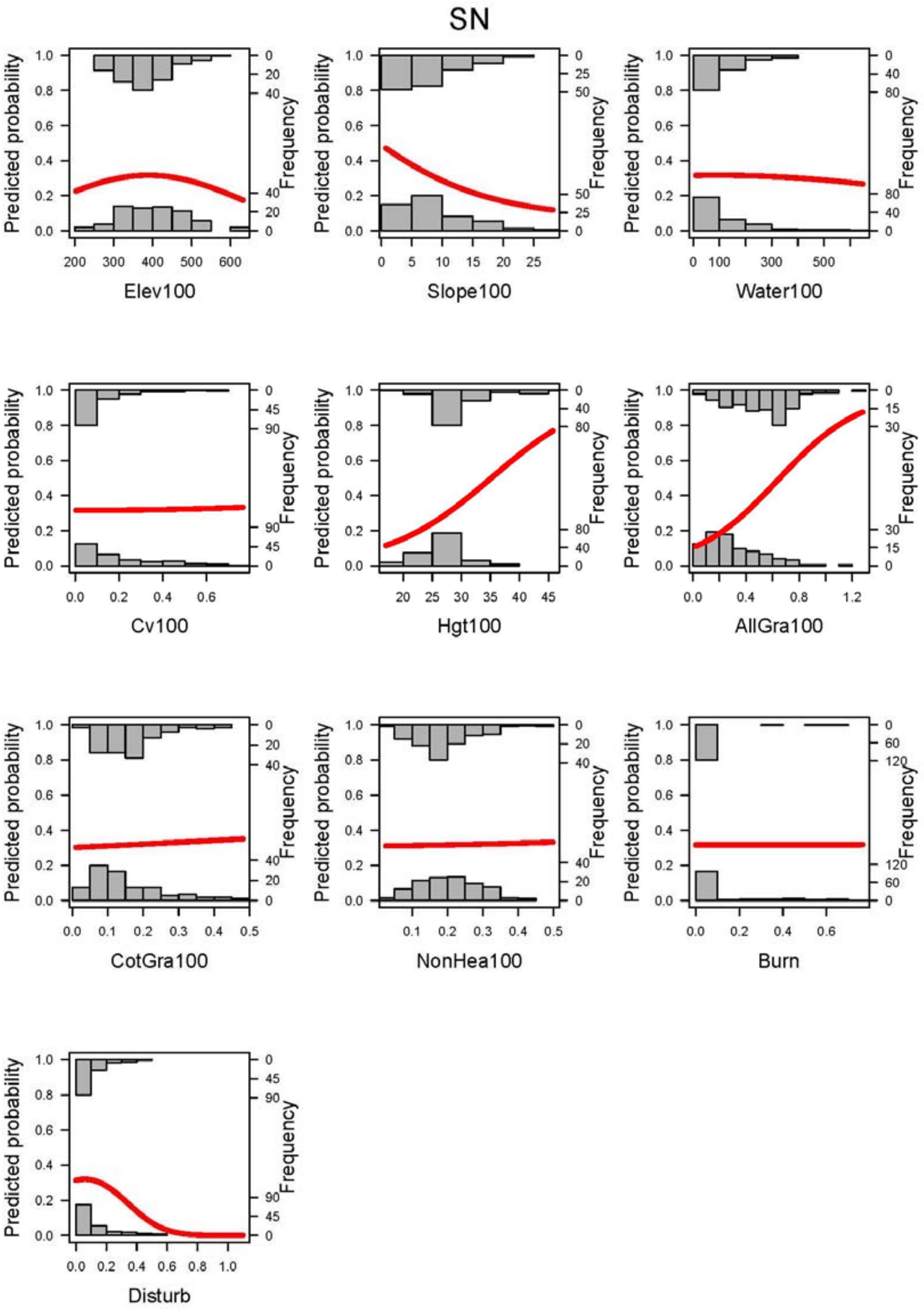


Figure 3.1h: Results from the fine-scale analysis for Snipe. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

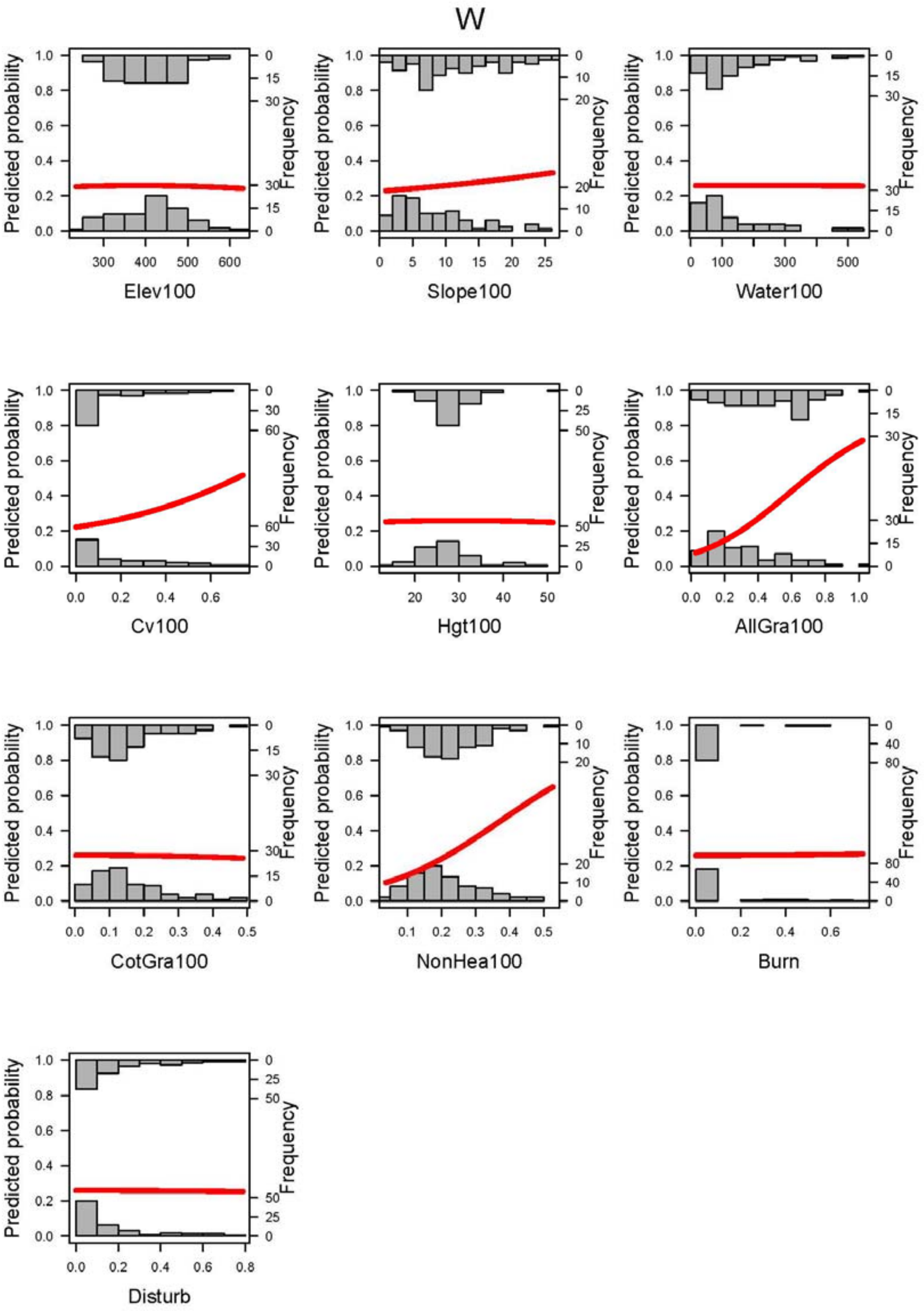


Figure 3.1i: Results from the fine-scale analysis for Wheatear. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

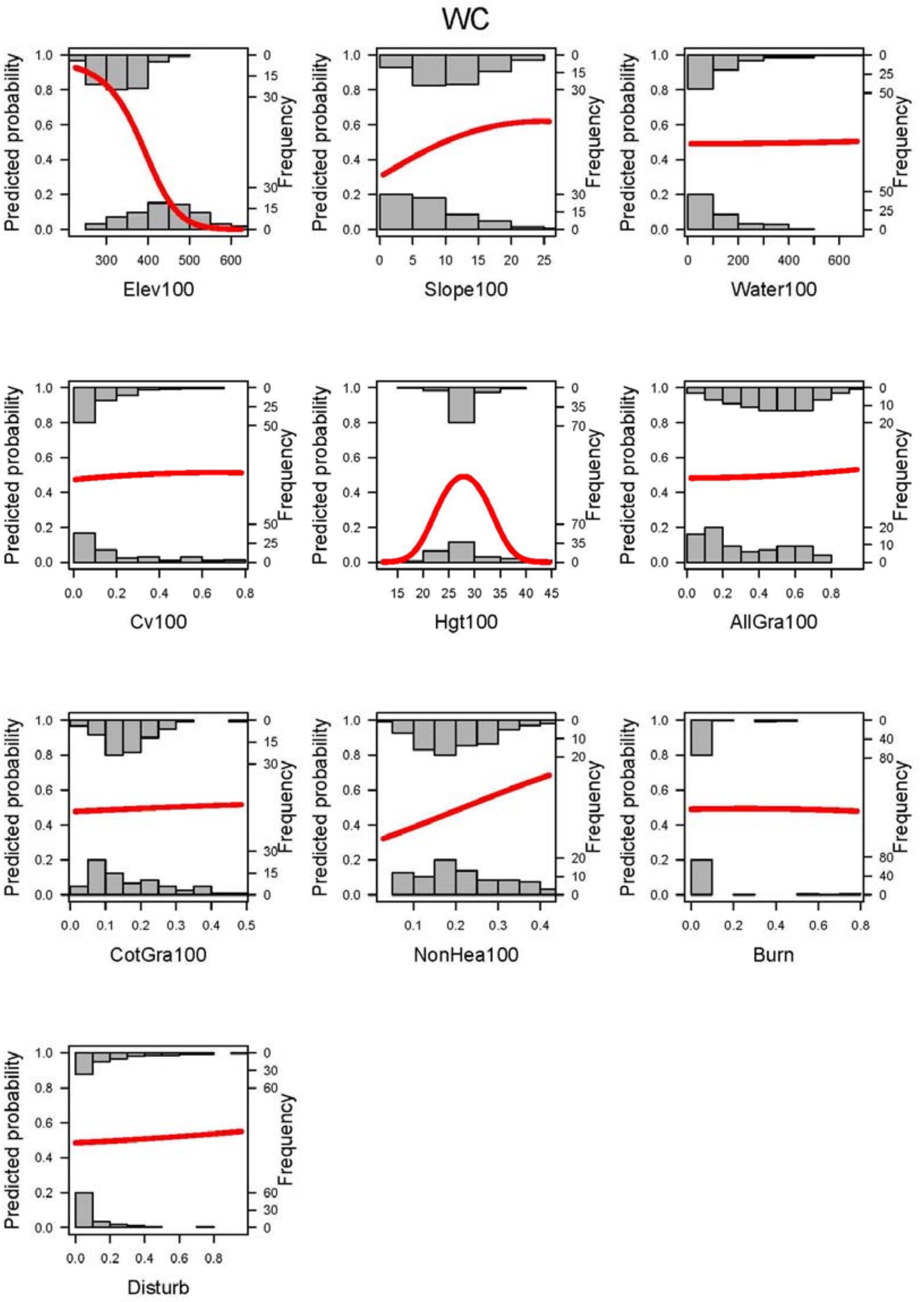


Figure 3.1j: Results from the fine-scale analysis for Whinchat. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

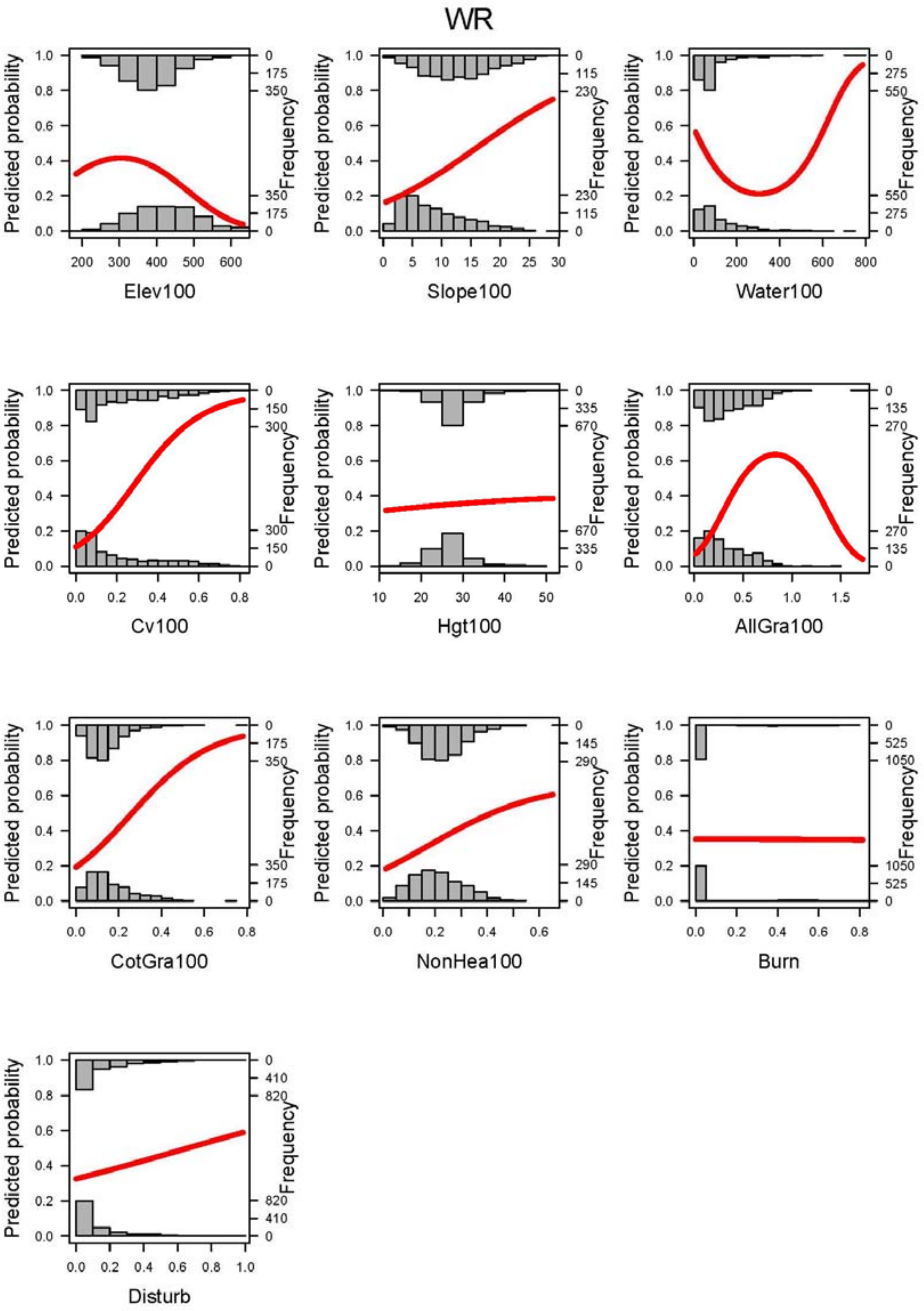


Figure 3.1k: Results from the fine-scale analysis for Wren. Graphs indicate the form of relationships between the probability of bird occurrence and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for the random points (bottom) and bird locations (top).

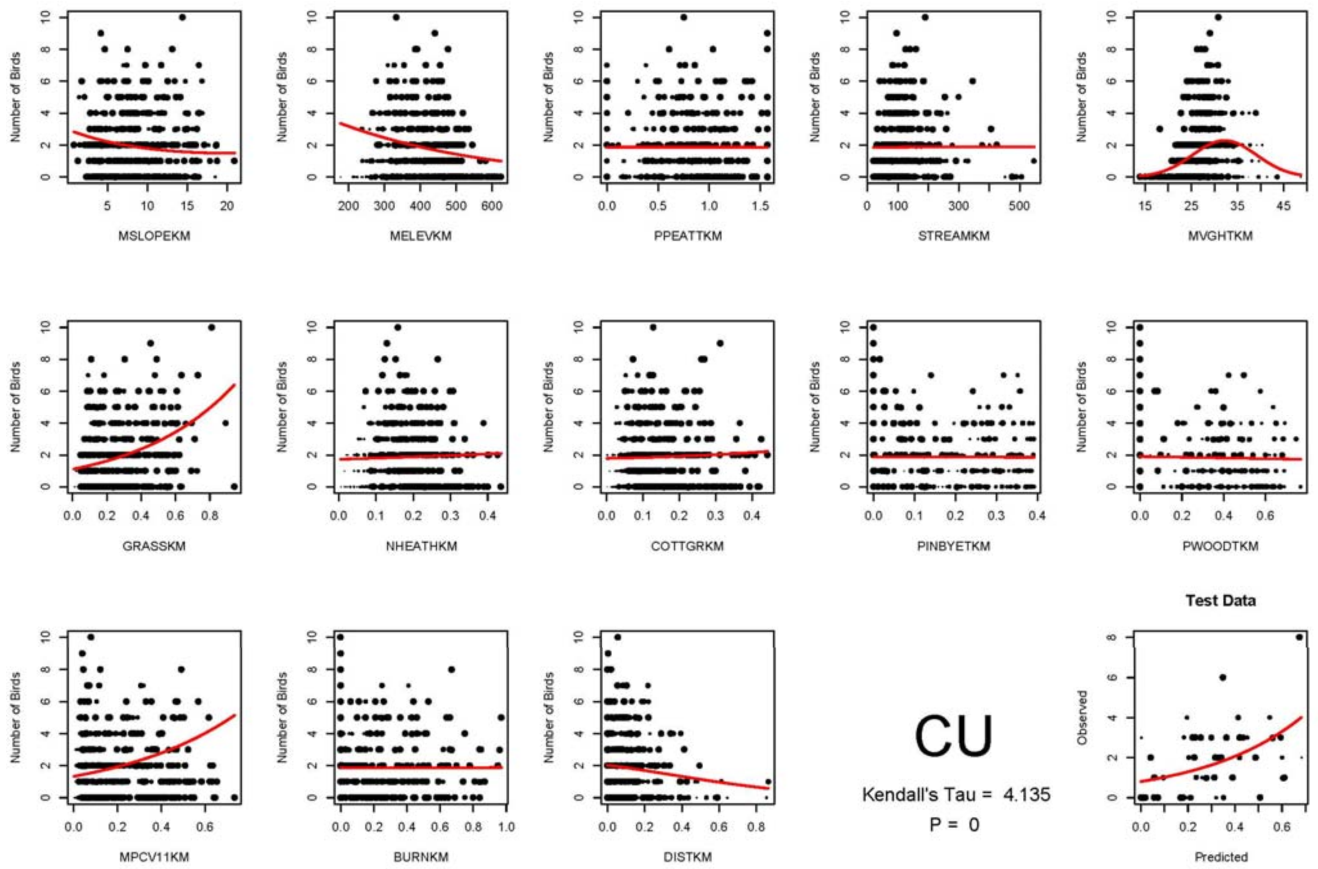


Figure 3.2a: Results from the 1 km-scale analysis of habitat associations for Curlew. Graphs ndicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

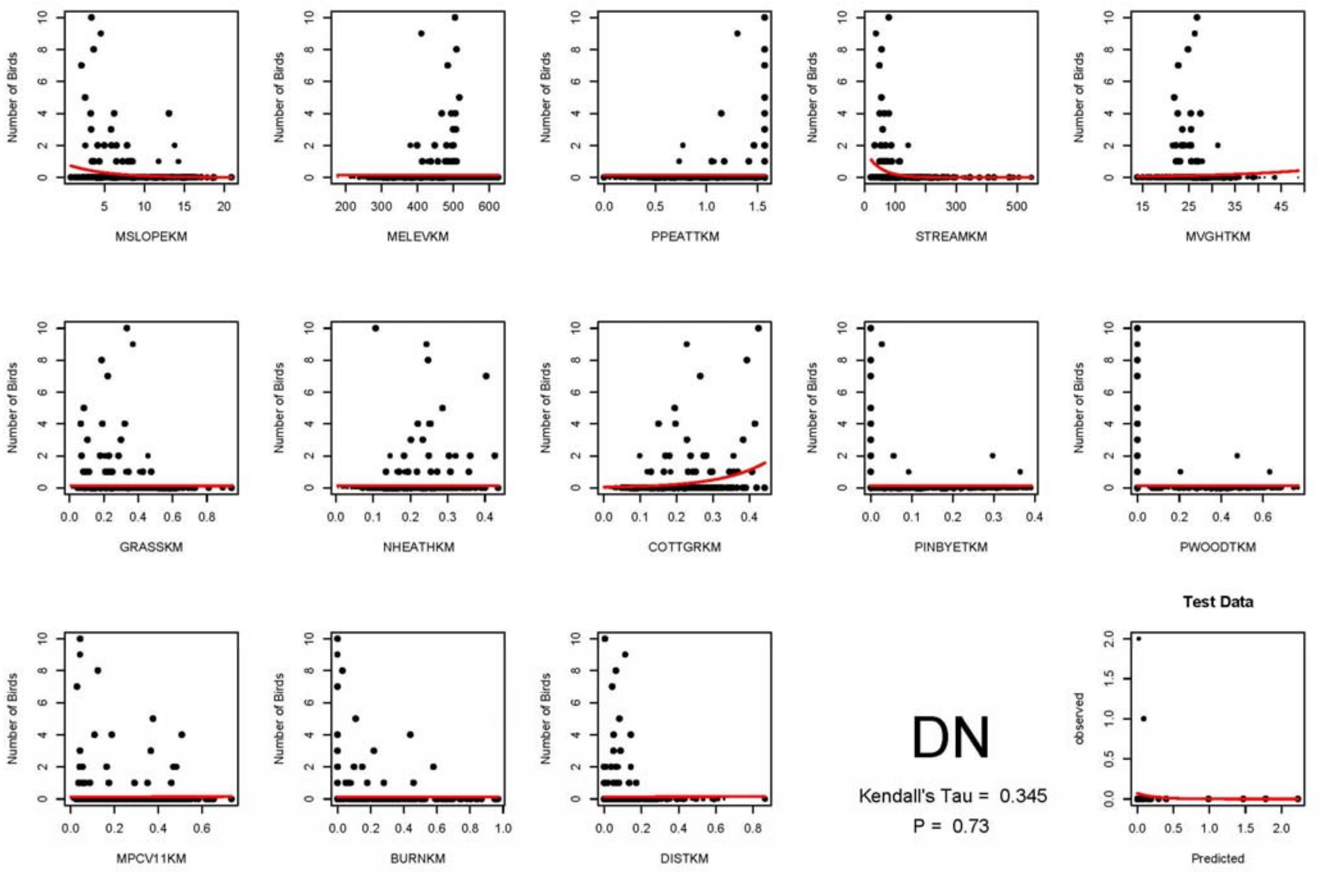


Figure 3.2b: Results from the 1 km-scale analysis of habitat associations for Dunlin. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

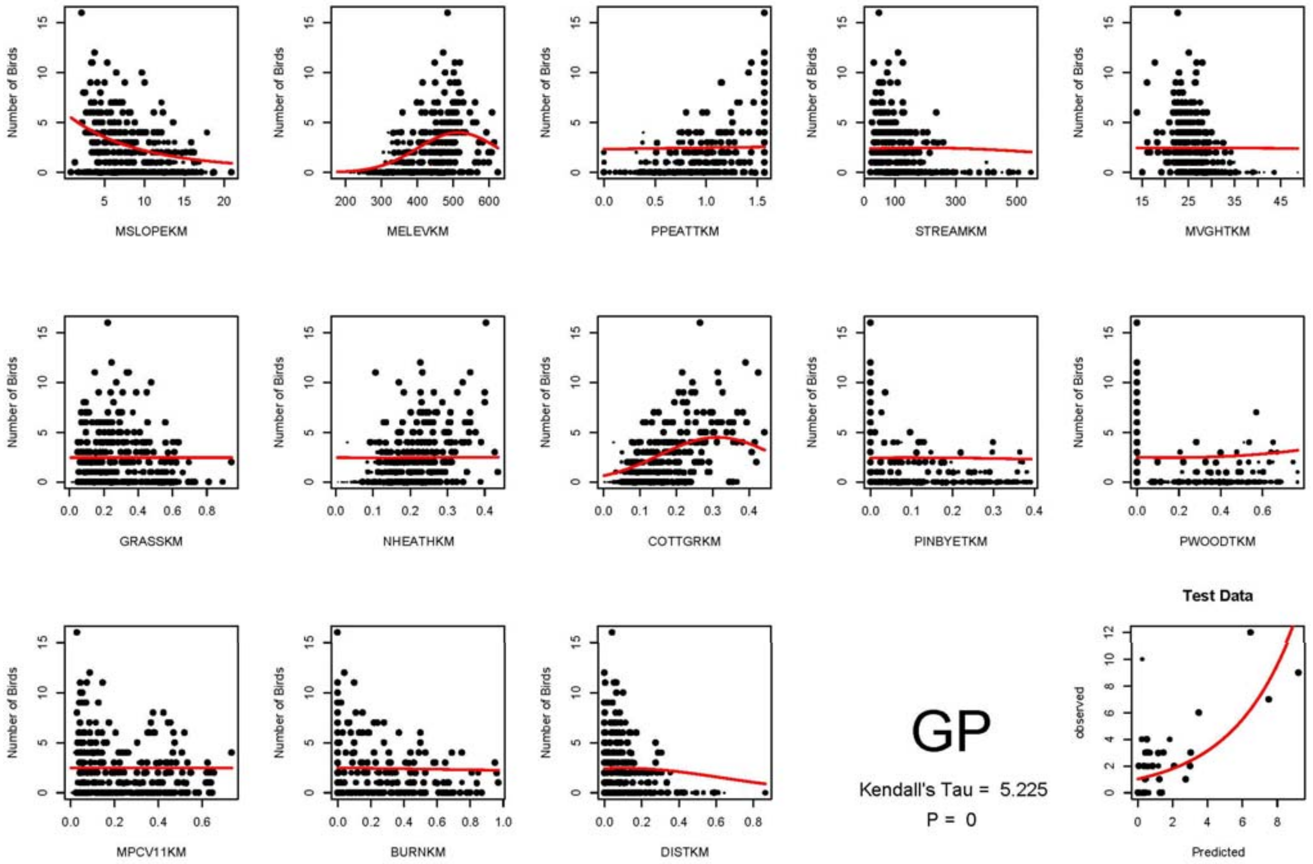


Figure 3.2c: Results from the 1 km-scale analysis of habitat associations for Golden Plover. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

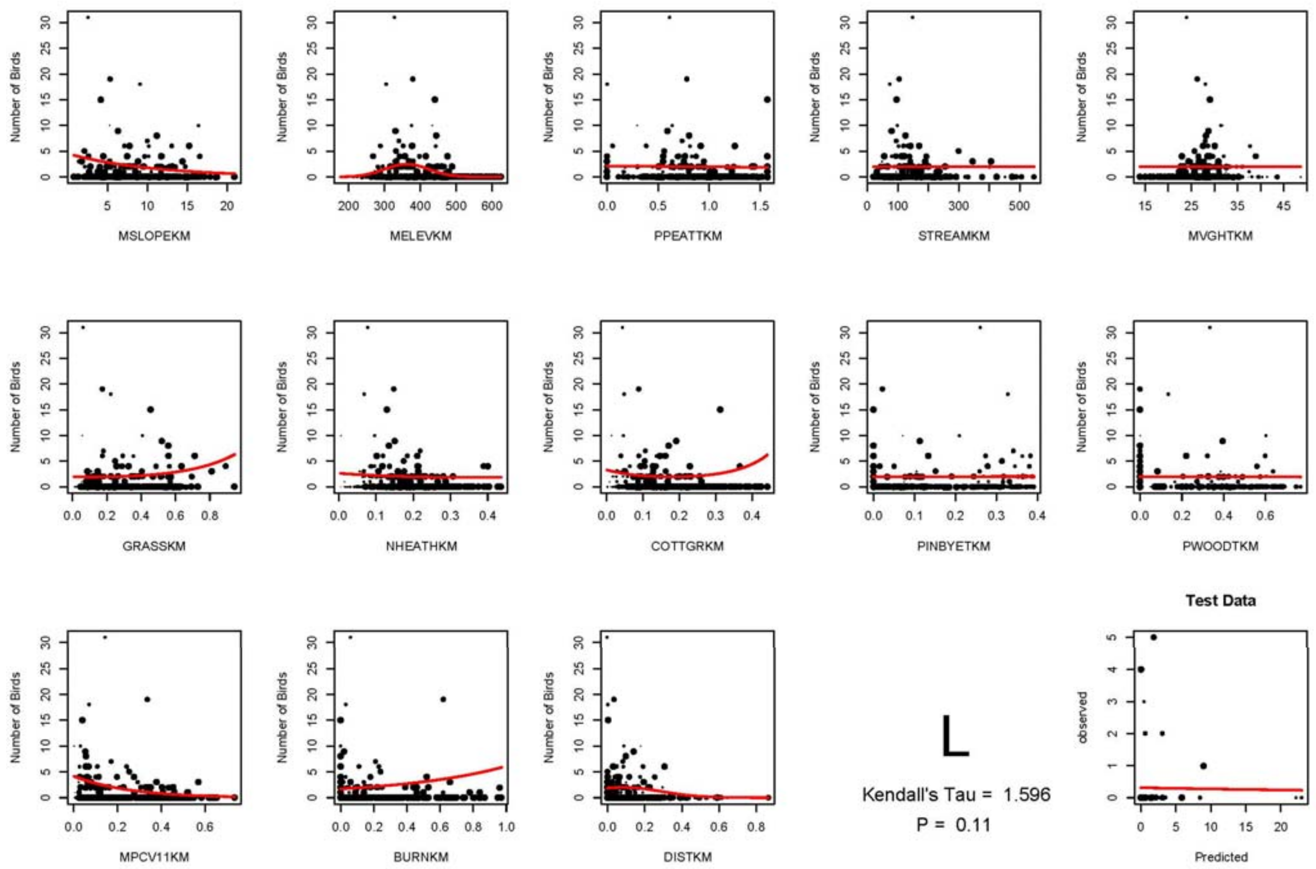


Figure 3.2d: Results from the 1 km-scale analysis of habitat associations for Lapwing. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

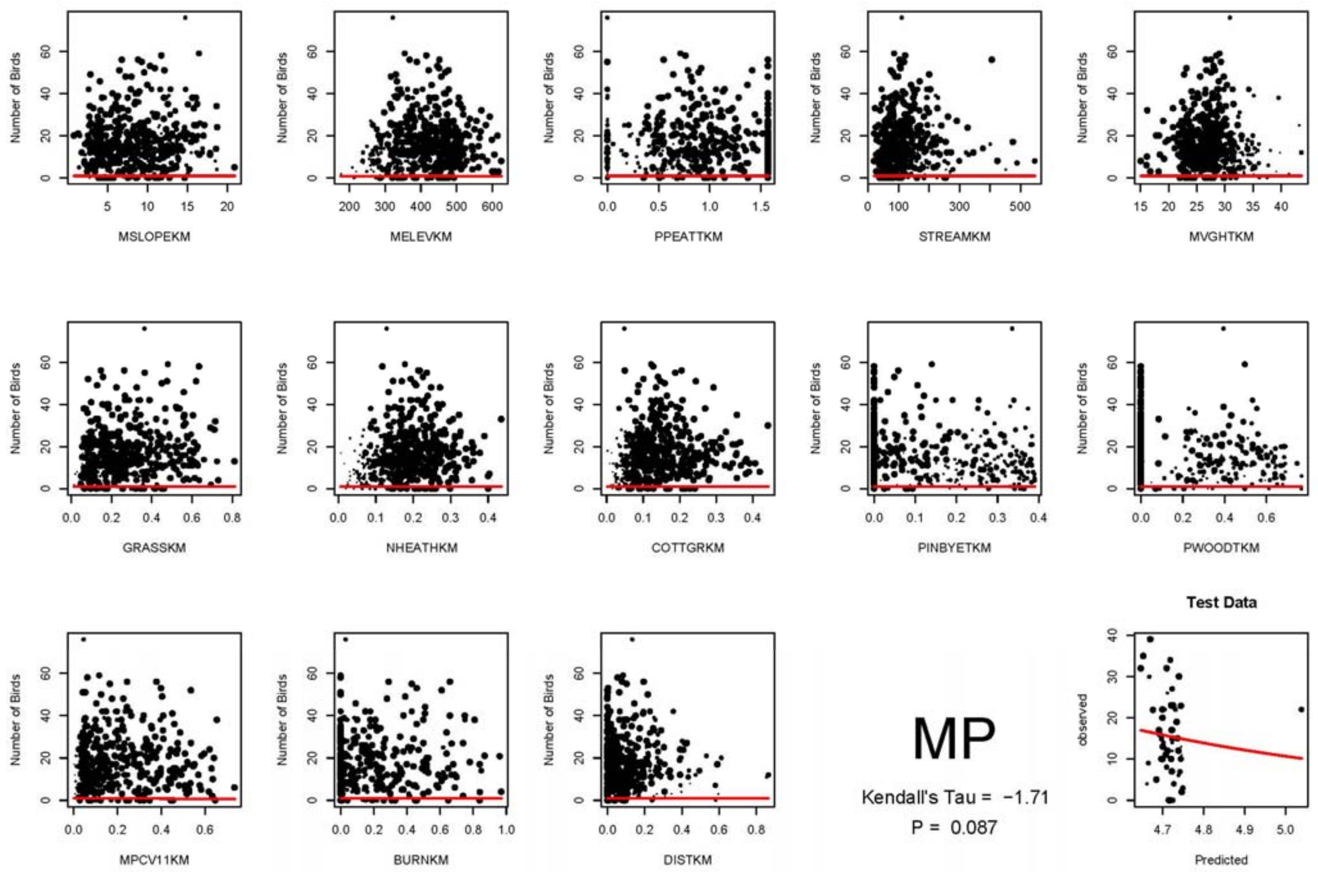


Figure 3.2e: Results from the 1 km-scale analysis of habitat associations for Meadow Pipits. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

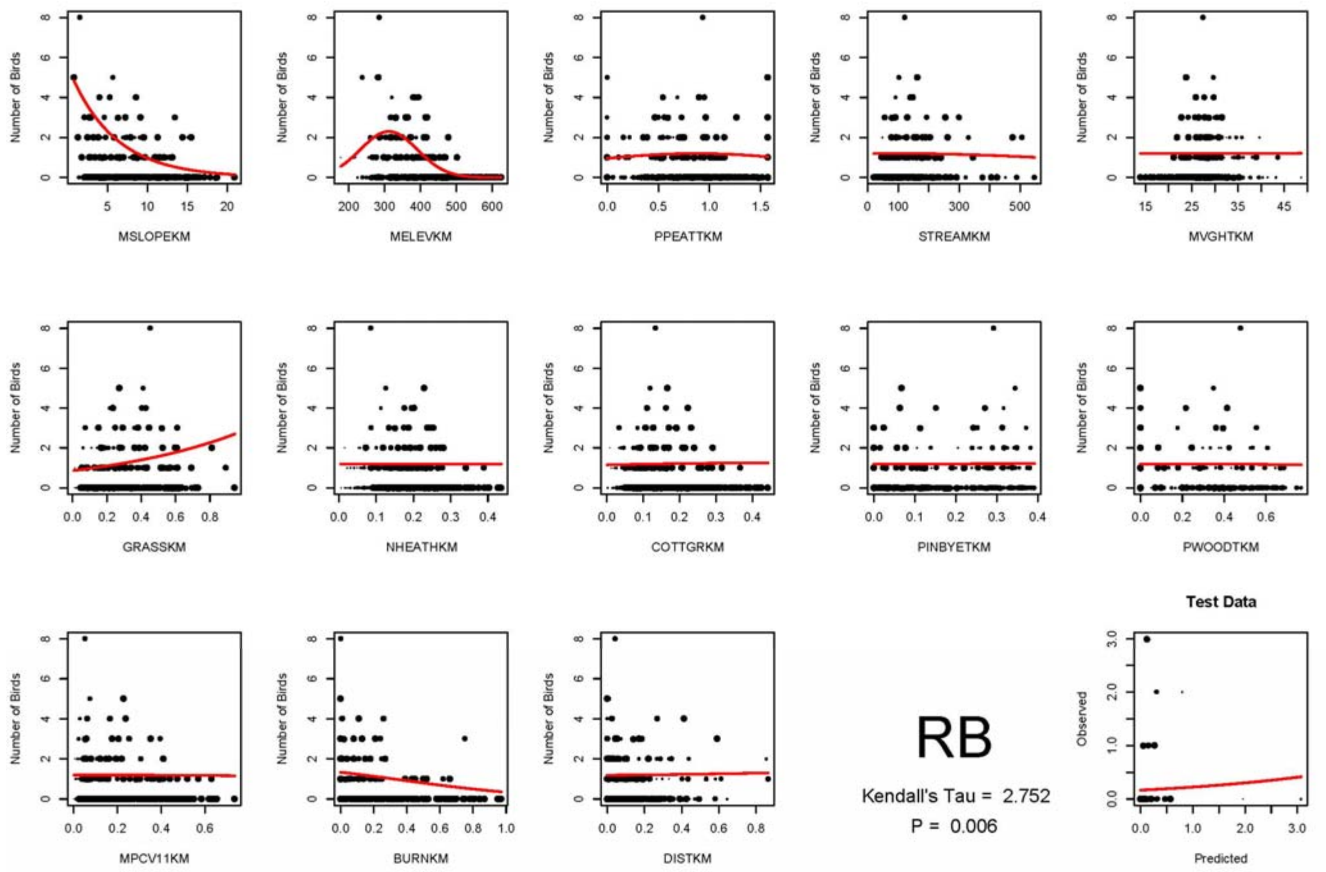


Figure 3.2f: Results from the 1 km-scale analysis of habitat associations for Reed Bunting. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

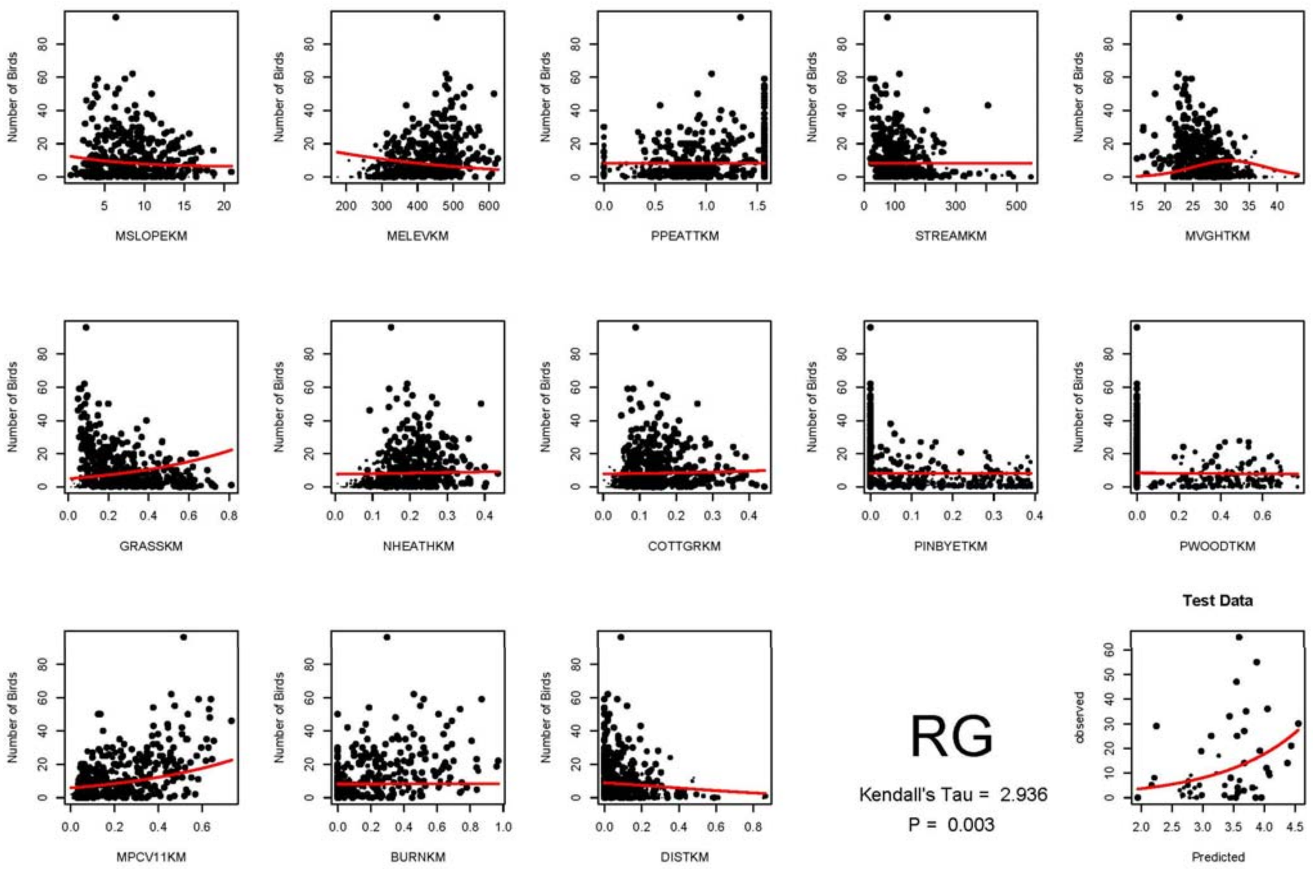


Figure 3.2g: Results from the 1 km-scale analysis of habitat associations for Red Grouse. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

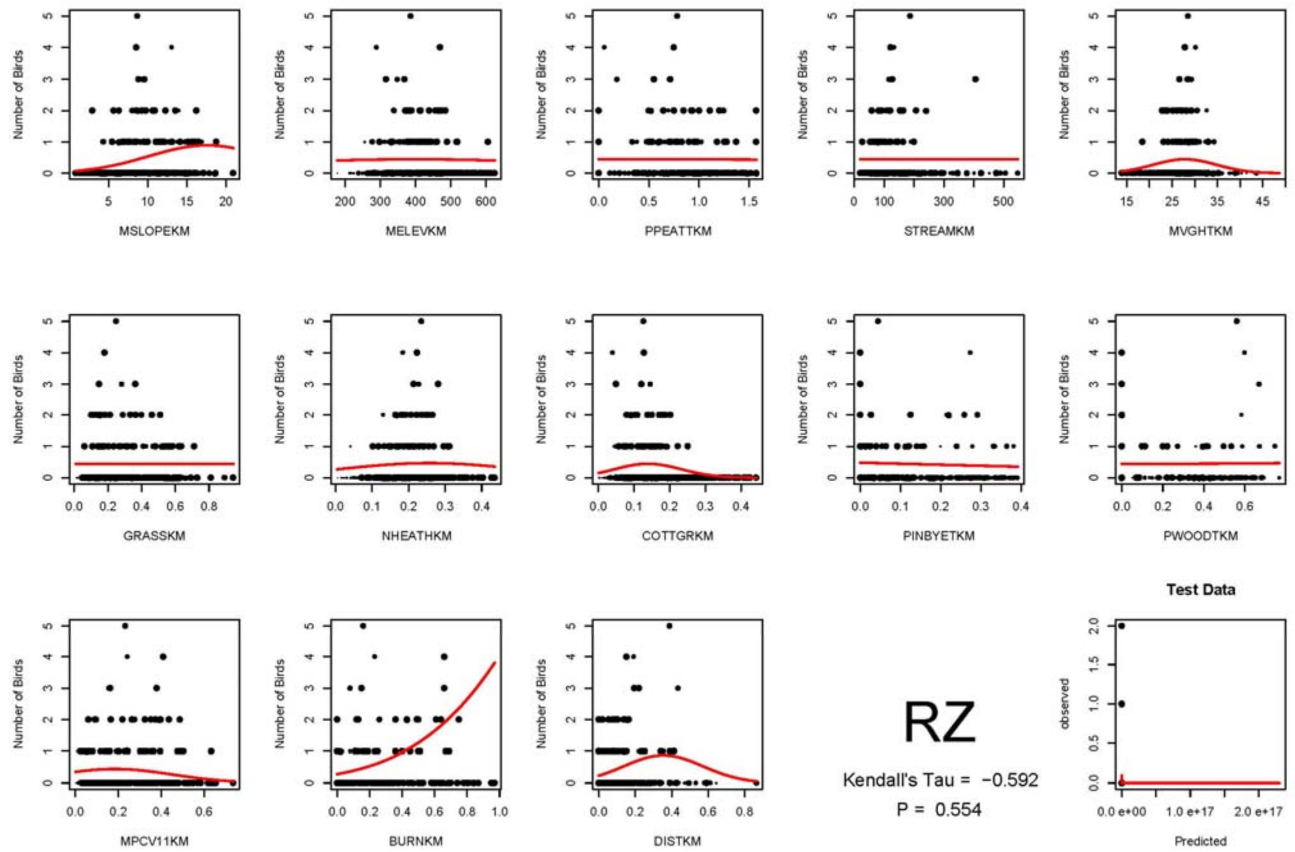


Figure 3.2h: Results from the 1 km-scale analysis of habitat associations for Ring Ouzel. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

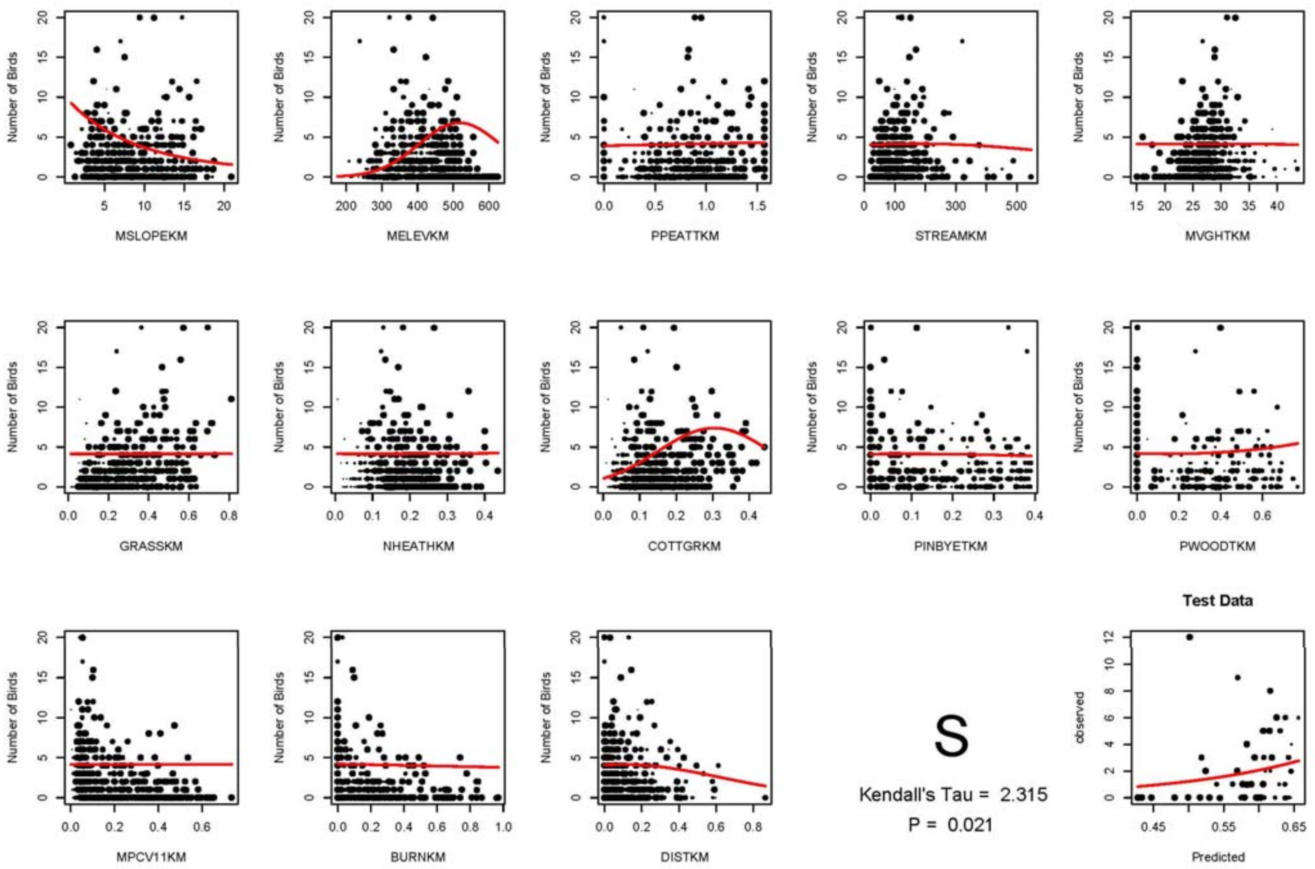


Figure 3.2i: Results from the 1 km-scale analysis of habitat associations for Skylark. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

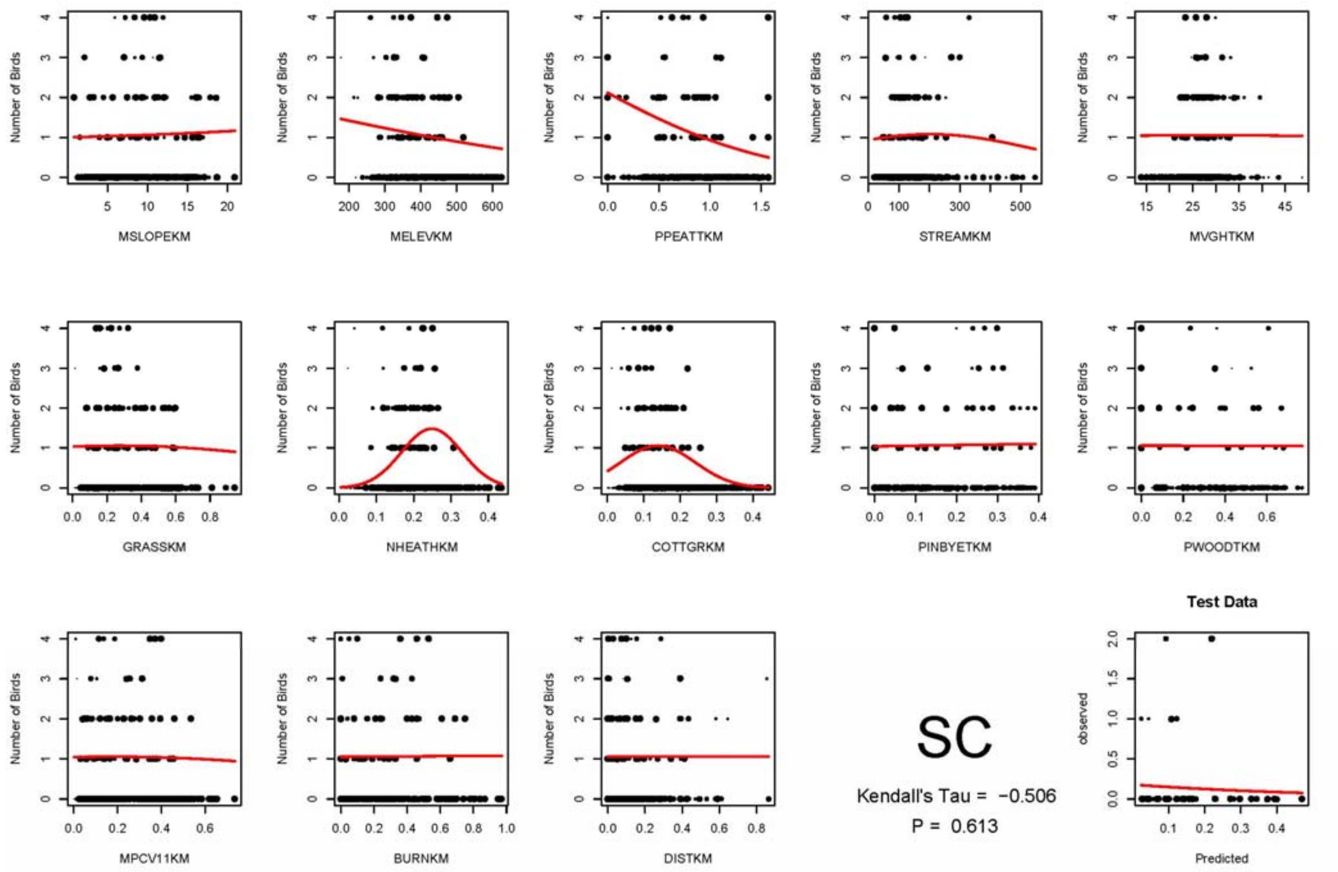


Figure 3.2j: Results from the 1 km-scale analysis of habitat associations for Stonechat. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

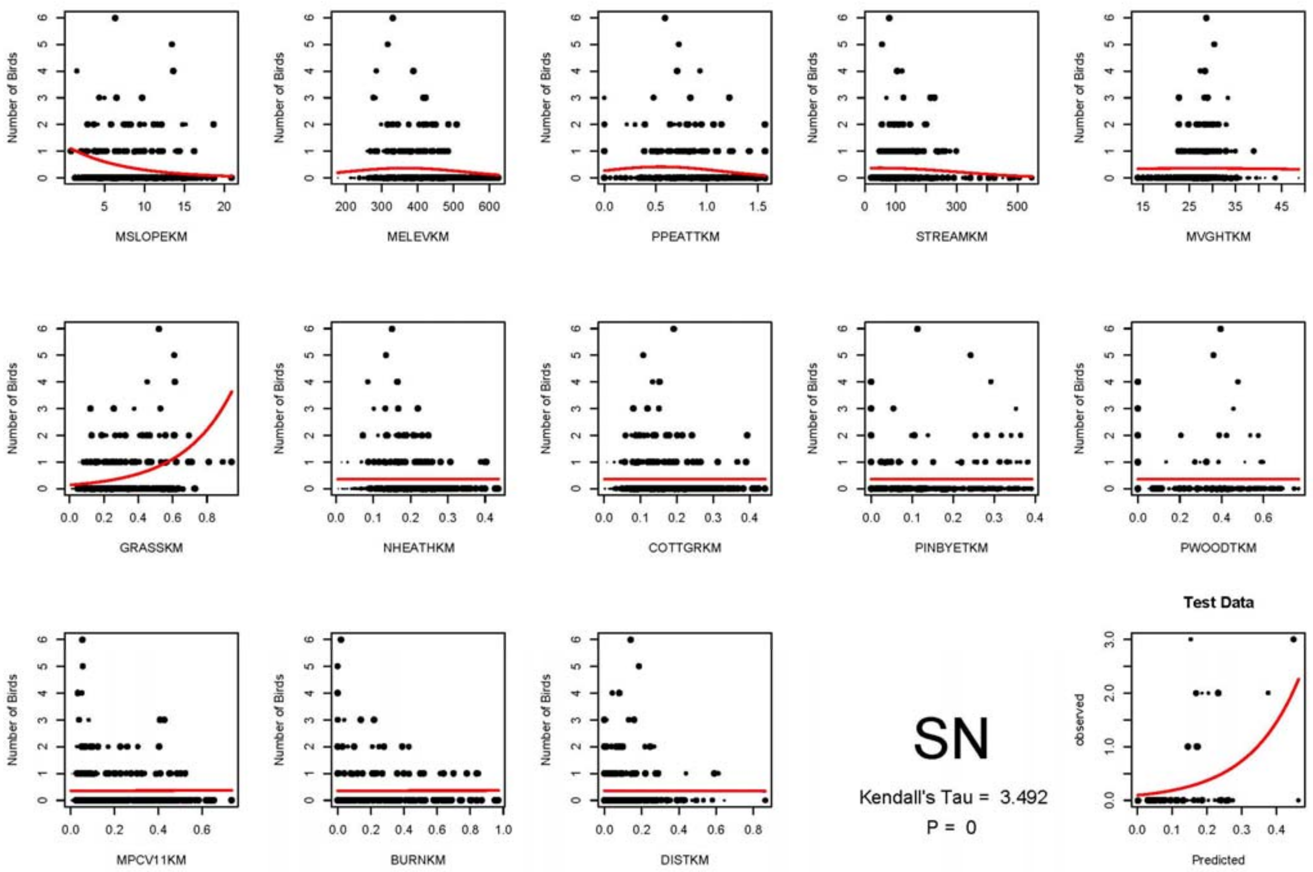


Figure 3.2k: Results from the 1 km-scale analysis of habitat associations for Snipe. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

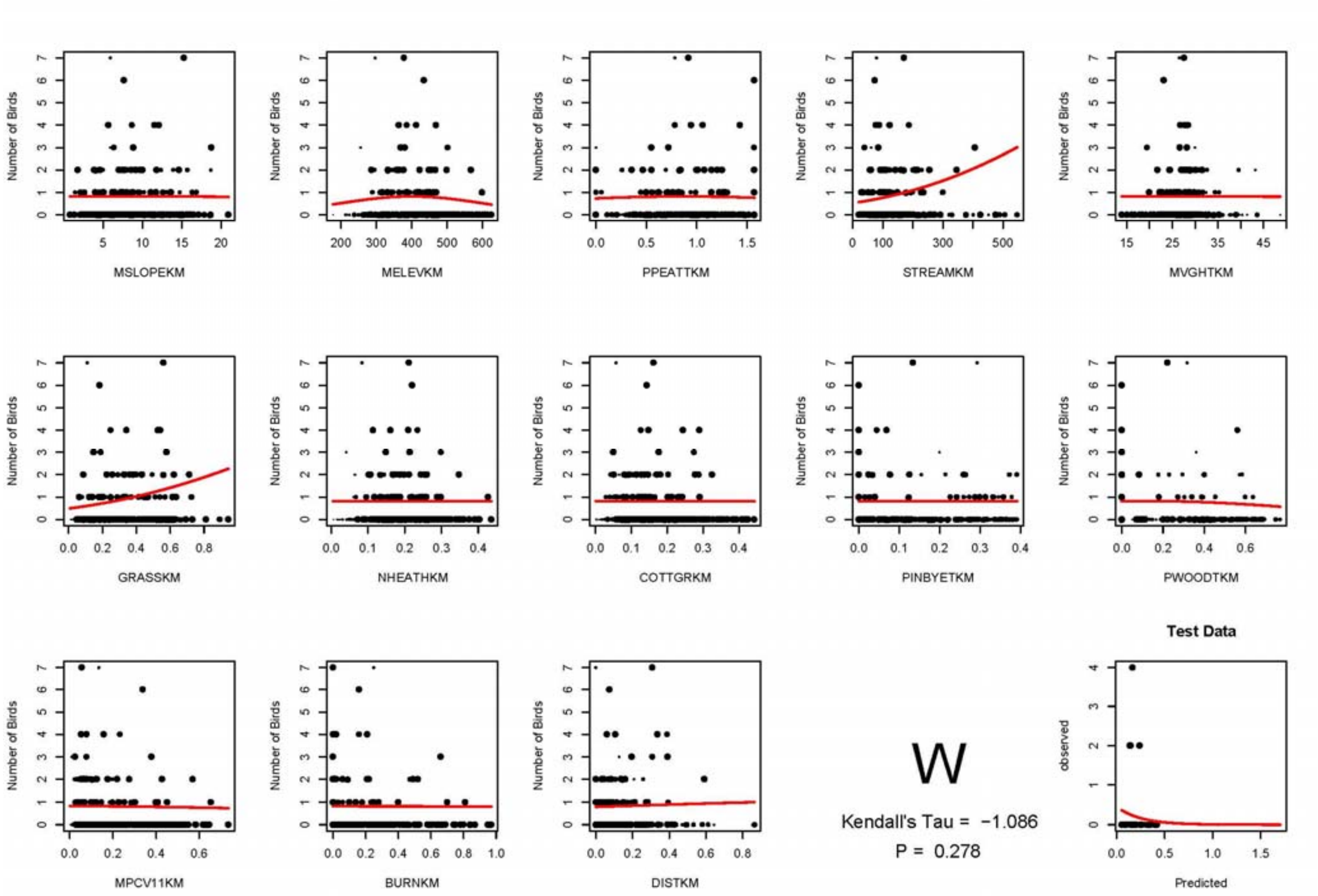


Figure 3.2l: Results from the 1 km-scale analysis of habitat associations for Wheatear. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

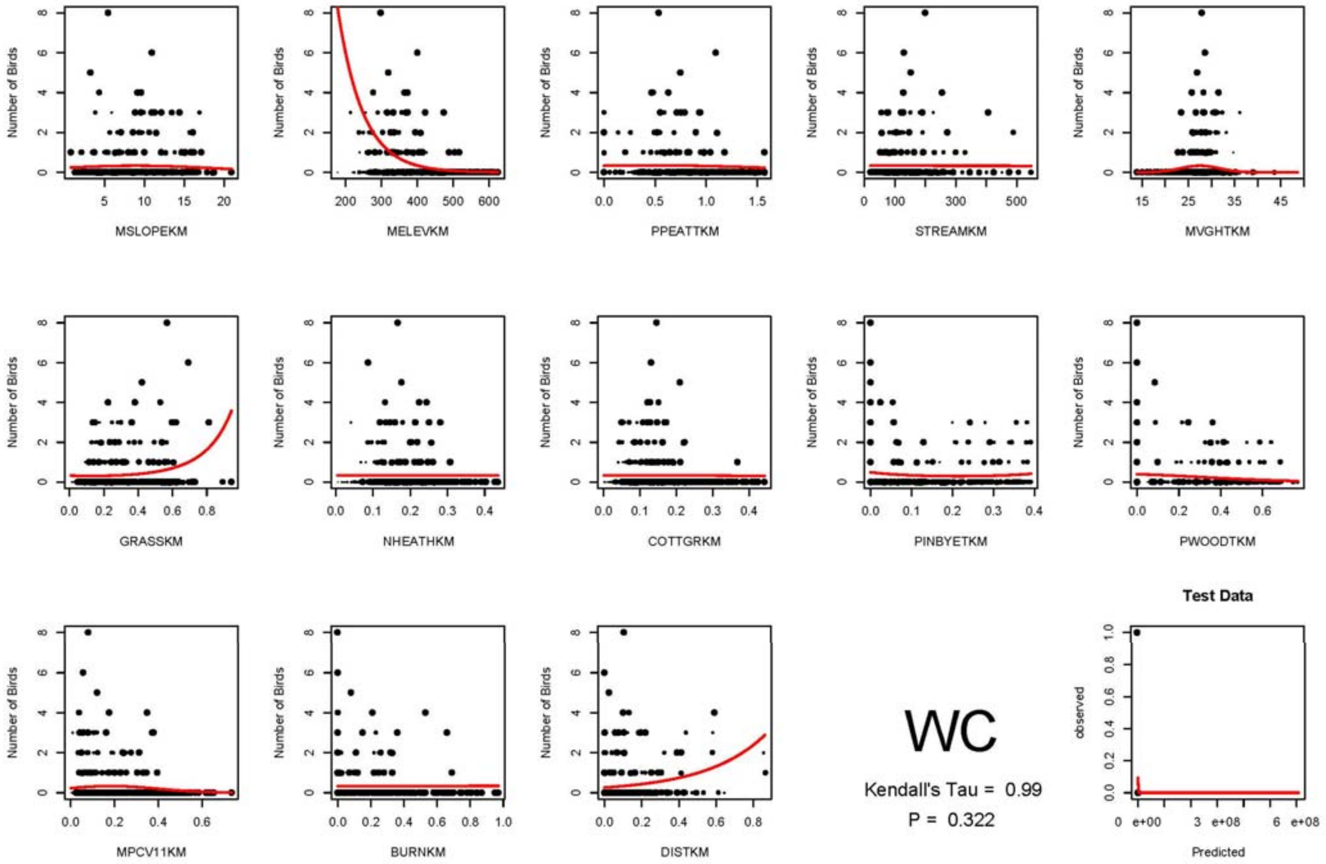


Figure 3.2m: Results from the 1 km-scale analysis of habitat associations for Whinchat. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

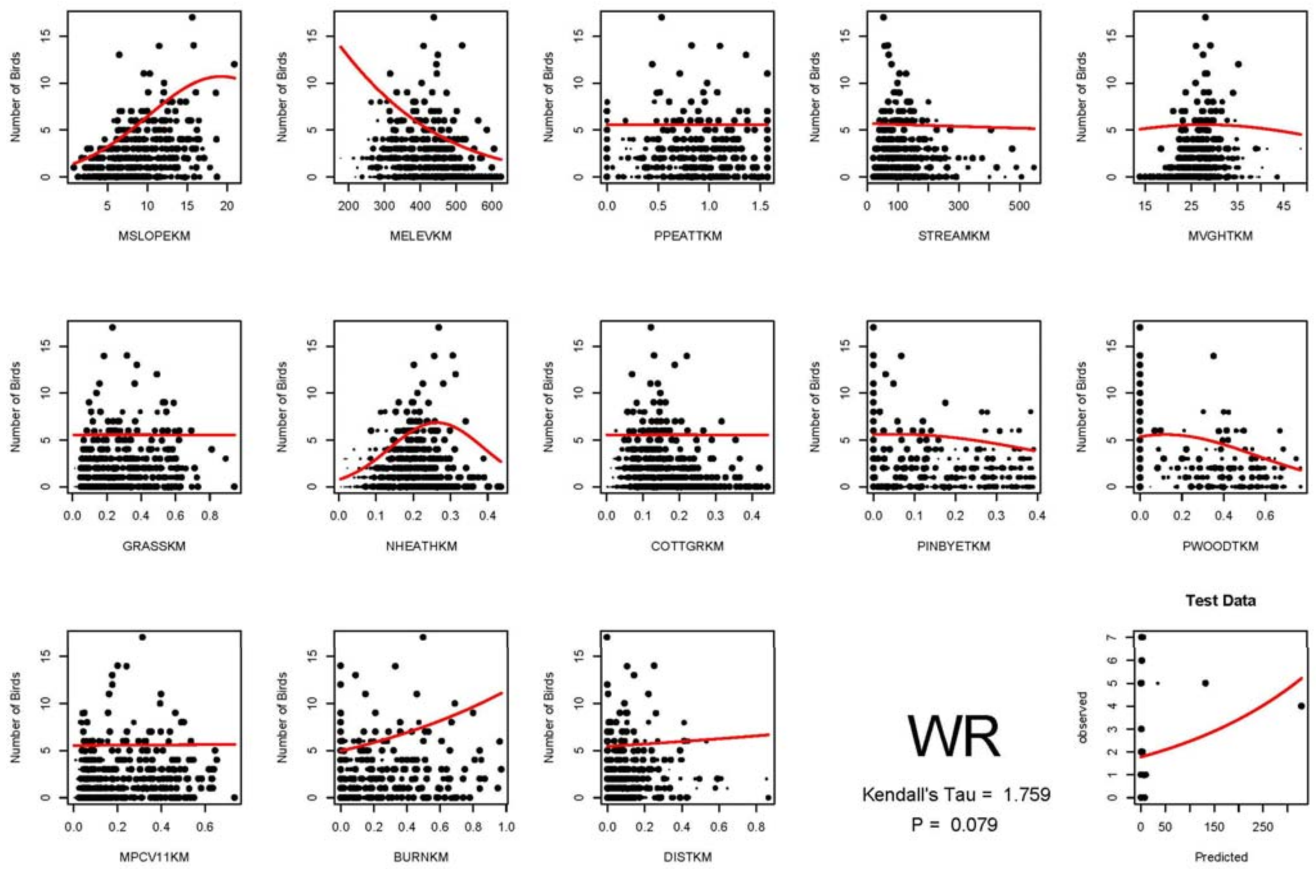


Figure 3.2n: Results from the 1 km-scale analysis of habitat associations for Curlew. Graphs indicate the form of the relationship between bird abundance and a particular explanatory variable (red line) compared with the observed data. The ability of the model to predict bird abundance is indicated in the graph in the bottom right-hand corner, which plots predicted against observed abundance across the test squares.

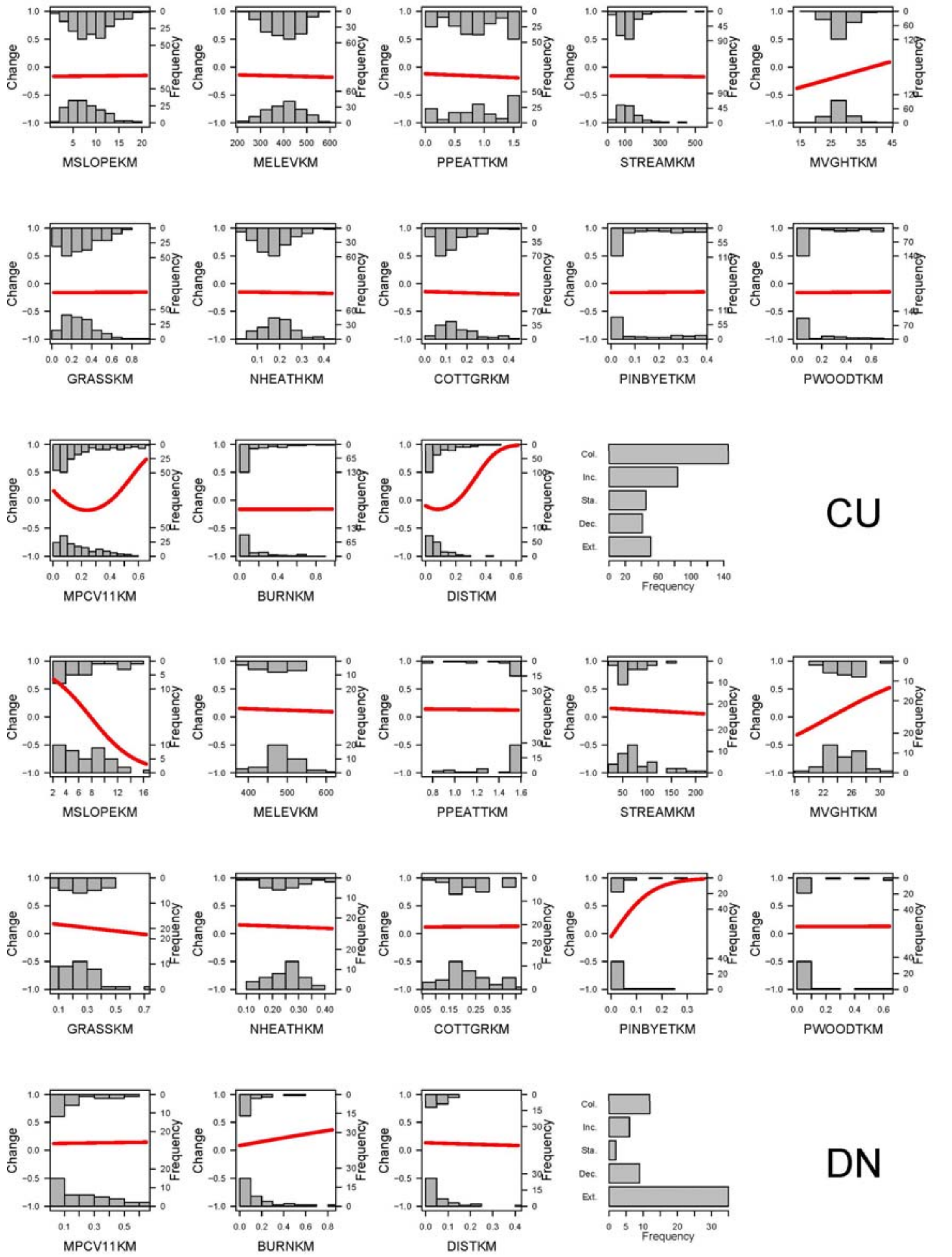


Figure 3.3a. Results from the 1 km analysis of change in bird populations for Curlew (top) and Dunlin (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

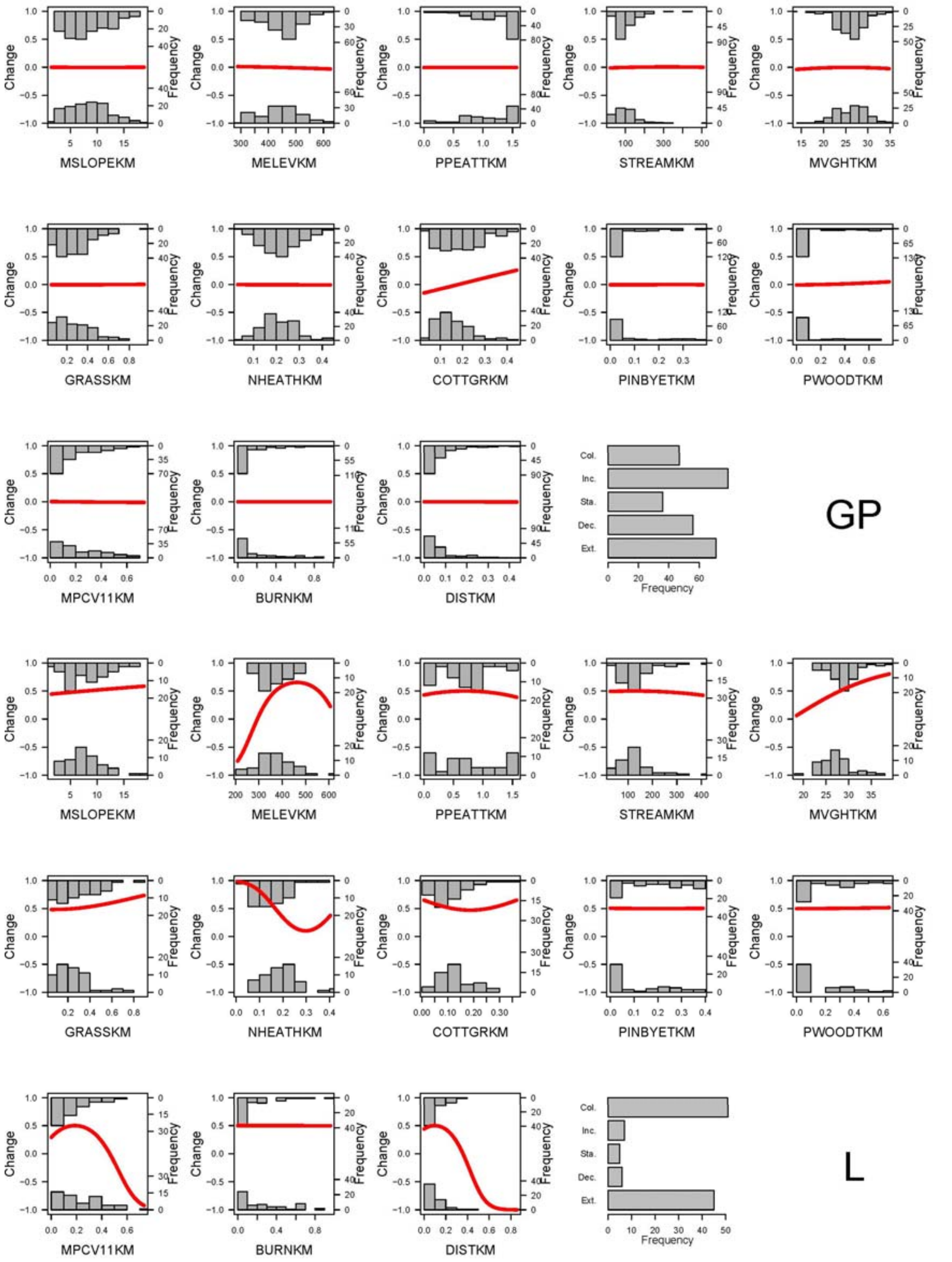


Figure 3.3b. Results from the 1 km analysis of change in bird populations for Golden Plover (top) and Lapwing (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

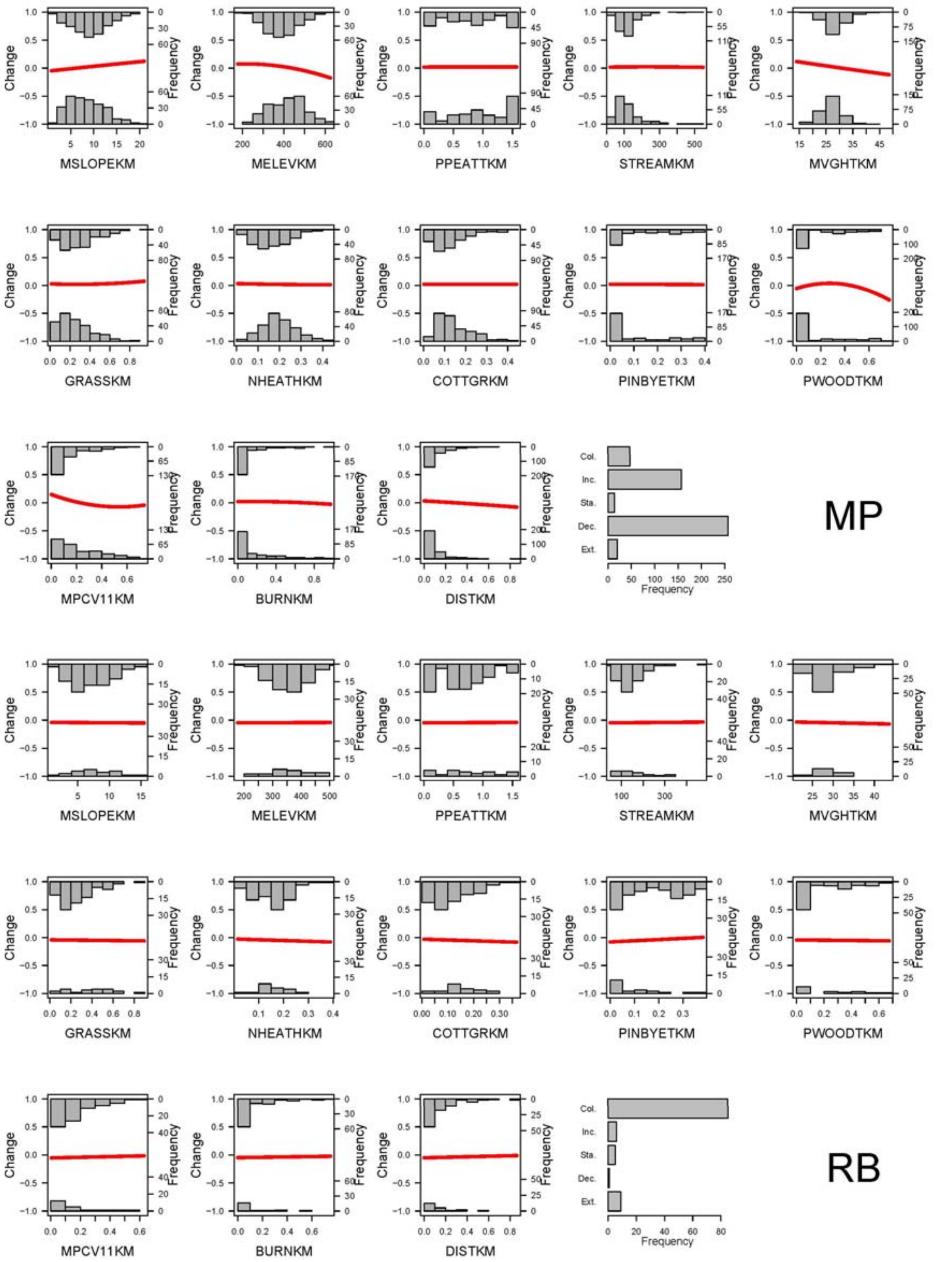


Figure 3.3c. Results from the 1 km analysis of change in bird populations for Meadow Pipit (top) and Reed Bunting (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

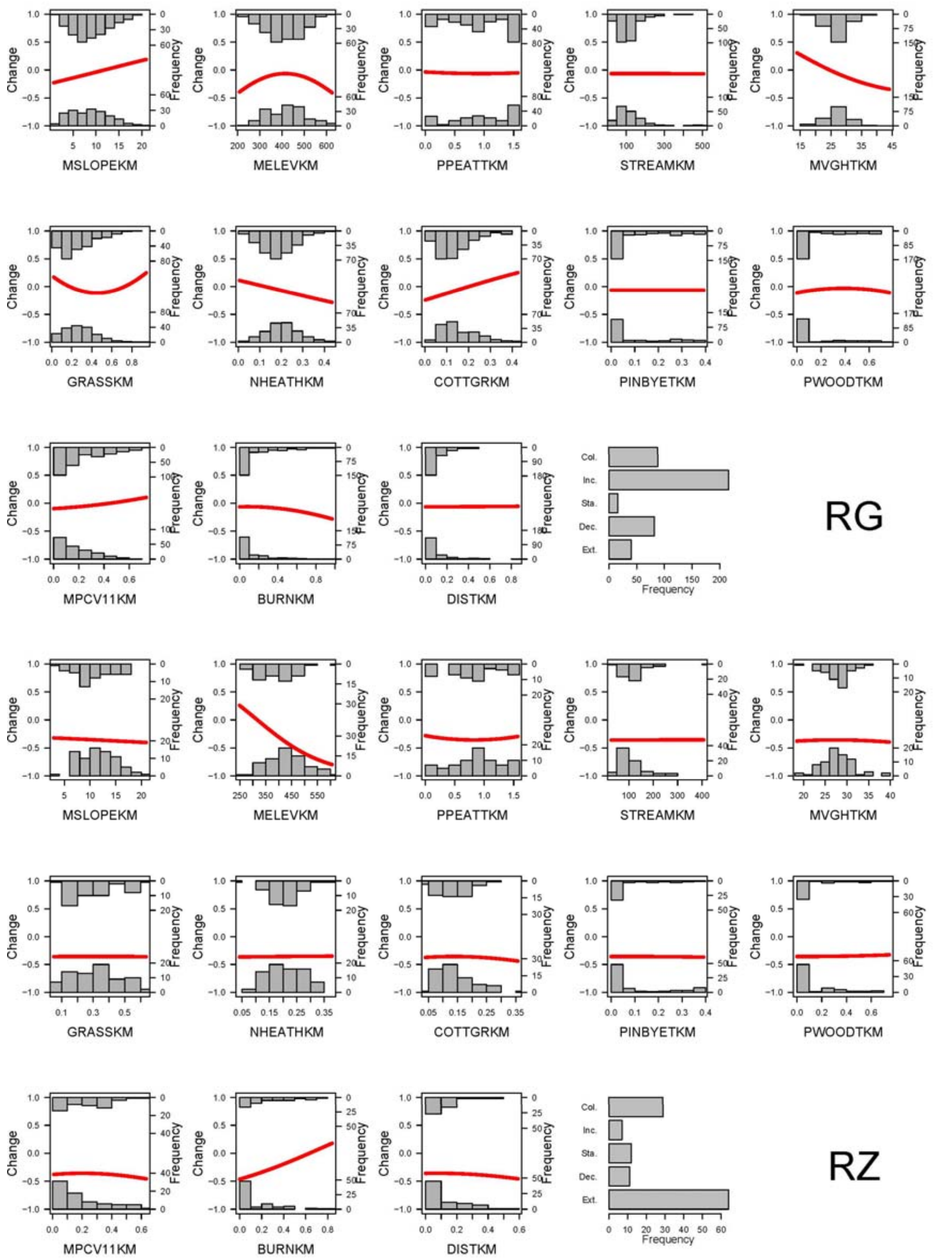


Figure 3.3d. Results from the 1 km analysis of change in bird populations for Red Grouse (top) and Ring Ouzel (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

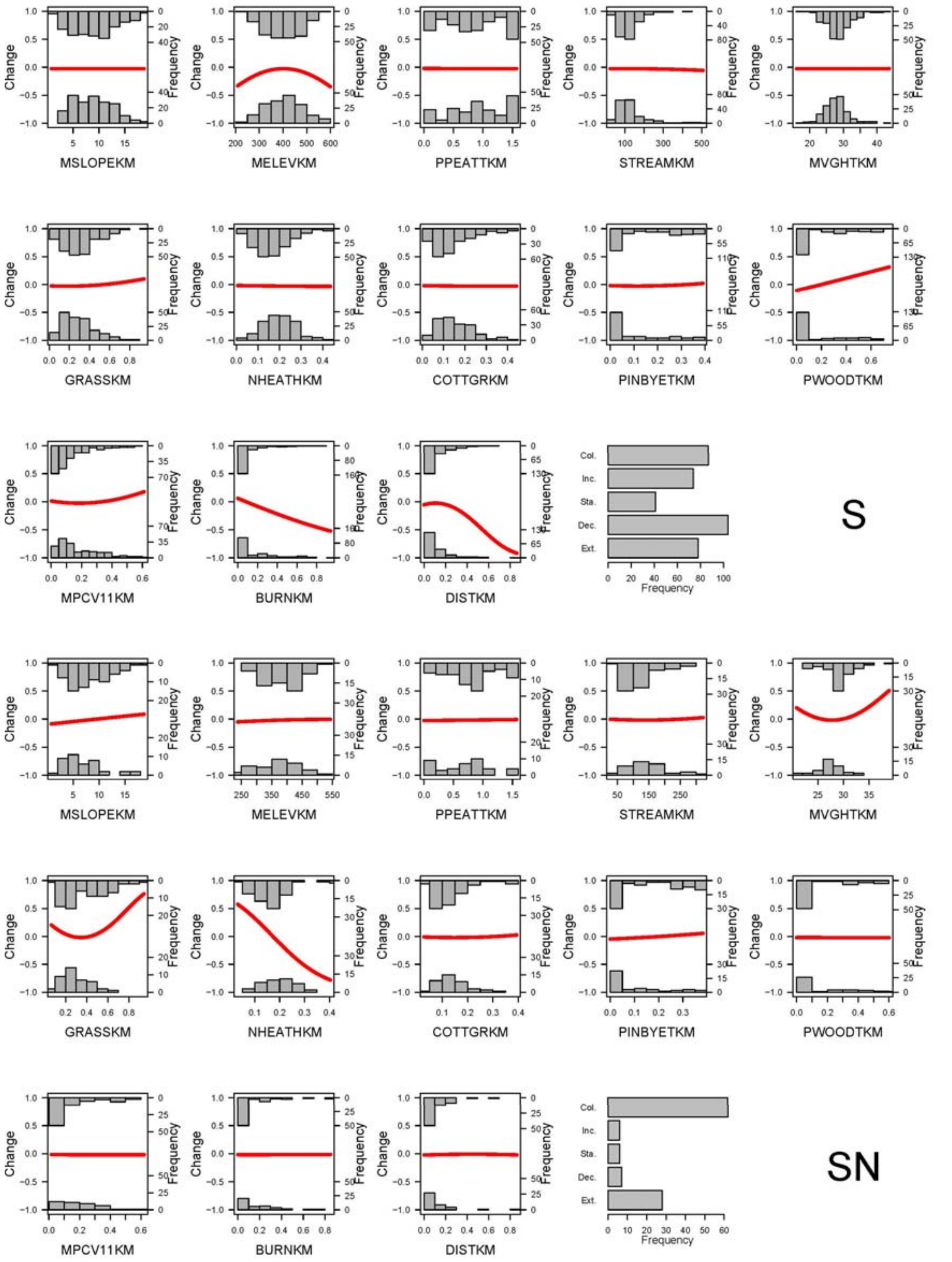


Figure 3.3e. Results from the 1 km analysis of change in bird populations for Skylark (top) and Snipe (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

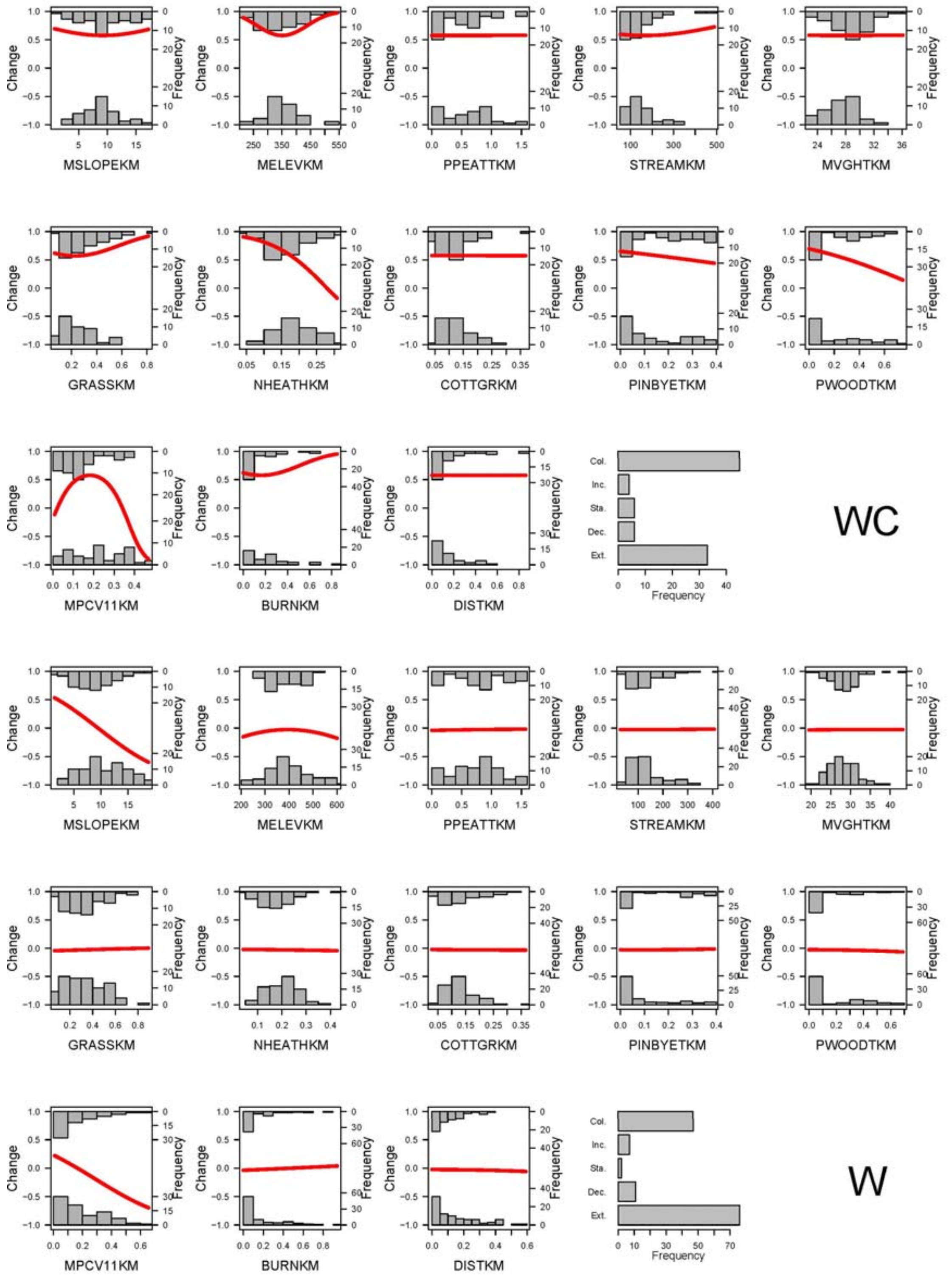
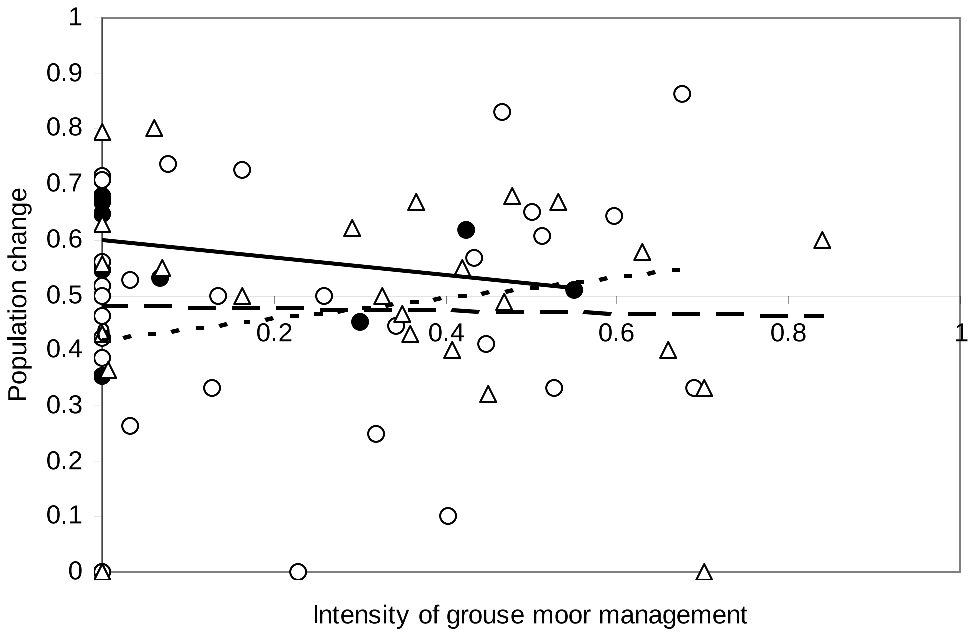


Figure 3.3f. Results from the 1 km analysis of change in bird populations for Whinchat (top) and Wheatear (bottom). Graphs indicate the form of relationships between change (positive values indicate a population increase, and negative values, a decline) and a particular explanatory variable (red line). The distribution of the underlying data is given by the histogram for squares within an increasing population (top) compared with a decline (bottom). The graph at the bottom right-hand corner indicates the number of squares which were colonized, contained increasing, stable, declining populations, or where the birds became extinct (see text).

a) Golden Plover



b) Dunlin

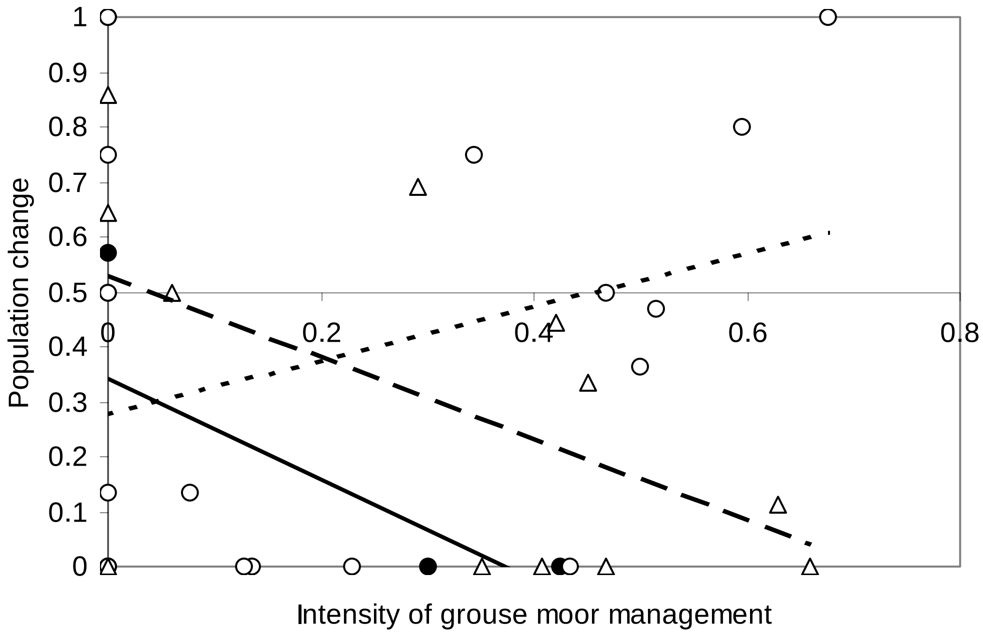


Figure 3.4: The relationship between an index of population change ((1990 – 2004) / (1990+2004)) and the intensity of grouse-moor management across three tiers of grazing management (LGU 0, filled circle, solid line; LGU 1, open circle, dotted line; LGU 2, triangle, dashed line).

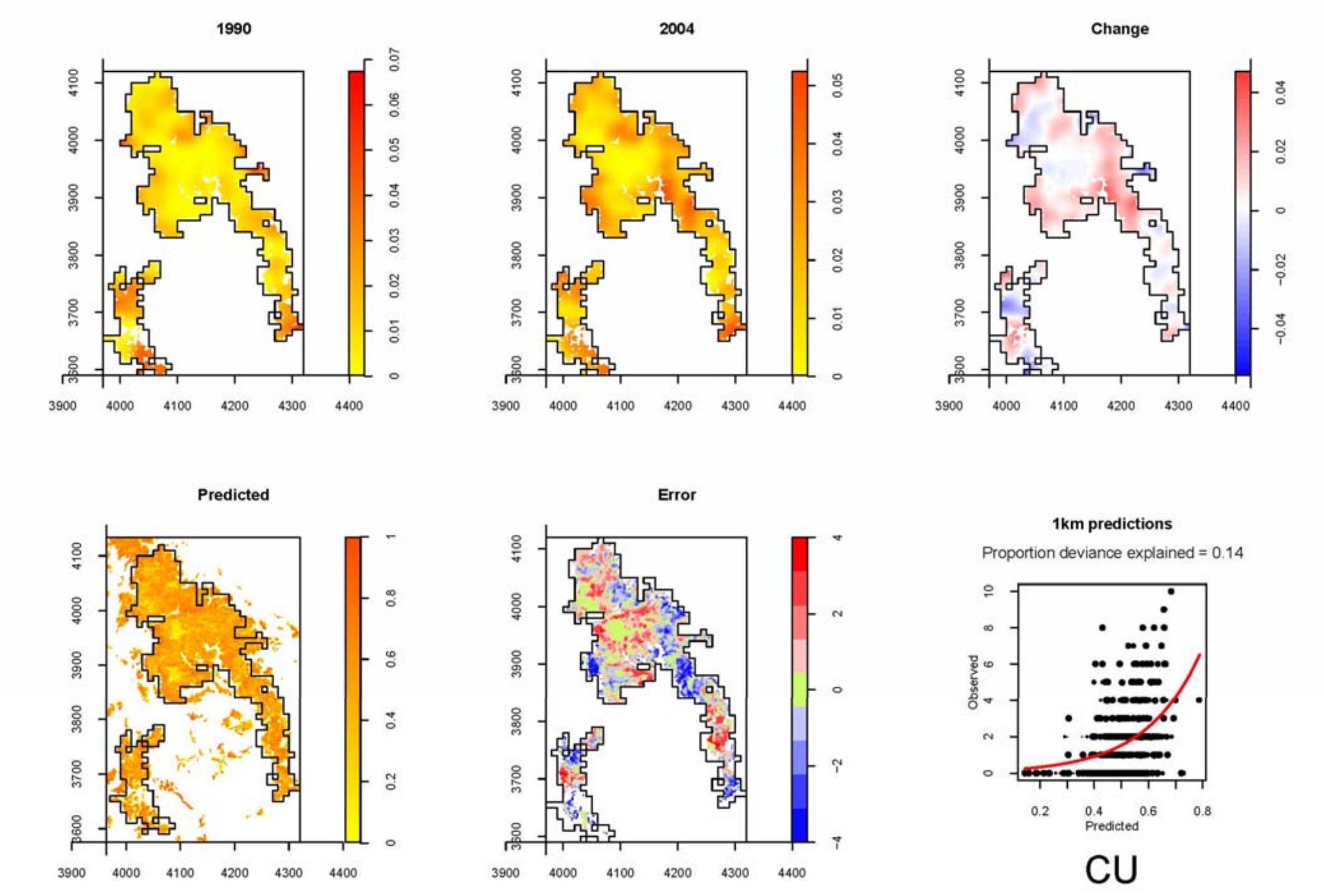


Figure 3.5a: Smoothed maps outlining the probability of Curlew occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

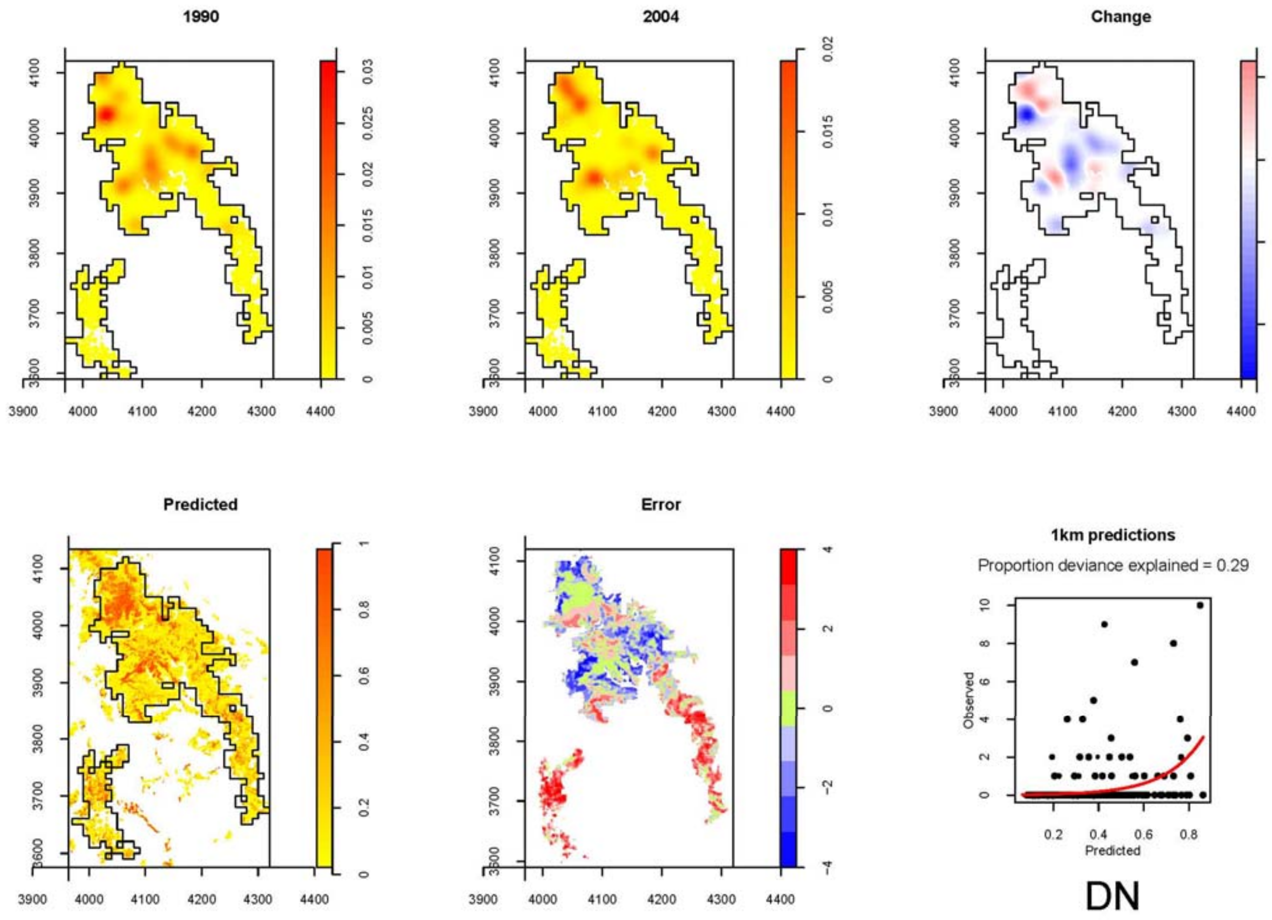


Figure 3.5b: Smoothed maps outlining the probability of Dunlin occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

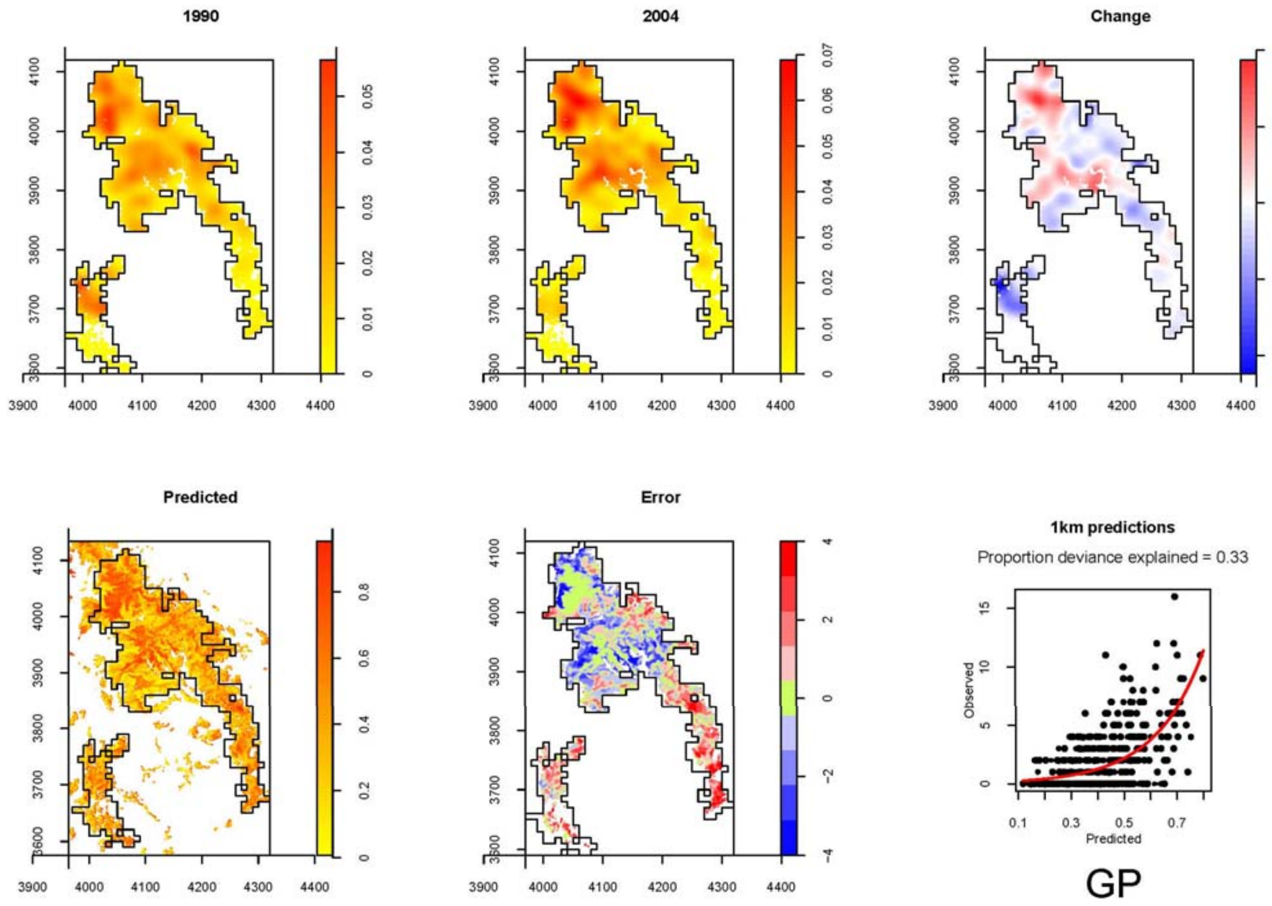


Figure 3.5c: Smoothed maps outlining the probability of Golden Plover occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

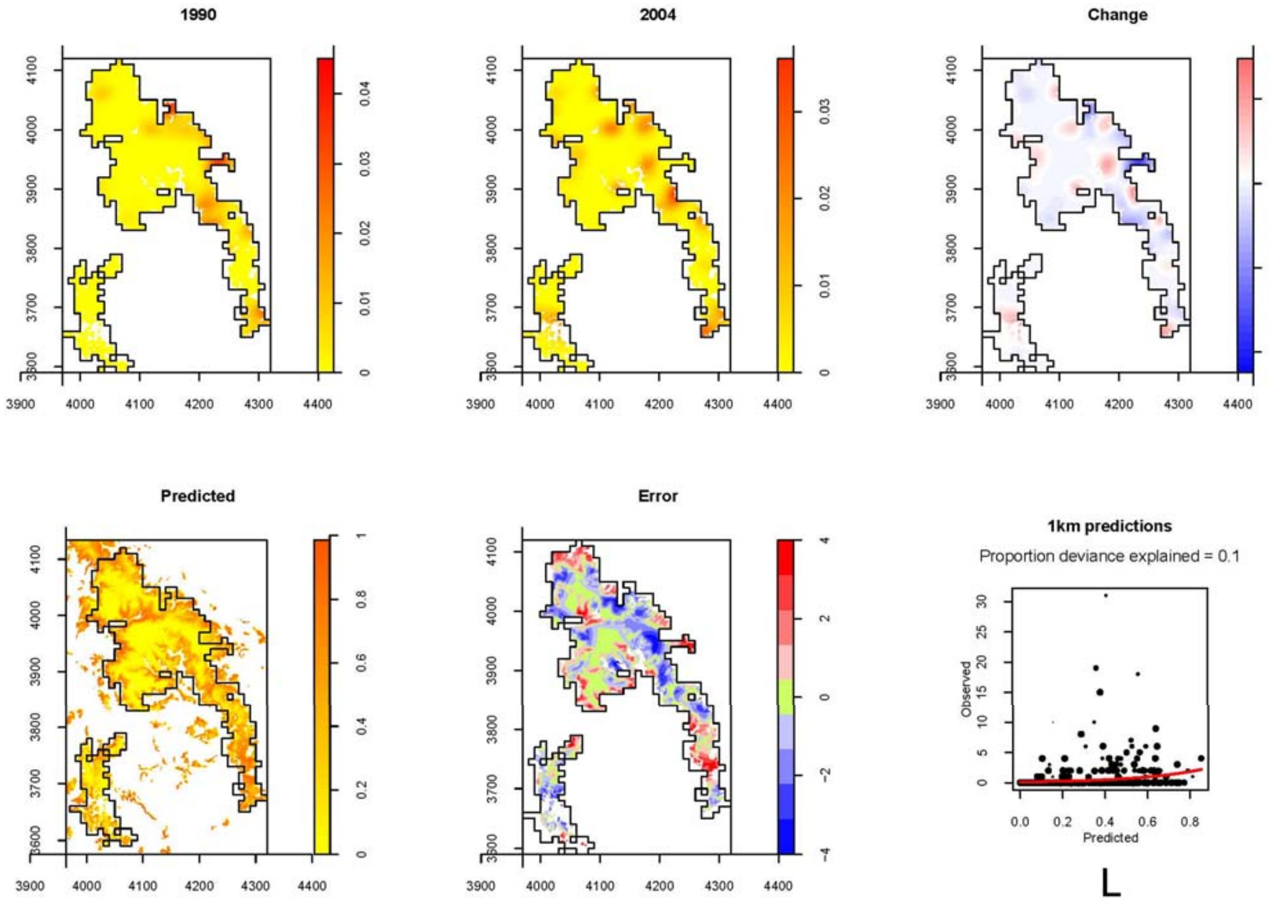


Figure 3.5d: Smoothed maps outlining the probability of Lapwing occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

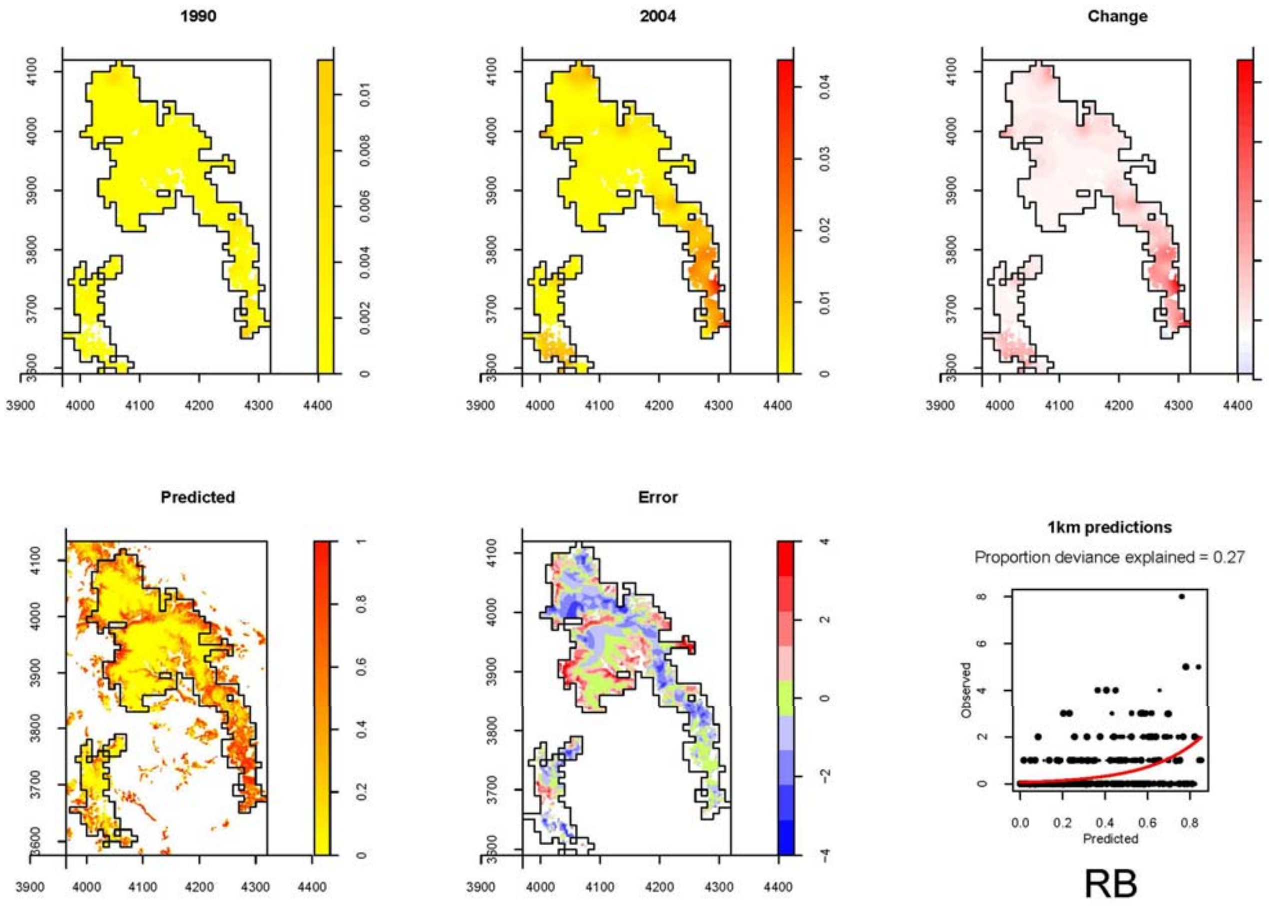


Figure 3.5e: Smoothed maps outlining the probability of Reed Bunting occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

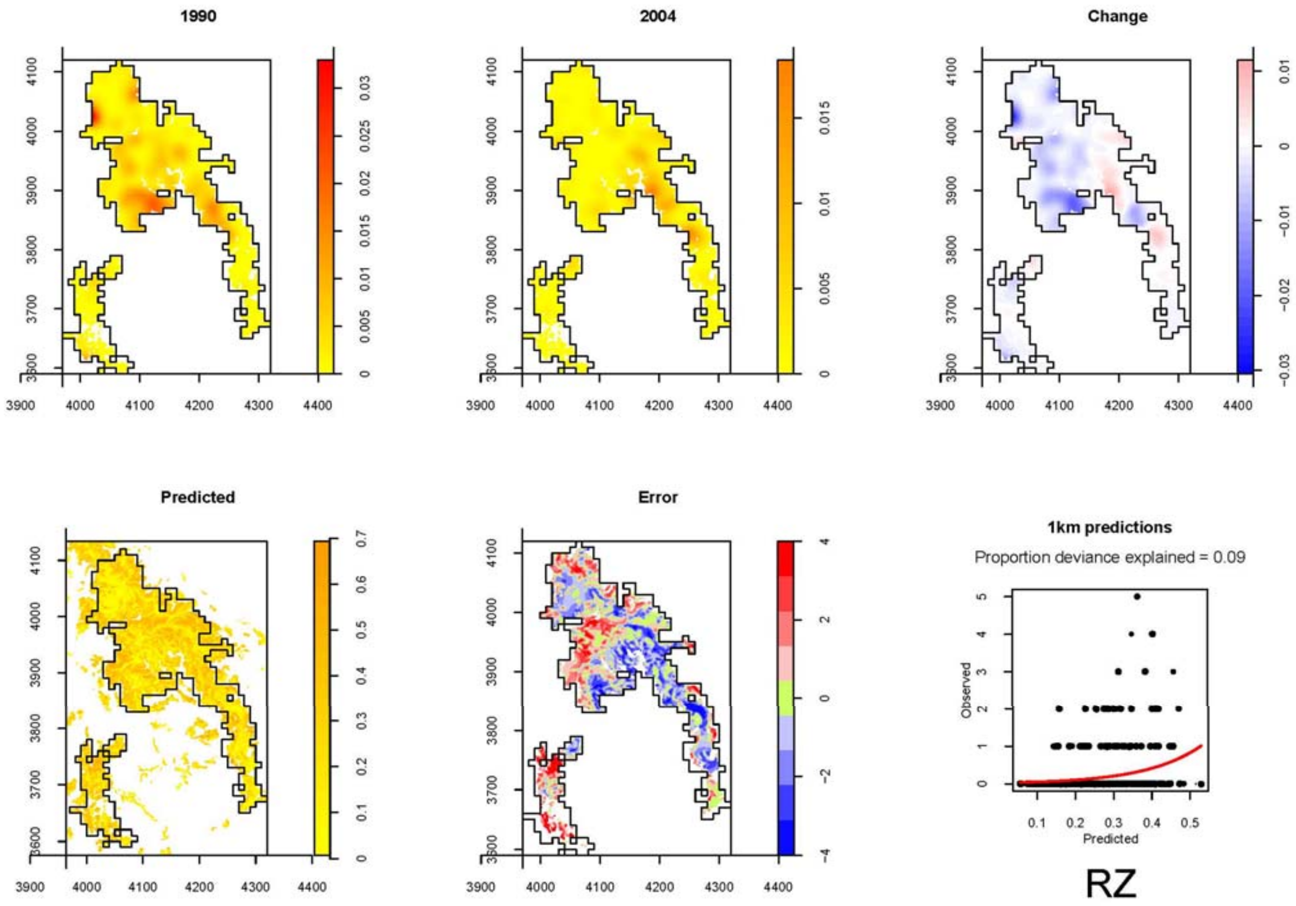


Figure 3.5f: Smoothed maps outlining the probability of Ring Ouzel occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

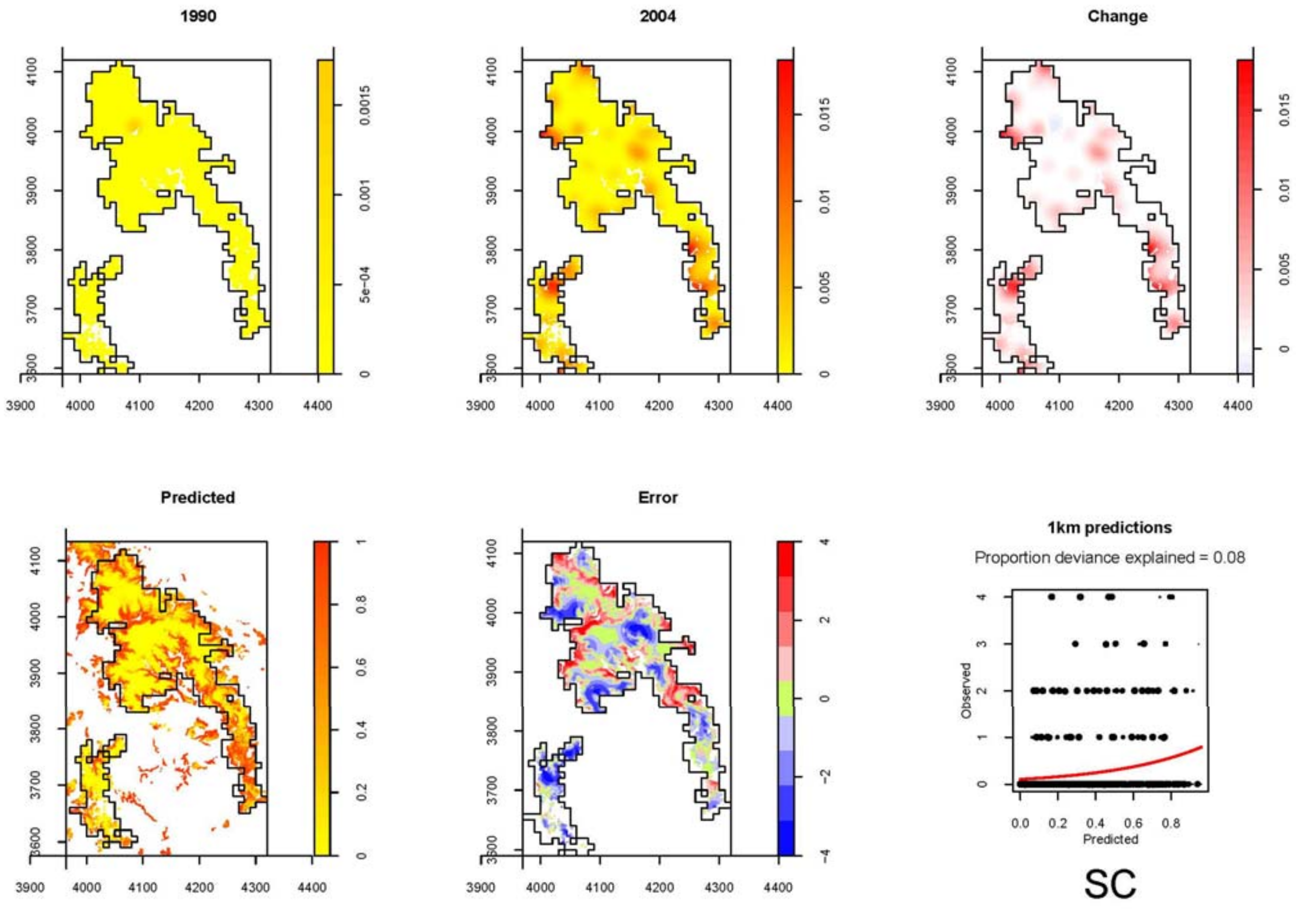


Figure 3.5g: Smoothed maps outlining the probability of Stonechat occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

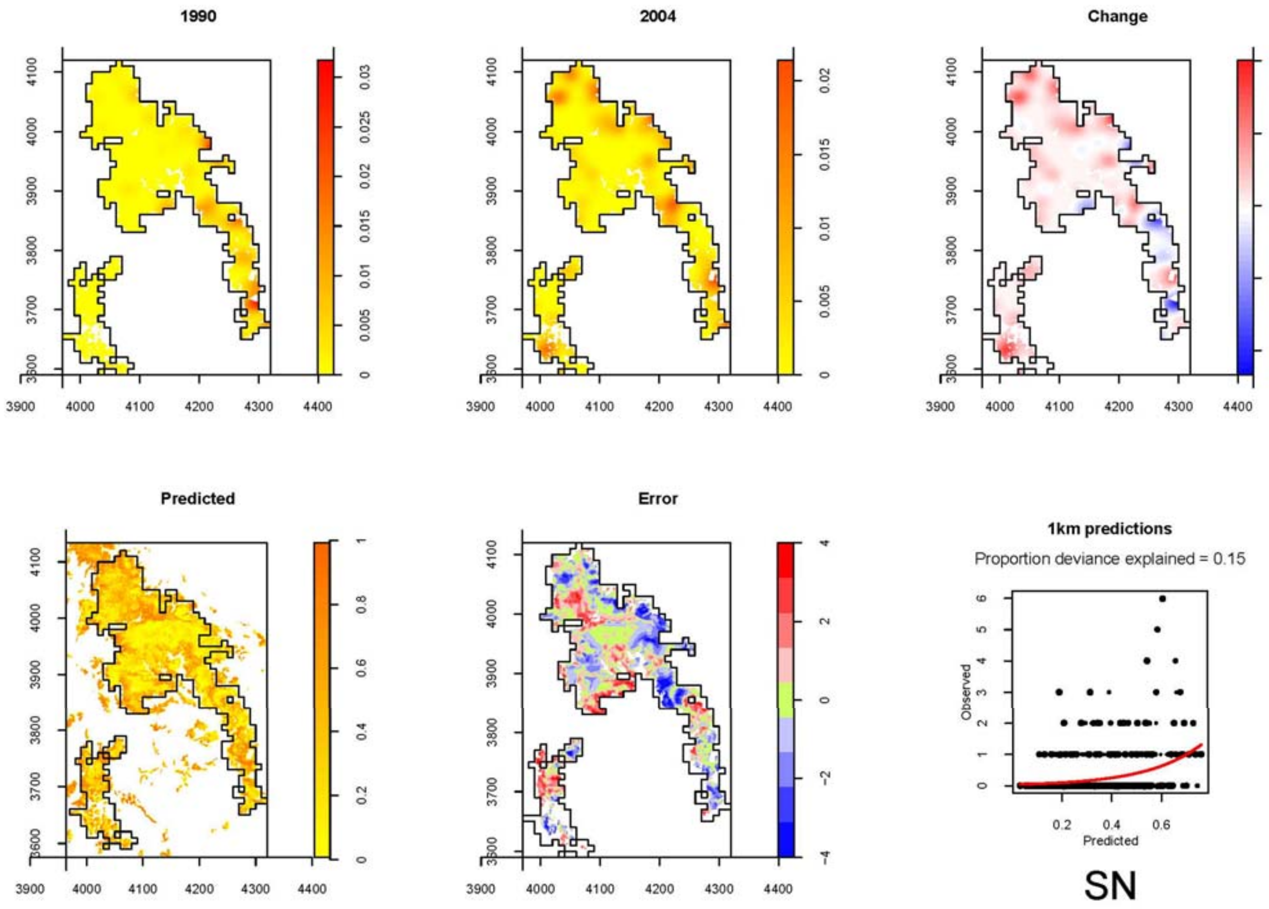


Figure 3.5h: Smoothed maps outlining the probability of Snipe occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

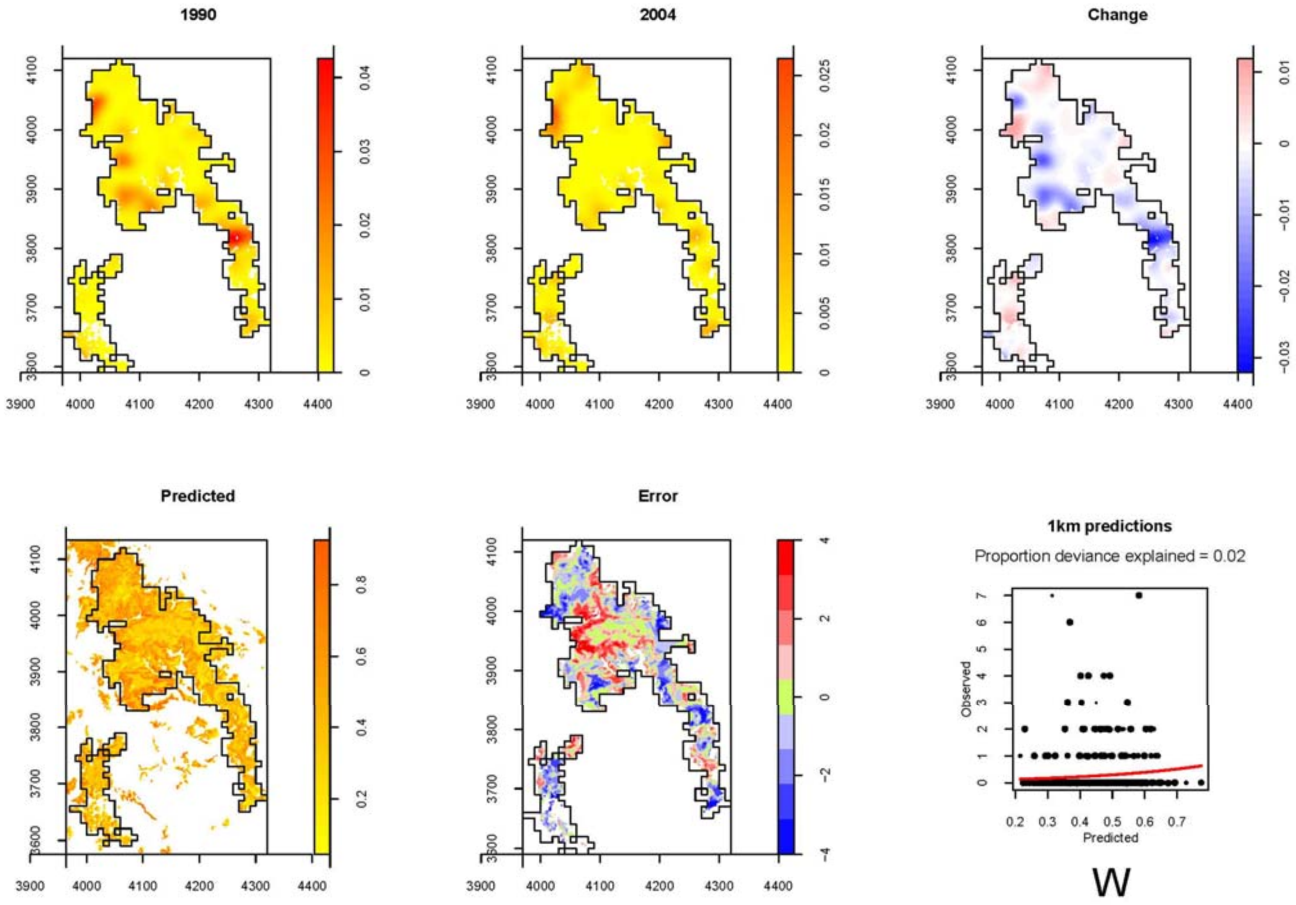


Figure 3.5i: Smoothed maps outlining the probability of Wheatear occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

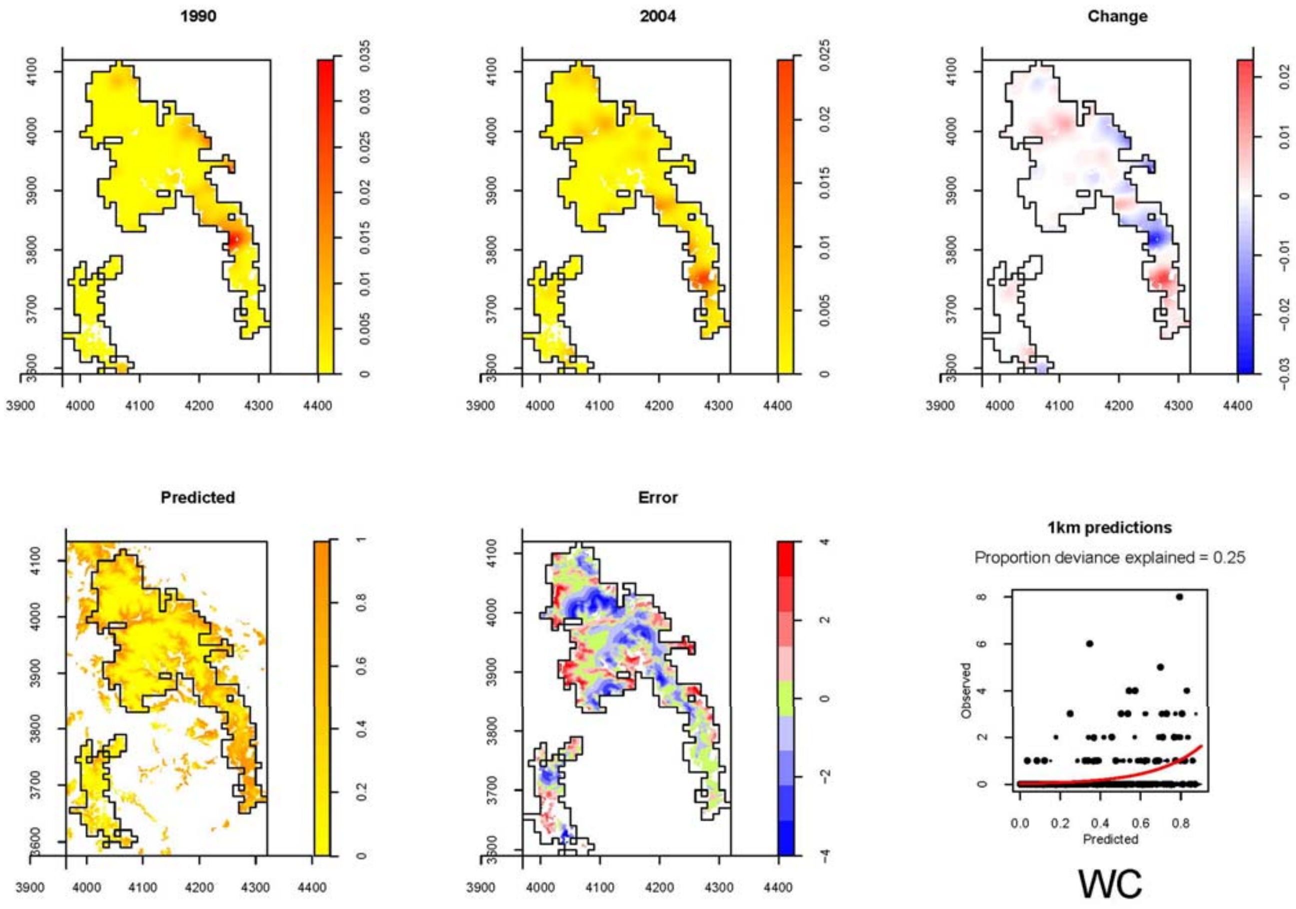


Figure 3.5j: Smoothed maps outlining the probability of Wren occurrence at any 100 m point in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

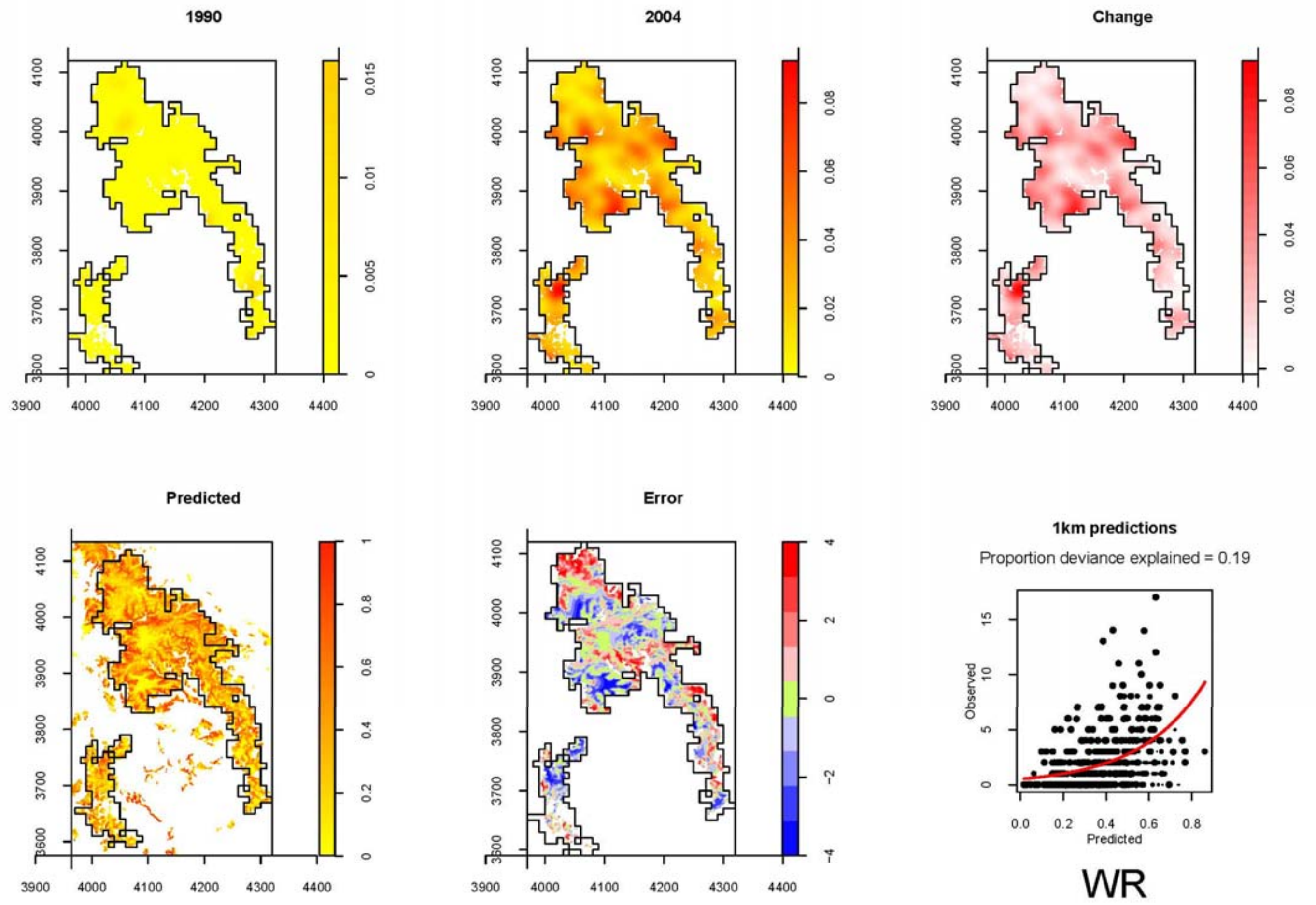


Figure 3.5k: Smoothed maps outlining the probability of Wheatear occurrence at any 100 m oint in 1990, 2004, and a change map produced by subtracting the 1990 probabilities from 2004. The 2004 map can be compared with the predicted map of the fine-scale models (Table 3.2), the accuracy of which at predicting abundance in each 1 km square is indicated by the scatter plot. The error map highlights the areas where the predicted map over-predicts (red) or under-predicts (blue) occurrence the most (see text).

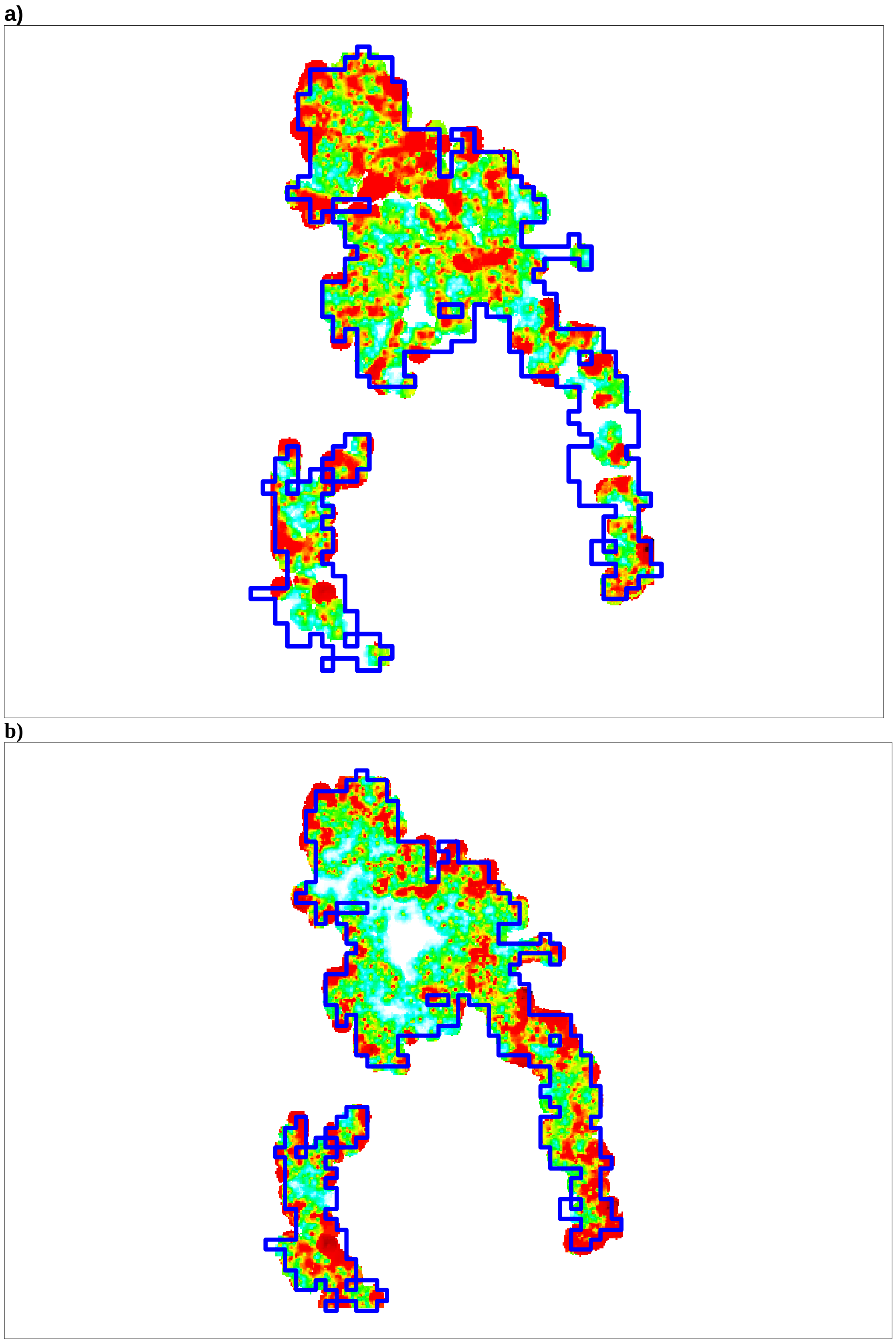
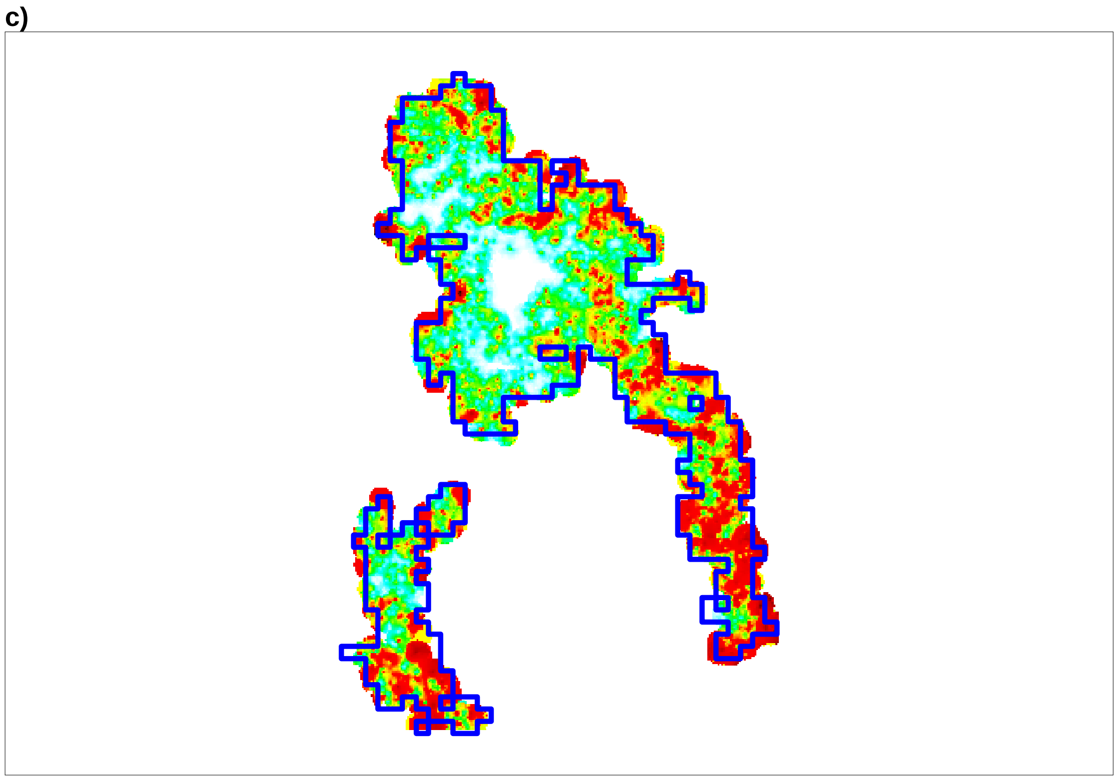


Figure 3.6. Wader conservation priority areas ranked by **a**) counts of all waders and **b**) counts weighted by conservation status listing where Amber species are weighted 3 times more than Green list species (there were no Red list waders present). Species included: Curlew, Dunlin, Golden Plover, Lapwing, Common Sandpiper and Snipe. Red= high priority , White=low priority.

c)



d)

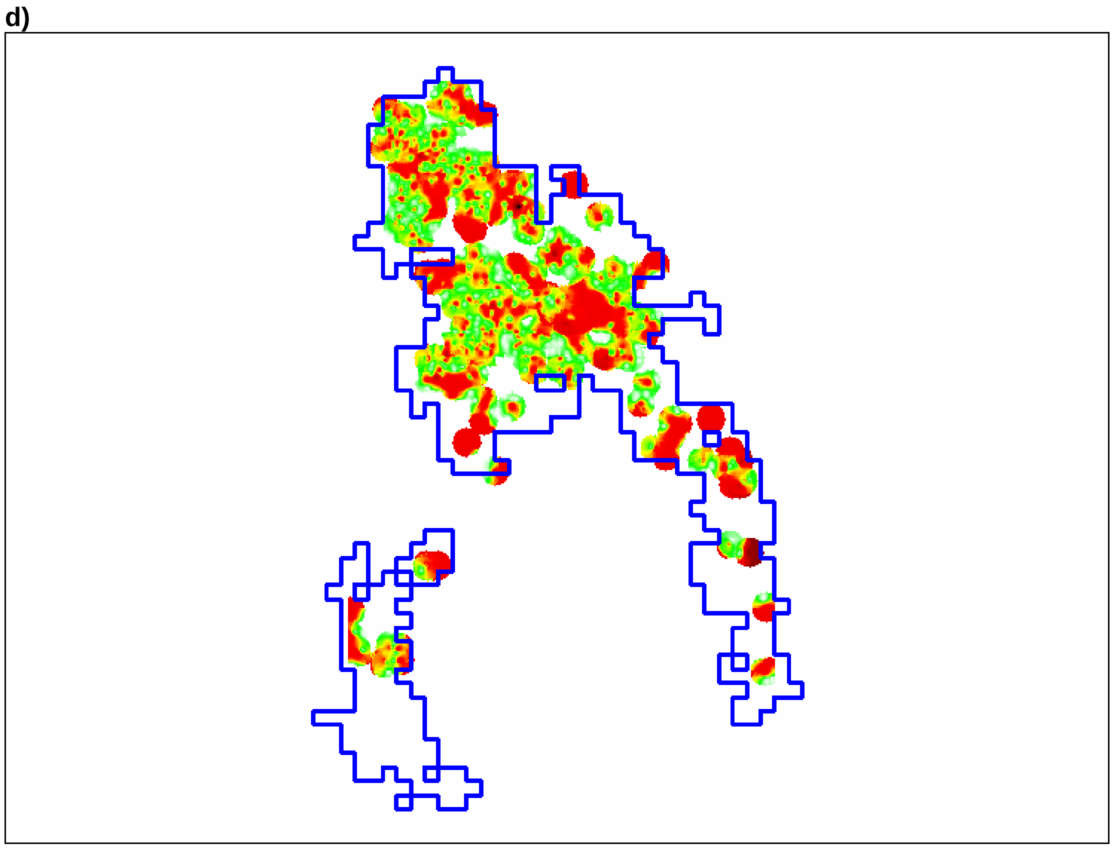
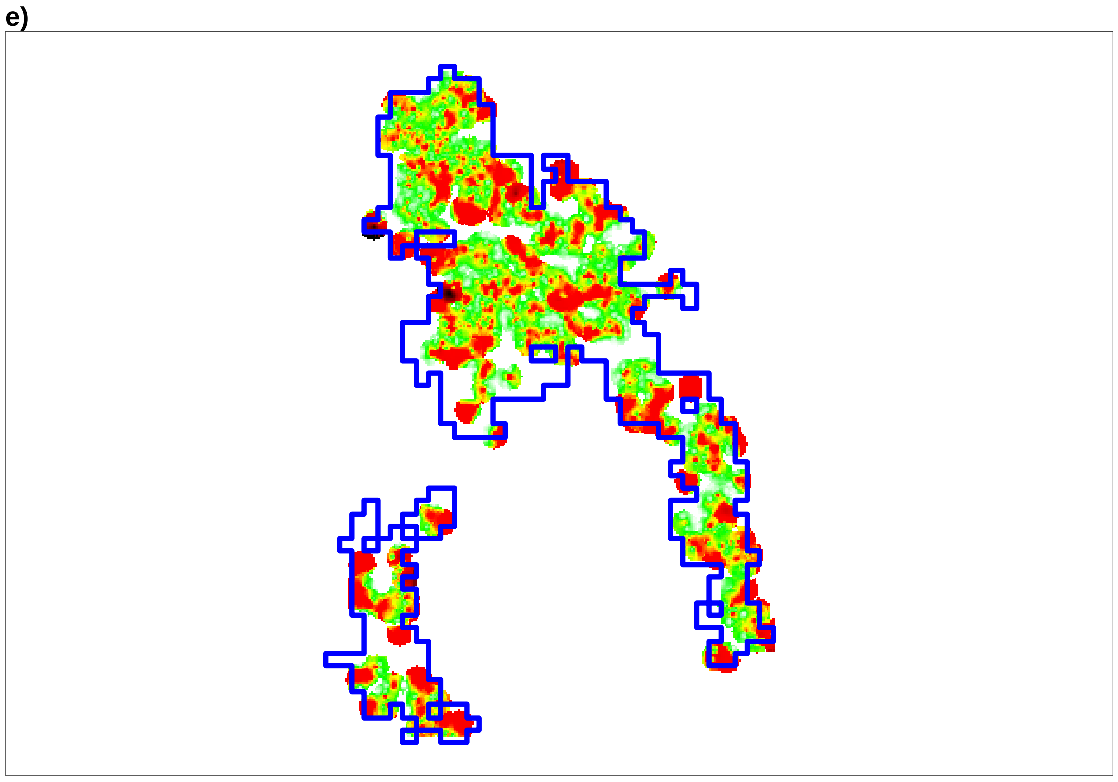


Figure 3.6. Conservation priority areas ranked by **c)** the abundance of Red and Amber listed species and **d)** the abundance of SPA designation species. Red list (Linnet, Reed Bunting, Ring Ouzel, Twite) species weighted by three relative to Amber (Meadow Pipit, Peregrine, Stonechat, Tree Pipit, Short-eared Owl, Curlew, Dunlin, Lapwing and Snipe). SPA designation species are Golden Plover, Peregrine, Merlin, Short-eared Owl and Dunlin, the latter an additional qualifying species under article 4.2 of the directive.

e)



f)

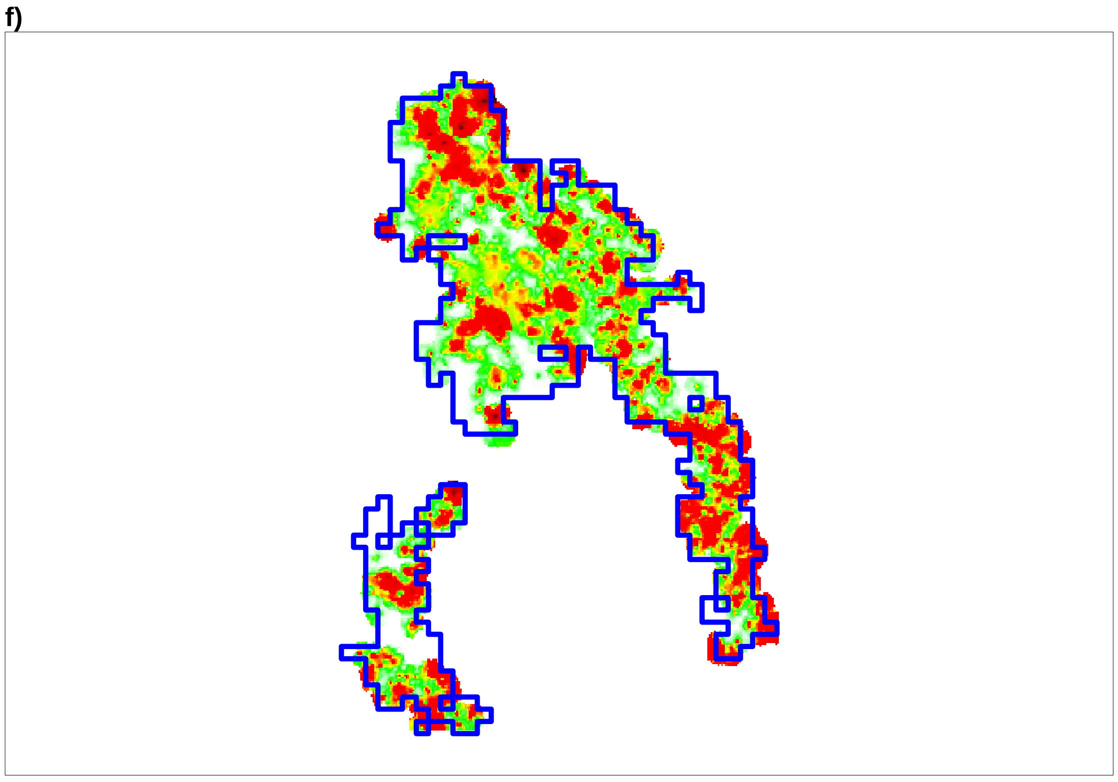


Figure 3.6 Conservation priority areas identified using **e**) total abundances of SPA and Red list species counted and f) relative abundances of SPA and Red list species present (see Fig 3.6c & d for species lists) within the survey area. Relative abundance for each species is determined by dividing each sighting by the population size for that species, thus one sightings of a rare species is given greater weight than for a common species.

4. DISCUSSION

4.1. Critique of approach

Models explaining a significant amount of the variation in the fine-scale associations of moorland birds were produced for all species, with particularly strong models produced for Golden Plover, Reed Bunting, Ring Ouzel, Snipe and Dunlin. Models based upon bird abundance and habitat variation summarised at the larger 1 km scale had a lower predictive power, with a significant correlation between predicted and observed abundance, for only six species. Furthermore, models of change in bird abundance between 1990 and 2004 at the same scale performed poorer still, with only the model for Curlew demonstrating predictive power. Similarly, we failed to detect strong effects of either agri-environment schemes or the intensity of grouse-moor management on changes in the abundance of most species. Our inability to construct good predictive models, may either indicate relatively weak habitat and management effects on birds, a failure to select biologically important predictors, or a failure to detect actual biologically significant effects (Type II errors) due to a lack of sufficient precision or reliability of bird and habitat data. It is important to attempt to distinguish between these two possibilities before discussing the implications of the results.

Unfortunately, few similar studies have tested the efficacy of their models by using data deliberately excluded from model construction as an independent test data set. Among these few studies, predictions of Red Grouse and Golden Plover abundance in south Scotland based upon similar remotely sensed data failed to correlate significantly with observed abundance (Buchanan *et al.* 2005), although this was partially attributed to the relatively small size of the test data set (n=10). Correlations between observed and predicted abundance for moorland bird species in South Scotland using models based on detailed field collected vegetation data performed far better, with r^2 values which exceeded 0.5 for five of eight species modelled (Pearce-Higgins & Grant, 2006). Outwith of moorland habitats, the few studies which attempt to model bird distributions show contrasting abilities to predict distribution and abundance (e.g. McPherson *et al.* 2004, Virkkala *et al.* 2005, Manel *et al.* 1999), which can vary with the habitat measures used (Parra *et al.* 2004, Seoane *et al.* 2004, Laurent e*t al.* 2005), sample size (McPherson *et al.* 2004) and species characteristics (Seone *et al.* 2005).

Three sources of error may have reduced our abilities to detect significant effects, and contributed to any Type II errors; error in the bird survey data, habitat data, or modelling approach.

Error associated with bird survey techniques

The bird survey data was collected using a two-visit Brown & Shepherd survey in both years, 1990 and 2004, by a range of surveyors. As a technique, it was devised to survey large areas of moorland rapidly, to indicate the presence of significant breeding wader populations (Brown & Shepherd 1993). The results of surveys using this method have been used to produce population estimates and to set boundaries to important bird areas, such as SSSIs and SPAs. When used to provide precise abundance estimates for particular sites, or individual 1 km squares, then there appears to be considerable error associated with these counts. For example, estimates of Golden Plover abundance from the same site can vary considerably within a year, depending upon the timing of any survey visits in relation to the main period of Golden Plover hatching (Pearce-Higgins & Yalden 2005). It is possible to reduce much of the intra-annual variation associated with the timing of survey visits in relation to the timing of hatching in Golden Plover, by assessing abundance from the maximum count from 3 visits (Pearce-Higgins & Yalden 2005), an approach also used to estimate Curlew abundance (Grant *et al.* 2000), although this significantly reduces the area of moorland which can be covered by each surveyor. Further, the approach of Brown & Shepherd to only count breeding individuals on the basis of strict behavioural codes, and then to calculate estimated abundance by comparing the locations of birds recorded on the two visits, also introduces an additional source of error (Pearce-Higgins & Yalden 2005). Following Sim *et al.* (2005), we have attempted to reduce this effect by basing our analysis of count data on the maximum number of individuals recorded from the two visits.

It is also worth noting that this same method is used to collect summary counts of Red Grouse, Meadow Pipits and Skylarks at a 1 km level. People counts of Red Grouse can produce a reasonable index of abundance when derived from a protocol with a fairly narrow survey window (Pearce-Higgins & Grant, 2006). However, the Brown and Shepherd estimates are likely to be subject to greater error as a result of the wide survey window for the first visit, as Red Grouse detectability tends to decline during the breeding season. This method is also likely to produce considerable error in relation to the counts of small passerines, which are best surveyed by line-transects (Thirgood *et al.* 1995, Buchanan *et al.* 2006).

A further potential source of error in such extensive surveys is the inevitable use of different fieldworkers to collect data in different areas and years. For some species, this can introduce not insignificant error, with mean percentage error between observers from a method based on total counts of Meadow Pipits and Skylarks recorded along line transects equaling 10 % for both species (Buchanan *et al.* 2006); error associated with estimates from Brown & Shepherd might be much higher (see above).

Overall, an approximate indication of the degree of error which may have been introduced through the use of Brown & Shepherd survey data is obtained by examination of Appendix 2, which presents correlations between the maximum count of species recorded across the Peak District in 2000 (from Sim *et al.* 2005) and 2004 (this data). Given the four-year interval between these two surveys, counts might be expected to be reasonably similar, accepting the potential for some changes in bird abundance in the interim. However, the relatively weak correlations for many species, particularly Snipe, Ring Ouzel etc, indicate that the results of Brown & Shepherd surveys of these species are likely to be highly variable at the 1 km level. Nevertheless, when used to assess change across larger blocks of land, such as the Peak District as a whole, the relative consistency of counts from the 2000 and 2004 data, apart from for Snipe and Lapwing (see introduction), are encouraging.

The Brown & Shepherd (1993) survey technique remains the most cost-effective way of surveying large areas of moorland. However, our results suggest that for some species, such as Snipe, it may give a relatively weak assessment of distribution; if data on the distribution and abundance of such species are important for a particular study, then more intensive survey methods should be used. Consideration should also be given to the appropriate spatial scale at which to analyse such data. For species with large home-ranges, such as breeding waders (cf. Pearce-Higgins & Yalden 2004), analysis of abundance at a 1 km scale may introduce additional error, as the movements of territorial pairs by only a few hundred metres could result in a pair being assigned to a separate square.

It therefore appears that the error associated with the survey techniques used to collect the bird data will have resulted in reduced power to detect habitat associations at the 1 km level, and even more so for the change analysis at this same scale which relies on two counts, doubling the potential error. The efficacy of the fine-scale analysis, which depends more on the accuracy of the mapping of bird locations, is unlikely to have been affected in this way.

Error associated with habitat variables used

A second potential source of error associated with our modelling was error in the habitat variables used. In relation to vegetation, three sources of data were available; The ESA habitat data derived mainly from 1988/89 aerial photos, CS2000, and from a Landsat 7 satellite image. The former was a habitat map in which polygons of similar vegetation types were delineated and classified from air photos with ground truthing in the field. Although separate categories were used in three different parts of the Peak District, these were combined into broad moorland vegetation categories that were roughly equivalent across the whole study area. The CS2000 map was produced for the whole country from satellite data (Fuller *et al.* 2002), and classifies moorland habitats into broad categories. The analytical techniques involved eliminating small habitat patches, and there is known to be considerable error in the classification accuracy for some of the moorland vegetation categories (Fuller *et al.* 2002). Our analysis of a Landsat 7 satellite image (methods) produced a vegetation map that predicted the vegetation height and composition of four vegetation categories of each 30 m pixel in the Peak District, with a known error rate. Given the known problems with CS2000, and the quantifiable error rate of our satellite derived habitat map, we decided to reject the former. The ESA habitat map summarised habitats into a finer number of categories than the satellite image, but had an unknown error rate, and was mapped 16 years prior to the second bird survey. Therefore, we decided to base our decision on whether to use the ESA data or our satellite derived data for analysis, on the predictive ability of models of fine-scale associations of birds produced using both sets of data. These predictions showed that the satellite-derived map produced models with much greater predictive power than those based on the ESA categories, a pattern also found by Seoane *et al.* (2004), where predictions of bird distribution based on satellite derived data were more accurate than from general purpose land-cover maps. As a result, the majority of the analyses is based upon the satellite-derived vegetation data, although we use some of the finer division of vegetation categories from the ESA data to conduct some broad assessments of habitat preferences of a wider range of species (Table 3.1). These are particularly informative for species that closely associate with vegetation types poorly represented by the satellite image, such as Whinchat and Bracken, and for species with a small sample size. The fact that these fine-scale models based on the remotely sensed data had good predictive ability suggests that the error associated with the vegetation data was relatively low. One potential caveat may be in relation to the vegetation height measure, as the fine-scale association with Lapwing and tall vegetation contrasts with findings from other studies (i.e. O’Brien 2002, Milsom *et al.* 2000) and may therefore suggest that some areas predicted to contain tall vegetation may have been of another vegetation type. However, correlations with vegetation height and other moorland bird species tend to fit with previous work (e.g. Pearce-

Higgins & Grant, 2006), suggesting that for most species, the vegetation height variable was measuring what we expect.

Information about non-vegetation variables came from a variety of sources. Topographical data (slope, elevation, distance to stream), and peat cover were derived directly from the relevant maps, and should therefore be accurate for the scale at which the data were summarised. Information concerning the placements of muirburn was derived from aerial photographs by the MFF team, again a direct measure of the distribution of this management type, although some error may have been introduced by the fact that these polygons delineated clusters of muirburns, rather than separate heather burns. However, we have some confidence that our measure of muirburn intensity is providing a useful index of the intensity of grouse-moor management for the analysis of change across moorland management units, as the measure used correlates significantly with gamekeeper density (see methods).

Whilst our measure of recreational disturbance was perhaps an arbitrary one, dependent upon the functions used to rank footpaths according to their use, and the reciprocal decay of disturbance levels with distance, it is perhaps difficult to *a priori* come up with a better alternative. Reassuringly, our measure produced correlations between habitat use and disturbance equivalent to those derived from more detailed work for Golden Plover (Finney *et al.* 2005). It is possible, however, that this measure was associated with some unmeasured features of habitat, suggested by the positive association between disturbance and Wren distribution, and this may have contributed to some of the observed relationships between our disturbance measure and the distribution of some species. There may be some error, therefore, resulting from how well some of our explanatory variables relating to habitat and management describe the elements they purport to measure, although in general, the vegetation measures in particular are perhaps as good as might be expected from a study such as this. These measures should not have contributed to the reduced predictive power of the 1 km models, particularly as the same variables appeared to produce good predictors of habitat use at the fine-scale.

Error associated with modelling technique used

The final potential source of error in the analysis may have been attributed to our modelling technique. In recent years, simple GLM approaches to such analyses have been widely criticised, in favour of model averaging approaches, which make better use of correlated predictor variables, and are less subject to bias resulting from the particular order of variable selection (Rushton *et al.* 2004). By adopting such a model averaging approach this approach, we are therefore making the best use of the latest statistical modelling techniques (e.g. Whittingham et al. 2005) that are designed for developing predictive models using correlated data, and assessing the relative importance of particular predictor variables. Although we initially anticipated using GLMMs, with appropriate covariance structures to take account of the potential problem of spatial autocorrelation, by calculating and incorporating an appropriate autocovariate term for each species and each analysis, we have controlled for this potential problem (Augustin *et al.* 1996, cf. Finney *et al.* 2005).

The most likely source of significant error in the analysis probably originated from the bird survey data, a problem doubled for the change analysis because it is based on two counts, which may explain the particularly low predictive power of the results from that analysis. However, even when the data were aggregated at a larger-scale, as with the analysis of change between moorland management units, which would have partially ameliorated this problem, many habitat effects remained weak. This could however, be because habitat data averaged across such a large-scale was too crude. There may also have been some additional error with respect to some habitat and management variables, which for some measures (i.e. vegetation) could be quantified, but for the management variables, could not. Analyses have therefore tended to show that it has not been possible to easily relate changes in bird populations to particular habitats or managements.

Although these results may be unhelpful in relation to using these analyses to inform the future management of the Peak District moorlands, they are perhaps not surprising. The most robust statistical models of species distribution tend to be produced for species with high abundance but a high degree of environmental specialisation, in other words, species that are found within a narrow range of habitats (Seoane *et al.* 2005). Yet, as population size and ranges change, then this degree of specialisation will also change. For example, in an increasing population, once all the good quality territories are occupied, new recruits to the population will be forced to occupy poorer quality sites, potentially on new habitats. Conversely, in a declining population, individual habitat preferences will cause poorer quality habitats to become unoccupied first. Combined, these effects make it difficult to interpret such analyses of change, as it may be management in the areas of stable populations which has resulted in population change in more peripheral localities. For example, if grouse-moor management is associated with increased breeding success of a particular species, but the population density of that species on a particular estate is at carrying capacity, then the extra birds produced as a result of the favourable management will be forced to colonise less favourable areas. Therefore, any analysis of changes in bird population in relation to management will fail to detect the positive benefits of grouse-moor management, or demonstrate a negative association between population change and management. These problems should be borne in mind when interpreting these results.

Overall, this does not mean that the Brown & Shepherd technique is a poor one for surveying moorland bird populations. In most cases, it remains the most cost effective way to assess broad population changes across large geographical areas, such as within SPAs and SSSIs. Neither, is the analytical approach taken here of limited value. It can be potentially a good way of generating useful hypotheses for future work (e.g. Buchanan *et al.* 2003). However, it is clear that such data, when used to help understand why bird populations have changed, may have limited application, and that perhaps more targeted, specific studies are required to fully understand the processes driving upland bird populations. These may involve more intensive survey work, involving a greater number of visits (e.g. Grant *et al.* 2000), or more detailed studies covering all aspects of the ecology of particular species. Information on species not covered extensively in the analysis (i.e. raptors), and rare species (i.e. Twite) are best obtained from other sources such as Carr & Middleton (2004) for population estimates and distribution maps for the Peak District, or species specific publications for more information about their ecology.

4.2. Species specific accounts

In the following accounts, we have attempted to summarise the results for each species, along with information from other studies, to provide an assessment of the likely drivers of population change, and where possible recommendations for management. After the general context, results summarised are (a) fine-scale associations from both the ESA habitat data and fine-scale analysis of satellite-derived data , (b) larger scale analysis of satellite-derived data and (c) analyses of change , to provide (d) recommendations for appropriate management prescriptions and the areas which should be focussed on .

Curlew

Curlew have a wider UK distribution than both Golden Plover and Dunlin, being less restricted to moorland habitats, although a large proportion of their populations still occur in the uplands (Gibbons *et al.* 1993, Stillman & Brown 1998). The UK breeding population has recently been estimated at 99,500 – 125,000 pairs, 39 % of the European total (O'Brien 2006), within which the moorland Peak District population of at least 453 pairs (Carr & Middleton 2004) represents c. 0.4 % of the UK population. Curlew are currently on the Amber list of conservation concern, although they have declined in most upland regions, apart from the South Pennines (Sim *et al.* 2005). This is reflected in the apparent doubling of the Peak District population from 1990 to 2004/5 (Carr & Middleton 2004, Appendix 1).

a) Fine-scale analyses of the habitats where Curlew were recorded indicate associations with both grass and heather cover, particularly where marshy or wetland habitats occur (Table 3.1, 3.2). Like other wader species, there was also a strong association with flat areas of ground (Table 3.2). Previous analyses of Curlew abundance have recorded similar conflicting correlations with either grass or heather cover (Brown & Stillman 1993, Stillman & Brown 1994), whilst a more detailed analysis of habitat associations in South Scotland found densities unrelated to variation in heather to grass cover, but correlated weakly with rush (*Juncus effusus and J. acutifloris*) and *Scirpus cespitosus* (Pearce-Higgins & Grant, 2006). Radio-tagged birds in the North Pennines did not show any significant selection for particular vegetation types (Robson 1998). Additional fine-scale associations of Curlew sightings with taller vegetation reflect the selection of Curlew on in-bye fields for taller swards (O'Brien 2001). In relation to management Curlew appeared to avoid the most heavily disturbed areas of moorland, in common with most other wader species, but showed no association with heather burns despite the fact they are known to favour burns for nesting (Robson 1999).
b) Large-scale analyses of Curlew abundance reflect the fine-scale results, with densities greatest on flat areas at low altitudes. In relation to vegetation, the fine-scale associations with grass or heather cover are also reflected at this larger-scale, with positive correlations between abundance and both heather and grass cover. An additional association with intermediate vegetation heights (Table 3.3) indicates that although Curlew select tall vegetation, the highest densities were on squares with mean vegetation height approximating to 30 cm, which supports previous analyses highlighting an association with structurally diverse moorland vegetation (Pearce-Higgins & Grant, 2006). In common with the fine-scale analysis, Curlew abundance was negatively correlated with levels of disturbance at the 1 km level, suggesting that visitor pressure may be having a slight effect on breeding density. Curlew was in fact the only species where such an effect was apparent at the 1 km level, and may reflect the greater sensitivity of Curlew to human disturbance, as measured by alarm-calling distances (Yalden & Yalden 1989).

c) The model of change in Curlew abundance was the only model with any apparent predictive power, and suggested that squares with the greatest increase in Curlew abundance were characterised by taller vegetation, no or high heather cover and also high levels of disturbance, although at the scale of land management units, none of these habitat or management effects remained significant. As outlined in section 4.1, it is difficult to determine whether such relationships indicate benefits of particular managements or habitats, or simply reflect increased occupancy of poorer quality territories in an increasing population. It is possible to describe the associations between change and both vegetation height (which appear influenced by a small number of squares), and heather cover, as further support to the fact that this species requires heather, grass and tall vegetation, although such an interpretation may be overly simplistic. There are two alternative explanations for the positive correlation with disturbance. Recent management along footpaths, such as resurfacing work, may have increased the likelihood of people remaining along footpaths (Pearce-Higgins & Yalden 1997), reducing the spatial extent of disturbance and allowing Curlew to occupy apparently more disturbed habitats in 2004, as has happened for Golden Plover (Finney *et al.* 2005). Alternatively, in an increasing Curlew population, individuals may be increasingly forced to occupy less-preferred and more disturbed territories. Given the large distances over which Curlew alarm call in response to human disturbance (Yalden & Yalden 1989), it is possible that disturbance is detrimental to this upland wader, although further research is required to determine how significant such effects are. Given the high level of visitor pressure within the Peak District, and increasing Curlew population, such effects do not currently appear limiting. There were no strong effects of the intensity of grouse-moor management of the change in Curlew abundance, or associations with muirburn management, within which Curlew on heather moorland tend to nest (Robson 1999). This is despite previous analyses highlighting the effects of predation by foxes on Curlew breeding success, and evidence that Curlew densities are higher on grouse-moors than non-grouse-moors (Grant *et al.* 2001, Tharme *et al.* 2001). Our inability to detect an effect of grouse-moor management may be because such effects operate at a larger-scale than 1 km squares or even land-management units. This is discussed in more detail below.

d) The increasing Curlew population within the Peak District is important, particularly when set against the background of widespread declines in other upland areas (Sim *et al.* 2005). Increases in the abundance of Curlew appear greatest away from the cotton grass dominated plateaux, with particular increases around the eastern moors of Howden, Derwent and Hallam (Appendix 1, Fig 3.5). However, given the generally catholic associations of Curlew on moorland (e.g. Pearce-Higgins & Grant, 2006), and limited vegetation change during the same period, such correlations should probably not be interpreted as indicating vegetation change has driven the increase in the Curlew population. Although we were unable to associate population changes with either grazing or grouse-moor management, given the fact that more detailed analyses indicate the vulnerability of Curlew to predation by generalist predators, and their usage of heather burns for nesting (Robson 1999, Grant *et al.* 2001, Tharme *et al.* 2001), it is likely that such managements are beneficial to this species. Given the general reduction in crow numbers in the Peak District from 1990 – 2004 (Carr & Middleton 2004) and increase in the intensity of heather burning (A. Yallop, unpubl.), the current moorland management regimes may well have increased the quality of the Peak District moors for Curlew, although our analysis provides no strong evidence that this is the case. More detailed research is needed to examine the impacts of grouse-moor management in particular upon Curlew breeding success, to test the hypothesis whether such management is indeed beneficial.

Dunlin

Like Golden Plover, Dunlin are another typical wader of the UK uplands on the southerly edge of their range, but occur at much lower densities throughout the English Pennines than Golden Plovers (Gibbons *et al.* 1993). The latest UK breeding population estimate for this species is 18,300 – 33,500 pairs (O'Brien 2006), of which the Peak District population of 67 pairs (Carr & Middleton 2004) represents 0.2-0.3 %. Although currently Amber listed, as a result of important wintering populations in the UK (Gregory *et al.* 2002), Dunlin populations appear to have declined in many mainland upland regions (Sim *et al.* 2005), a pattern reflected in the Peak District (Carr & Middleton 2004, Appendix 1).

a) In the Peak District, Dunlin tend to associate with cotton grass and eroding moorland (Table 3.1, 3.2), and therefore show similar associations to those of Golden Plover, although areas classed as bare peat are avoided (Table 3.1). Although there is little information on Dunlin diet from the UK uplands, studies elsewhere suggest that Diptera larvae such as tipulids, are favoured (Buchanan *et al.* in press), and it is therefore likely that the reasons for this association are similar to those described for Golden Plover, above.

b) At the 1 km level, the strongest determinant of Dunlin habitat remained cotton grass cover, with additional negative correlations between abundance and mean slope and distance to stream (Table 3.3). Thus, the previous associations of Dunlin with high altitude plateaux within the South Pennines of Haworth & Thompson (1990) and Stillman & Brown (1994) do not appear to have changed, despite the declining population size.

c) Although there were a number of significant correlates of change at the 1 km level, the strongest two, which are arguably the most biologically meaningful, were a negative correlation with slope, and positive correlation with vegetation height (Table 3.4). Dunlin populations were most likely to decline on steep slopes with short vegetation, although there was no evidence of an association with vegetation height in either the fine-scale or 1 km resolution analyses of habitat associations (Table 3.2, 3.3). The apparent positive effects of the proximity of in-bye on Dunlin populations appears heavily influenced by only two squares (Fig. 3.3), and therefore must be regarded with caution. The effects of grazing and grouse-moor management regimes on the change in Dunlin abundance show a similar pattern to that for Golden Plover, albeit with stronger negative effects of grouse-moor management, particularly on estates with the highest grazing regimes (Fig 3.4).

d) This analysis tentatively suggests that intensive management, both through high levels of grazing and grouse-moor management, may be detrimental to Dunlin. Such effects could occur through reductions in vegetation height; areas with taller vegetation appear to have had more stable populations (Fig. 3.3), or changes to the hydrology (Dunlin favour the wettest areas of blanket bog; Lavers *et al.* 1996) and invertebrate populations. However, there has been little detailed research on the habitat requirements, diet and population dynamics of

Dunlin in the UK uplands, and such research should be a high priority if population declines in this species are to be reversed.

Golden Plover

This wader is typical of the UK uplands, where it is on the southerly edge of its global range, but locally found at very high density (Ratcliffe 1976, Gibbons *et al.* 1993). The latest UK population estimate is of 38,400 – 59,400 breeding pairs (Thorup 2006), with a minimum estimated 420 pairs within the Peak District (c. 1 % of the national total; Carr & Middleton 2004). Although currently not regarded as being of conservation concern, populations have declined in some regions (Gregory *et al.* 2002, Sim *et al.* in press). Golden Plover are listed under Annex 1 of the EU Wild Bird directive, and are one of the qualifying species for the South Pennines SPA, within which the Peak District is located.

a) The fine-scale distribution of Golden Plovers shows associations with cotton grass moorland from both the ESA and satellite derived habitat variables. The strongest associations were with eroding moorland, whilst non-heather dominated dry bog and bare peat habitats were also favoured (Table 3.1). Analysis based upon the satellite data, which incorporated topographical and management variables, as well as vegetation height, also highlighted the fact that Golden Plover were located in short vegetation, at intermediate altitudes on flat ground (Table 3.2, Fig 3.1). These results support the previously known associations of this species for flat, high altitude plateaux (Haworth & Thompson 1990, Stillman & Brown 1994, Pearce-Higgins & Grant, 2006). In relation to vegetation, associations with cotton grass are widely known from the literature, although in some regions, particularly of Scotland, plovers tend to associate with heather cover (Parr 1980, Haworth & Thompson 1990, Brown & Stillman 1993, Stillman & Brown 1994, Whittingham *et al.* 2000). Cotton grass dominated blacket bog is the preferred nesting habitat, and provides an important foraging area for young chicks, particularly associated with the emergence of adult tipulids in late May and June (Pearce-Higgins & Yalden, 2004). The associations with bare ground, eroding moorland and non-heather dominated dry bog can also be explained by reference to our knowledge of chick ecology. Areas of bare peat form an important foraging area for chicks, providing easy access to the tipulid larvae which dominate their diet, whilst older chicks increasingly utilise areas of bilberry and crowberry heath for caterpillars and some beetles (Pearce-Higgins & Yalden, 2004). The fine-scale association with short vegetation reflects that of foraging birds on pasture (Pearce-Higgins & Yalden 2003a). Like other plovers, Golden Plovers forage by sight and feed on surface or subsurface prey in the top of the soil, and therefore require short, open vegetation to forage effectively, a feature which is positively correlated with Golden Plover abundance on moorland in South Scotland (Pearce-Higgins & Grant, 2006). In relation to management, the fine-scale avoidance of areas of high disturbance by Golden Plovers has been highlighted previously, and appears to reflect a true avoidance of areas with high visitor pressure (Finney *et al.* 2005).
b) Associations at the 1 km scale reflect the previously discussed topographical effects in relation to altitude and slope, and the strong positive association with cotton grass cover (Table 3.3, Fig 3.2). Within the Peak District, the main determinants of Golden Plover abundance at this scale therefore appear to be for

flat, intermediate altitude plateaux of cotton grass cover, as previously determined by Yalden (1974) and Stillman & Brown (1994).

c) Cotton grass cover was the only significant correlate of change, a result apparent both from analysis at the 1 km scale, and aggregating data across individual management units (Table 3.4; Results 3.5). Populations on areas of high cotton grass cover tended to have increased, whilst declines were apparent in other habitats. As a result, it is the northern and western moors of Saddleworth, Bleaklow and Snake Summit where numbers have increased (see also Pearce-Higgins & Yalden 1997). Given the strong association of Golden Plovers with cotton grass, this appears to represent a contraction to the most favoured habitats, and may reflect an improvement of the condition of these habitats relative to others within the Peak District. However, without further research, it is difficult to determine the underlying mechanism. Given the importance of tipulids in the diet of Golden Plovers (Pearce-Higgins & Yalden 2003a, 2004), and association of tipulids with cotton grass blanket bog habitats (Pearce-Higgins & Yalden 2004), it is possible that such changes are related to invertebrate prey. There was a weak interaction between the intensity of grouse-moor management and grazing level in the larger-scale analysis of land management units, suggestive of a significantly more positive effect of grouse-moor management on land under moorland enhancement prescriptions, but negative effect on other ESA Tiers. Given the association between Golden Plovers and grouse-moors in parts of the uplands (Tharme *et al.* 2001) and the relatively high nesting success of Golden Plovers on some grouse-moors (Pearce-Higgins & Yalden 2003b), and requirements of Golden Plovers for short vegetation, favoured by grazing (Pearce-Higgins & Grant, 2006), these contrasting correlations are difficult to interpret. However, for each tier, the slope of the relationship between the change in Golden Plover abundance and grouse-moor management does not differ significantly from zero, suggesting that the correlations may be of little biological significance.

d) Although the analyses detect strong and consistent positive effects of cotton grass vegetation, which account for the high and increasing population densities on many of the high plateaux areas of the Peak District, these results do not point strongly to particular management regimes which appear to favour this species. Previous analyses highlight the potential benefits associated with relatively high grazing pressure in maintaining appropriate sward structures for Golden Plover (Pearce-Higgins & Grant, 2006), and the role of grouse-moor management to again promote open vegetation structures through heather burning, and reduce the abundance of generalist predators (Tharme *et al.* 2001, Pearce-Higgins & Yalden 2003b). Given the importance of blanket bog vegetation for this species, it would be valuable to examine the effects of burning management on invertebrate populations in such areas, particularly focussing on tipulids. The current stability of the Peak District population (Carr & Middleton 2004, cf. Appendix 1) indicates that the management regimes currently in place on those areas of increasing population should be maintained and supported.

Lapwing

Lapwing populations have been declining widely across the UK (Gregory *et al.* 2002), and these trends are widely reflected in the uplands (Sim *et al.* 2005). Reasons for these declines are many, but generally appear to reflect poor productivity of many poulations

(Peach *et al.* 1994). However, numbers appear to have increased in the Peak District from 1990 – 2004 (Carr & Middleton 2004), although this contrasts with apparently stable populations in the South Pennines as a whole (Sim *et al.* 2005). The current population estimate within the Peak District moorlands of 131 pairs, represents less than 0.1% of the national total of 137,000 – 174,000 breeding pairs, although of course does not include birds breeding on in-bye fields.

a) Lapwing appear to be most abundant at low altitude, and associated with taller vegetation at the fine-scale (Table 3.2). In terms of vegetation type, the majority of sightings were recorded from dry or marshy grassland habitats (Table 3.1). Although there has been little work examining the habitat requirements of Lapwing on moorland, because it is predominantly a bird of grassland habitats (Gibbons *et al.* 1993, Wilson *et al.* 2001), the association with vegetation height contrasts with the weight of work conducted on Lapwing habitat associations on grassland, and where birds are found on fields with short swards (O'Brien 2002), for the reasons outlined for Golden Plover, above. As with Snipe, our correlation between Lapwing and vegetation height may indicate an association for areas with high rush cover, indicative of wet and nutrient rich areas which are likely to support high densities of invertebrate food. Alternatively, because of the relatively heterogeneous nature of moorland vegetation, squares with relatively tall vegetation may provide invertebrate food and cover for Lapwing, but also contain sufficient areas of short vegetation for the birds to nest, and chicks to move about. This relationship could also result from the misclassification of suitable Lapwing habitat by the satellite image. As with other wader species, Lapwing appear to avoid areas of high disturbance.
b) Although models at the 1 km level had no predictive power, they contain some potentially strong effects (Fig. 3.2, 3.3). Thus, 1 km squares with the greatest Lapwing abundance tended to be flat, and at low altitude, with a weak association for reduced heather cover. This analysis supports the fine-scale associations of Lapwing for low altitude grassland squares, and highlights that Lapwing only really occur on moorland habitats which are similar to the pastoral grassland habitats with which they more commonly associate.
c) Squares with increasing counts tended to be at higher elevation, but again, with little heather cover. Thus, there is some weak evidence that Lapwing distributions have moved up the hill. As with Curlew, these effects may represent the colonisation of previously unused habitats within an increasing population. Despite the strong effects of grazing on Lapwing abundance on fields, and positive effects of grouse-moor management on abundance (O'Brien 2002, Tharme *et al.* 2001), there was no evidence that either moorland agri-environment schemes or grouse-moor management, were related to the spatial variation in Lapwing population trends.
d) Lapwing populations on moorland areas of the Peak District may be marginal, and reflect 'over-spill' from adjacent areas of high population densities breeding on unsurveyed in-bye fields. Interestingly, there appears to be a relatively close association between the distributions of Snipe and Lapwing in 2004 (Fig 3.5), and it may be that similar management for Snipe to maintain wetland areas, particularly if such areas have areas of relatively short sward, will benefit Lapwing as well. However, it is likely that it will be the management of adjacent in-bye fields which will be the key to sustaining Lapwing populations in the Peak District, which given their current widespread declines across the UK, will be an important conservation priority.

Meadow Pipit

Meadow Pipits are widespread in the uplands, where they tend to be the most common bird species. Meadow Pipits are currently Amber listed in birds of conservation concern as a result of declining populations across the UK, although recent analysis suggests that there have been few declines in the uplands (Sim *et al.* 2005). However, numbers appear to have reduced slightly in the Peak District (Carr & Middleton 2004). The locations of individual pipit sightings were not mapped, and so analyses on this species are confined to the 1 km level.

b) Analysis at the 1 km scale shows that Meadow Pipits are relatively widespread, but weakly associate with cotton grass areas on peat, and tend to be at reduced density on heather areas, particularly those subject to burning (Table 3.2). There is a widespread association between Meadow Pipits and intermediate levels of heather cover (Redpath & Thirgood 1997, Smith *et al.* 2001, Vanhinsberg & Chamberlain 2001, Pearce-Higgins & Grant, 2002, in press), although the mechanism underlying this relationship is unclear. Whether the difference between negative relationship between Meadow Pipit abundance and heather cover highlighted by our analysis, and quadratic relationships found in the other studies, is significant remains unclear. However, these results suggest that in the Peak District, pipit densities are greatest on cotton grass, potentially linked to the availability of invertebrate prey such as tipulids, and but at lowest density on moorland with the greatest heather cover. Related to this, densities also appeared to be reduced on squares subject to a high intensity of heather burning, which has previously been noted by Tharme *et al.* (2001). It is however worth noting that counts of Meadow Pipits tend to show a high degree of variability (Appendix 2, Buchanan *et al.* 2006), which potentially accounts for relatively weak habitat effects in analysis such as these, and the poor predictive power of this model (Table 3.5).

c) The only correlate of the change in Meadow Pipit abundance was a negative correlation with heather cover, thus squares with the greatest heather cover were those most likely to have a reduction in pipit abundance. This supports the findings of the previous spatial analyses, and suggests that management to promote heather cover as a way of reducing the conflict between Hen Harrier conservation and grouse-moor management (Redpath & Thirgood 1997) may reduce Meadow Pipit abundance.

d) Despite being Amber listed, Meadow Pipit populations in the uplands are of little direct conservation importance. Although they have received some attention because they are a keystone prey species for moorland raptor species such as Merlin and Hen Harrier, results from this analysis tend to support previous studies. Thus, management to manipulate heather cover will have some impact upon pipit densities, although whether the ensuing changes in pipit abundance will then impact upon raptor populations, remains to be tested (Redpath & Thirgood 1997).

Red Grouse

This is a subspecies of the Willow Grouse endemic to Britain and Ireland, which has an extensive distribution across upland areas of heather moorland (Gibbons *et al.* 1993). It is currently given an Amber conservation status within the UK (Gregory *et al.* 2002), but Red listed in Ireland (Newton *et al.* 1999), although there is currently no accurate assessment of overall population size. Many populations are managed to artificially high densities for shooting interest, although there is some evidence of declines in many areas (e.g. Hudson 1992), particularly as a result of heather loss through high grazing levels (Thirgood *et al.* 2000, Pearce-Higgins & Grant, 2006). Because of the cyclic nature of many Red Grouse populations, it is difficult to accurately assess population change from two annual surveys, so that the apparent doubling of the population within the Peak District Carr & Middleton (2004) may simply reflect different stages in this cycle. The detailed location of individual Red Grouse sightings were not mapped, and so analyses on this species are confined to the 1 km level.

b) The abundance of Red Grouse was strongly related to topography, with the highest counts from squares with sloping ground and few streams. Counts were strongly negatively correlated with grass cover; reflecting the well established association of Red Grouse with heather (i.e. Brown & Stillman 1993, Stillman & Brown 1994, Pearce-Higgins & Grant, 2006), and were lower in squares surrounded by in-bye grassland, which also contained less heather (Table 2.6). Finally, Red Grouse abundance was negatively correlated with the recreational disturbance score, which may again partly reflect the association with heather (Table 2.6), which is of reduced cover in areas of high disturbance, but may potentially be a real effect of disturbance.

c) Although there were no 'significant' correlates of change, there was some evidence that squares where Red Grouse numbers had increased tended to be at intermediate altitudes with steeper slopes, have shorter vegetation and higher cotton grass and to a lesser extent, greater heather cover, than squares where Red Grouse had declined (Fig. 3.2). Given the 1 km resolution of the count data, it was not possible to analyse the direct effects of agri-environment schemes or grouse-moor management upon Red Grouse populations, although it is likely that such management promotes abundance (Hudson 1992, Tharme *et al.* 2002).

d) The majority of Red Grouse were found across the central areas of the Dark Peak, from Park Hall, in the west, to Derwent and Howden Moors in the East (Carr & Middleton 2004, Shepherd 2005). Further north, isolated areas of high density were recorded, whist densities in the South-West Peak and Eastern Moors were generally low (Carr & Middleton 2004, Shepherd 2005). It is these central areas of cotton grass and heather where populations appear to have increased the most, and it is likely that in the long-term, combinations of appropriate grazing, to maintain dwarf shrub cover, and predator control on these areas, will be required if these large populations are to be maintained (Tharme *et al.* 2001, Pearce-Higgins & Grant 2005). Management for Red Grouse on more peripheral areas should focus on restoring heather cover, where limiting, and predator control, based on previous work (Hudson 1992). The negative correlation between abundance and disturbance levels does not necessarily mean that Red Grouse are detrimentally affected by disturbance, due to the inter-correlations between predictor variables, but interested parties may wish to consider conducting more detailed research on the impacts of high visitor pressure on Red Grouse, as has happened for other moorland species (cf. Finney *et al.* 2005).

Reed Bunting

Reed Buntings are Red listed in Birds of Conservation Concern as a result of widespread declines in lowland farmland habitats, with a 62 % decline on CBC plots (Gregory *et al.* 2002). There is some evidence that this decline has been driven by reduced survival rates (Peach *et al.* 1999). In contrast, populations appear to have increased significantly on the Peak District moorlands by almost 800% (Carr &

Middleton 2004), but are unlikely to be important from a national context. This is the first attempt to examine the habitat associations of this species in upland habitats.

a) Fine-scale analysis of Reed Bunting data, in common with data for many other passerines, indicate associations with Bracken, marshy grassland and wet bog habitats (Table 3.1). Analysis with respect to the satellite-derived vegetation data support these findings, with a strong effect of vegetation height, flat ground and a mosaic of moorland vegetation types. Together, these relationships suggest that Reed Buntings are species of tall, wet vegetation, not explicitly measured by the satellite image, reflecting preferences from lowland habitats (Gibbons *et al.* 1993).
b) Analysis at the 1 km scale supports these findings, with the areas of greatest abundance being low altitude, flat ground covered in grass (Table 3.3). Such a broad association will tend to reflect the locations of wet flushes and marshy vegetation highlighted by the analysis of the ESA data (Table 3.1).
c) Analysis of change in Reed Bunting populations highlighted a couple of negative correlations with non-heather heath and cotton grass cover; habitats that contrast with the preferred low-altitude, grassy habitats (Table 2.6). There were no strong management related drivers of change, and it is likely that the spread of Reed Buntings into the uplands, has been facilitated by the recent trend for warmer weather, but may also represent a reduction in the quality of the previously occupied habitats.
d) It is clear from the distribution maps, and results of the fine-scale modelling, that it is the South-Eastern Moors where the greatest increase in the abundance of this species has occurred, reflecting the importance of these areas for a range of passerine species, as indicated by the hotspot maps. Within these areas, management should aim to maintain the important marshy and wetland habitats which will also be utilized by other species of interest.

Ring Ouzel

Ring Ouzels are one of the most totemic upland species, restricted to upland and montane areas with steep slopes (Gibbons *et al.* 1993). It is, however, one of the UK's fastest declining bird species, having undergone an estimated 60 % decline between the 1989/91 and 2000, and is Red listed as a result (Wotton *et al.* 2001, Gregory *et al.* 2002). Reflecting the national trend, within the Peak District the population has declined by an estimated 15 % from 1990 – 2004 (Carr & Middleton 2004), although this is less than the 64 % decline across the South Pennines recorded by Sim *et al.* (2005).

a) Fine-scale analysis of Ring Ouzel distribution indicate associations with bracken, dwarf shrub and dry grassland habitats (Table 3.1). Ring Ouzels nest in heather, and bracken, but forage in grass, and tend to be associated with moorland comprising mosaics of the two vegetation types on steep slopes (Burfield 2000, Buchanan *et al.* 2003). Fine-scale associations from the satellite-derived vegetation data indicate associations with areas of mixed cotton grass and non-heather dwarf shrubs on steep slopes, with slope being the strongest effect. As with Stonechat, it is possible that such associations with cotton grass in particular, may reflect a miss-classification of vegetation types on steep slopes.
b) Analysis at the 1 km scale also highlight the strong association with steep slopes. At this resolution, a positive correlation with the amount of heather burning reflects an association with heather at this scale due to the strong

correlation between the two (Table 2.6). There was also a quadratic correlation between abundance and the level of disturbance, which combined with an almost significant positive association between Ring Ouzel fine-scale habitat selection and disturbance (Table 3.3), suggests that Ring Ouzel habitats tend to be subject to heavy disturbance, but provides little evidence that this is detrimental.

c) Analysis of population change indicate that areas subject to heather burning, and therefore the most heather dominated areas, were those most likely to maintain their Ring Ouzel populations (Table 3.4). Ring Ouzels were also most likely to decline in high altitude squares, and appear to have suffered a distributional shift down the hill within the Peak District. Previous equivalent analysis of correlates of Ring Ouzel decline in Scotland highlighted that losses were greatest from shallower slopes, low and very high altitudes, close to coniferous plantations and on areas of heather:grass mosaics, although only the topographical effects were particularly strong (Buchanan *et al.* 2003). The contrasting relationships between change and altitude from the Scottish data and the Peak District is difficult to account for, but may relate to different variations in management with altitude between the two areas. In addition to these effects, there were no significant associations between changes in Ring Ouzel populations and either grazing levels or grouse-moor management. Taken together, these relationships suggest that it is difficult to specifically link changes in Ring Ouzel populations to habitat and management, apart from some suggestion that areas subject to heather burning, and therefore with greater heather cover, may be most likely to retain their populations.

d) Examination of the spatial variation in distribution and population trends show that Ring Ouzels appear to have been largely lost from the western moors, but appear to have increased on some eastern sites (Fig. 3.5, Appendix 1). It is difficult to propose the appropriate management which should be undertaken in these areas to maintain such populations. Breeding birds require areas of heather on steep slopes for nesting, and short grass rich in earthworms for foraging (Burfield 2002), and maintaining adequate areas of both within the moorland landscape is likely to be beneficial, although it does not appear that the losses in the west can be strongly linked to either losses of such heather of grass areas. Given the widespread declines of Ring Ouzels across the UK, it may be that larger-scale processes are responsible, such as climate change, with recently published research linking declines in territory occupancy to the recent trend for warmer British summers (Beale *et al.* 2006).

Skylark

Skylarks have widespread distribution throughout the UK, with about 10 % of the population within upland habitats (Brown *et al.* 2000). The population has undergone a significant decline in the last 20 years or so, largely as a result of agricultural intensification (e.g. Wilson *et al.* 1997) affecting both breeding performance and over-winter survival. As a result it is on the Red list of birds of conservation concern. Populations in upland areas have also undergone a moderate decline (Sim *et al.* 2005), reflected in the trends from the Peak District (Carr & Middleton 2004). The location of individual Skylark sightings were not mapped, and so analyses on this species are confined to the 1 km level.

b) Analysis of the Peak District data at the 1 km resolution (finer-scale data were not collected for Skylark) has shown that the birds occur in the highest numbers on low altitude squares in close proximity to in-bye fields, with little heather cover. Additionally, there was a weak positive association with cotton grass cover (Table 3.2). These relationships reflect previous associations with low altitude, rough grassland habitats in the uplands (Brown & Stillman 1993, Chamberlain 2001, Pearce-Higgins & Grant 2002, in press), and that the long-term conversion of heather to grass, as a result of high grazing levels in many upland areas (Fuller & Gough 1999, Pearce-Higgins & Grant 2002), is likely to have improved habitat conditions for Skylark.

c) Analysis of change in Skylark abundance shows that squares in close proximity to woodland, low levels of disturbance and low muirburn intensity were those where numbers were most likely to increase, although the model has little predictive power (Table 3.3). The negative association with muirburn suggests that populations have been most sustained on non-heather habitats, as might be expected, although the positive correlation between abundance and woodland cover is difficult to explain, whilst that with disturbance appears strongly influenced by a small number of squares and potentially spurious.

d) Although Red listed, the majority of the conservation efforts for Skylark have been in the lowlands, as it is declines on lowland farmland that has driven much of this trend. Given that recent trends in upland management have probably improved habitat conditions for upland Skylarks, notably an increase in grass cover as a result of sustained high grazing levels (Fuller & Gough 1999, Pearce-Higgins & Grant 2002), it is possible that declines in upland Skylark populations are due to reduced over-winter survival rates on lowland farmland (Robinson 2001). Therefore, as with Wren, it is unlikely that management on Peak District moorland will be specifically targeted at Skylark populations at present.

Snipe

Snipe is a wader species of wetland habitats, which has a significant proportion of its UK population in the uplands (Gibbons *et al.* 1993, Stillman & Brown 1998). The estimate of the moorland Peak District population of 135 breeding pairs represents only 0.2-0.3 % of the current UK population estimate of 44,600 – 61,000 pairs (O'Brien 2006). Although currently Amber listed in Birds of Conservation Concern, there is recent evidence that populations, particularly in upland areas, are stable, if not increasing in some regions (Sim *et al.* 2005).

a) The fine-scale analyses highlight an association of this species in the Peak District with flat, grassy areas of moorland, particularly of marshy grassland or wet bog habitats (Table 3.1, 3.2). These reflect the strong requirement of this species for wetland habitats on both moorland (Stillman & Brown 1994, Pearce-Higgins & Grant, 2006), and pastoral land (O'Brien 2001, Green *et al.* 1990), habitats, linked to the availability of earthworms, the main prey of Snipe (Buchanan *et al.* in press). The positive association between the location of Snipe sightings and vegetation height may indicate that such areas tend to have taller vegetation, such as rushes, which were not measured directly from the satellite image, although could also indicate a true association for taller vegetation, as recorded on in-bye fields (O'Brien 2001). Additionally, Snipe appeared to avoid the most heavily disturbed areas of habitat.

b) At the 1 km scale, the negative correlation between abundance and slope, and positive correlation with grass cover reflect finer scale analyses, and also previous equivalent analyses (Haworth & Thompson 1990, Brown & Stillman

1993). Further, the negative association with peat cover is probably related to earthworm abundance (see above), which tend to only be found on mineral soils.

c) The model of population change highlighted only a slight decrease of abundance in areas of non-heather dwarf shrubs, which is difficult to account for directly, apart from such vegetation being strongly associated with peat, and tend to be short (Table 2.6). However, this model failed to have strong predictive power, which was perhaps unsurprising given the difficulties associated with surveying Snipe (cf. Appendix 1). Similarly, there were no strong correlations of change when the effects of grouse-moor management and grazing intensity were analysed at the scale of land-management units.
d) In common with other studies, this work re-enforces the importance of wetland areas for Snipe, particularly away from peat-dominated substrates. This probably accounts for the relatively small population within the Peak District, with the majority of flat areas (with which such waders associate) being blanket bog vegetation with few earthworms, and therefore few Snipe. The main hotspots of high Snipe density tend to be at lower altitudes in areas close to in-bye fields, although there appears to have been a northward and westward expansion of the Snipe range within the Peak District between the two survey periods that includes some more upland areas (Fig 3.5). Given the apparently weak impacts of grazing and grouse-moor management on Snipe populations (Tharme *et al.* 2002, O'Brien *et al.* 2005, Pearce-Higgins & Grant, 2006), it appears that maintaining the integrity of the wetland areas associated with Snipe hotspots should be the greatest conservation priority for this species. The precise locations of these areas can be determined from Fig. 3.5. However, as Snipe are increasing within the Peak District (Carr & Middleton 2004), as well as in many other upland areas (Sim *et al.* 2005), and across the UK, it is likely that population changes are currently being driven by large-scale processes, such as the recent run of mild winters.

Stonechat

This species is currently colonising the uplands, where the population has undergone a massive increase, due in all probability, to the current sequence of mild winters (Sim *et al.* 2005). In 1990, only one Stonechat was recorded within the Peak District, whilst in 2004, 83 breeding pairs were recorded (Carr & Middleton 2004). It is currently listed on the Amber list of conservation concern, due to an unfavourable conservation status in Europe (Gregory *et al.* 2002). Previous analysis has highlighted associations with tall vegetation, particularly heather (Pearce-Higgins & Grant, 2006), although given the strong climatic drivers on the population, habitat has, at least historically, not been limiting.

a) At the fine-scale, Stonechat records were associated with bracken, heather, grassland, marshy grassland and scrub habitats (Table 3.1), these each tending to be habitats with taller or denser vegetation, found previously to influence habitat selection on moorland (Pearce-Higgins & Grant in press). Fine-scale analysis based upon the satellite-derived data highlighted strong associations with lower altitude sites with steep slopes, and surprising associations with cotton grass and non-heather dwarf shrub cover.
b) At the 1 km level, the correlation with non-heather dwarf shrub vegetation remained, with abundance highest in squares with 20 – 30 % cover of this vegetation type, whilst there was a negative correlation between abundance and cotton grass cover, reflecting the avoidance of peat dominated soils. This model,

however, had little predictive power, and it is possible that as Bilberry and Crowberry were associated with peat, that these associations may describe an alternative vegetation type which occurs on such moorland edge habitats, such as scrub or Bracken cover.

c) Given the lack of Stonechats recorded in 1990, it was not possible to conduct a separate analysis of change for this species, but instead, the model of associations presented in (b) indicates that it is areas of non-heather dwarf shrub vegetation on mineral soils that have been colonised by Stonechats.

d) There is some similarity between the distribution of Whinchat and Stonechat, with the highest Stonechat densities occurring in the eastern moors, and South West Peak. Given the large and geographically widespread increase in Stonechat populations in the UK, which are probably due to the recent trend for mild winters, it appears that management for this species on moorland is of low conservation priority. If the availability of the species habitat, particularly heather on steep slopes (Pearce-Higgins & Grant, 2006), and bracken and scrub (Table 3.1), were limiting, then the simple exclosure of suitable areas from grazing to promote such vegetation would be beneficial.

Wheatear

Wheatear is a widespread species of upland Britain, that has declined from many previously occupied lowland habitats (Gibbons *et al.* 1993). Although currently Green listed within the UK, recent data suggest that they have undergone population declines in some upland regions, including the Peak District (Carr & Middleton 2004, Sim *et al.* 2005).

a) Analysis as the fine-scale highlighted a range of habitat associations for cliffs, eroding moor, wet bog and dry grassland (Table 3.1), whilst analysis based on the satellite image indicated associations with grass and non-heather dwarf shrub habitats (Table 3.2), although the fine-scale model was one of the weaker predictive models (Table 3.5). Brown & Stillman (1993) demonstrate a strong positive correlation between Wheatear numbers and grass/bracken cover in the eastern Highlands, whilst Wheatear numbers were positively correlated with grass moor in the South Pennines (Stillman & Brown 1994).

b) At the 1 km level, none of the habitat terms were significant, although there was a weak positive correlation between abundance and grass cover that may reflect the finer-scale associations. Previous analysis of the associations between Wheatear and habitat also suggest relatively weak effects of both vegetation composition and structure on the abundance of this species (Pearce-Higgins & Grant in press). This may reflect the fact that fine-scale nest site availability, related to rabbit burrows, scree or stone walls, may be a major determinant of distribution that is impossible to measure across large areas.

c) In relation to population change, declines were most apparent from squares with steep slopes, and high heather cover, potentially related to the association with grass cover just described. Overall, the analysis suggests that the population has perhaps contracted to the more favoured grass dominated habitats, although the pattern of strong declines on steep slopes is more difficult to account for. There were no significant effects of management on the trends in Wheatear populations at the large-scale analysis of change in relation to moorland management units.

d) Given the lack of strong management-related effects apparent for Wheatear in this analysis, it is difficult to propose particular management regimes for the conservation of this species which may be at the start of widespread declines in the uplands (Sim *et al.* 2005). The weak association of Wheatear with grassland habitats from this, and other studies, suggest that the recent trends of vegetation change in the uplands, with the conversion from dwarf-shrub to grass dominated vegetation communities as a result of high grazing pressures are unlikely to have been detrimental. With such a migratory species, which winters south of the Sahel, declines may be caused by factors on migration, or habitat change in the wintering grounds.

Whinchat

Populations of this passerine are stable or increasing across the UK uplands (Sim *et al.* 2005), with an increase in abundance observed in the Peak District where approximately 100 pairs were recorded (Carr & Middleton 2004, Appendix 1).

a) Fine-scale analyses of habitat associations indicate a strong association with bracken, and a secondary association with marshy grassland (Table 3.1). Such strong associations with bracken cover are recorded from other upland regions, and appear widespread (Stillman & Brown 1993, Brown & Stillman 1994, Allen 1995, Pearce-Higgins & Grant, 2006), whilst Pearce-Higgins & Grant also recorded an association with rush cover. Complementary analyses using the satellite derived vegetation data, which did not identify bracken habitats due to the timing of the May capture date of the image, highlighted weak associations with non-heather dwarf shrubs and quadratic association with heather cover. The strongest determinants of distribution were topography, with very strong associations for low altitudes with steep slopes, and a close association with areas of 25 - 30 cm vegetation height. Whinchat tend to associate with tall and dense vegetation (Pearce-Higgins & Grant, 2006), presumably as such areas have higher abundances of their invertebrate prey, and provide adequate nesting opportunities, although it is interesting that in this analysis, it is not the tallest vegetation that they prefer, but areas of intermediate height.

b) The associations with elevation and vegetation height were also apparent at the 1 km level, in conjunction with a positive correlation with disturbance levels. This presumably reflects the association of an unmeasured habitat feature, such as bracken, with footpaths. Thus, although the 1 km model appears to have little predictive power, it does support the results from the finer-scale associations, as discussed above.

c) Numbers of whinchat have increased most on low, and particularly high altitude sites, and on squares with low to intermediate heather cover and low non-heather dwarf shrub cover. Because we were unable to measure bracken cover using the satellite image, it is possible that these vegetation associations may be surrogates for the expansion of bracken into such areas (cf. Allen 1995). Consequently, it is difficult to make generalisations concerning the effects of habitat upon Whinchat population trends, from this analysis. There were no significant effects of either grazing or grouse-moor management upon Whinchat population trends.

d) Whinchat were patchily distributed at lower altitudes, with hotspots of abundance particularly associated with Pikenaze, Derwent and Big Moors. Given the relatively weak apparent effects of management on this species, which appears to require taller vegetation, particularly bracken, and possibly also marshy grassland areas, then management should ensure that sufficient patches of such habitats be maintained within the Peak District. However, as there have been widespread increases in Whinchat abundance in the UK uplands, it seems likely that larger-scale processes are driving the population expansion.

Wren

Although not commonly thought of as an upland bird, a significant and increasing population occurs in upland areas, and like Stonechat, presumably a consequence of the recent mild winters (Sim *et al.* 2005). The population in the Peak District has increased by a large amount in recent years. Few Wren were recorded in 1990, but they were the second most abundant species recorded in 2004. Their habitat associations on moorland have not previously been described.

a) Wrens tended to be recorded from bracken, scree, dwarf shrub heath, marshy grassland and scrub habitats, thus like Stonechat, they tend to favour vegetation categories with taller vegetation (Table 3.1). In the fine-scale analysis based on the satellite-derived variables, there were strong topographical associations with steep slopes at low altitude, close to streams. In relation to vegetation, heather and then cotton grass vegetations were the most favoured, with an additional weak, but significant, positive correlation with vegetation height (Table 3.2). A positive association with levels of disturbance may be linked to an association with particular habitat features close to paths, such as bracken (cf. Whinchat).
b) At the 1 km scale, the strong topographical effects on distribution remained, along with an additional quadratic correlation with non-heather dwarf shrub cover (Table 3.3). A positive association with the frequency of burning may be related to the association with heather (Table 2.6). It therefore appears that Wren occupy a fairly wide range of habitats, particularly of heather, but are restricted primarily to sloping ground at lower altitudes.
c) Given the few Wrens recorded in 1990, it was not possible to conduct a separate analysis of change for this species. However, the model of associations presented in (b) indicates that it is low altitude areas of with steep slopes, particularly of heather, that have been colonised.
d) The large population currently present within the Peak District Moors is probably a function of recent trends for warmer winters, and represents only a very small fraction of the national population. It is therefore unlikely that conservation management on moorlands will be specifically targeted at Wrens, and therefore are not considered further.

4.3. General principles of moorland management

There are three main land use drivers and issues relating to the management of moorlands within the Peak District; grazing management, grouse-moor management and tourism. Our analysis highlights general principles associated with each of these, which shall be discussed in turn, below.

Grazing management

In the long-term, high levels of grazing pressure, particularly caused by sheep, result in the conversion of dwarf shrub dominated vegetation, to grass, sedge and rush swards (e.g. Thompson *et al.* 1994). In the Peak District, this has lead to increases in the coverage of cotton grass, *Molinia*, *Nardus* and other grass species at the expense of heather moorland, with the resulting community depending on soil type and moisture. Intermediate grazing levels can promote Bilberry and Crowberry cover over heather (e.g. Rawes 1983, Welch 1984), but under chronic grazing regimes, these vegetation types are replaced. Grazing also has a strong effect on vegetation structure, with reductions in grazing intensity resulting in significant increases in vegetation height and density within a year (Rawes 1981, Hope *et al.* 1996, O'Brien *et al.* 2005). The results from the habitat association analyses can be used to place each bird species along these gradients of grazing related vegetation change; Heather – Bilberry/Crowberry – sedge/grass and tall – short vegetation.

In relation to the heather – grass gradient, results from this study suggest that it is Red Grouse populations which are most susceptible to sustained, high levels of grazing pressure, although habitat utility for Wren and Stonechat may also be reduced. Habitats produced by intermediate grazing levels tend to be favoured by species such as Whinchat, Ring Ouzel and Meadow Pipit, with the latter two both associated with mosaics of heather and grass. Finally, Golden Plover, Dunlin, Snipe, Wheatear and Skylark all appear to favour either sedge-dominated cotton grass bogs, or upland grasslands; habitats which tend to be promoted by long-term grazing pressures (Thompson *et al.* 1995).

In terms of vegetation structure, there is some suggestion that Dunlin, Snipe, Whinchat, Stonechat, Curlew and Lapwing may favour taller vegetation, although of course, the latter species is widely associated with short swards on in-bye fields (see species specific accounts). Both Skylark and Golden Plover, in contrast, tend to be associated with relatively short vegetation, such as produced by higher levels of grazing.

Thus, these broad-scale associations can be used to suggest how the density of each species may be affected by long-term grazing pressure, although the lack of strong effects of reduced grazing levels under different agri-environment schemes suggests that perhaps responses to variation in grazing levels by birds may be relatively weak, or that the grazing prescriptions currently associated with upland agri-environment schemes are insufficient to deliver the appropriate responses. Such schemes have tended to have limited success in restoring dwarf shrub vegetation (e.g. Hulme *et al.* 2002), and therefore it may not be surprising that we failed to detect any such effects.

Grouse-moor management

Grouse-moor management is associated with two legal practices that are likely to impact upon moorland birds. The first is the rotational burning of heather stands to generate young shoots for Red Grouse, and the second is the control of generalist predators such as corvids, Foxes and mustelids. Although a number of moorland bird species have been shown to occur at higher densities on grouse-moors, compared to non-grouse-moors, for a given habitat type (Tharme *et al.* 2001), the work presented in this report is the first attempt to relate changes in bird populations to the intensity of such management.

The distribution and changes in abundance of four bird species were related to burning management, although as areas of heather burning were closely linked to heather cover (Table 2.6), it is difficult to disentangle the management effect from the effect of habitat cover on species. Thus, there were apparently positive effects of burning on Ring Ouzel and Wren, but negative effects on Reed Bunting and Meadow Pipits. Much more research is required to assess in detail the impacts of burning on moorland birds. Although for some species, such areas of burning provide important nesting sites within an otherwise dense heather sward (Robson 2001, Whittingham *et al.* 2001), it is unclear how such management then subsequently affects invertebrate prey populations, a factor which should be the focus of future research.

We failed to detect any strong effects of such grouse moor management,
particularly on ground nesting waders. This is perhaps surprisingly, given the previous spatial associations, and detailed work illustrating the importance of predation in influencing the breeding success of some wader species (Parr 1993, Grant *et al.* 2001, Pearce-Higgins & Yalden 2003b). As discussed above (section 4.1), this may be because of density-dependence influencing territory settlement, as high breeding success in populations on grouse-moors at carrying capacity results in population increases on neighbouring areas, or be due to the relatively crude measure of grouse-moor intensity used. It would be valuable to examine variation in breeding success in relation to such management, which, could be achieved for Curlew using the five-visit survey method of Grant *et al.* (2000), without the need for intensive nest finding methods.

It is therefore difficult to draw strong take-home messages from this analysis, in relation to the effects of grouse-moor management on Peak District bird populations. Other studies have, and are, looking at this issue in more detail. It is noteworthy that Curlew, Golden Plover and other wader populations appear to be stable or increasing (with the exception of Dunlin) in the Peak District, in contrast to many other upland regions (Sim *et al.* 2005). As this is a region with a relatively high intensity of grouse-moor management, and it is plausible that such management may have contributed to these positive trends. However, this study provides no good evidence that this is the case. Sim *et al.* (2005) identify other upland regions with relatively high intensities of grouse-moor management remaining where wader populations have declined, so that the relationship between grouse-moor management and wader population trends does not appear a simple one. Indeed, the correlates of change for some species, such as the association for, and increases in, Golden Plover populations on cotton grass, suggest that other processes may be driving populations. In the absence of further information, the future conservation of these wader species within the Peak District is probably best promoted by continuing the current practices of burning and legal predator control, but more detailed monitoring of the impacts of both on breeding success would be valuable. In particular, the continuing decline in Dunlin populations provides cause for concern and needs careful monitoring.

Recreational Disturbance

The Peak District receives up to 22 million day visits per year, which results in heavy recreational use of some long-distance footpaths (Pearce-Higgins & Yalden 1997) and areas close to certain access points. There has been a long-history of concern over the impacts that such high visitor pressure has upon the breeding bird assemblage of the Peak District moorlands, particularly on ground-nesting waders, which has increased following the recent open access designations under the CRoW Act 2000.

This study suggests that a number of such species, particularly ground nesting waders such as Curlew, Golden Plover, Lapwing and Snipe avoid areas of habitat close to footpaths. The contrasting habitat associations included within this list of species, suggests this is a result of widespread phenomena, which is supported by more detailed research on Golden Plover (Finney *et al.* 2005). However, at the 1 km level, these effects appeared to have little impact on density, other than for Red Grouse, and there is some debate as to whether such avoidance of footpaths has a meaningful impact at the population level. It is likely that in some areas, high levels of disturbance, such as along the Pennine Way, does locally impact upon wader populations, although such effects are greatest where the movements of people are widespread and unpredictable (Finney *et al.* 2005). In such cases, it is possible to mitigate against detrimental effects of disturbance through the provision of a surfaced footpath (as provided through the Moors for the Future footpath restorations) with restricted access points (Finney *et al.* 2005), allowing high levels of visitors to a site with no detectable impact upon breeding success (Pearce-Higgins *et al.* in press). Increased effort should also be made to encourage people to keep dogs on a lead (cf. Pearce-Higgins & Yalden 1997), as stray dogs may contribute to the mortality of young birds in particular.

There is also concern within the Peak District over levels of disturbance to gully and cliff-nesting species such as Ring Ouzels from climbers. Results suggest that Ring Ouzels tend to occupy squares with intermediate levels of disturbance, but may avoid the most disturbed areas (Table 3.3). However, disturbance was not a factor in fine-scale distributions (Table 3.2); whilst numbers appear to be increasing in some of the more disturbed edges (cf. Fig. 3.5). This suggests that disturbance effects on this species are not strong, although more detailed work to investigate territory occupancy and breeding success in relation to spatial variation in disturbance may be required to be certain.

Despite very high levels of disturbance, populations of many of the wader species regarded as sensitive to disturbance are increasing in the Peak District. This suggests that visitor pressure is not currently a major driver of population change, a conclusion supported by stable Golden Plover, and increasing merlin populations within the South Pennines SPA since the early 1970s, despite this being a time of considerable increases in recreational pressure (A. Brown *in litt.*).

3.4. Conclusions

We have conducted detailed analysis of the habitat associations of moorland birds within the Peak District at two spatial scales, and assessed the importance of topographical, habitat and management features associated with population changes. In general, the results from the habitat associations support previous work within the South Pennines, or from other upland regions. Importantly, we have been able to model relationships with vegetation height and levels of disturbance in more detail than has previously been achieved for such large-scale analysis. Fine-scale analysis provided more accurate predictions of distribution than larger 1 km scale associations. Analysis of change at two spatial scales failed to highlight strong and simple drivers of change for many upland species, which may reflect the fact that for many species, in contrast to conservation issues on lowland agriculture, there are no strong, single drivers of population change. Alternatively, processes of change may be operating at larger spatial scales than examined here, for example relating to habitat fragmentation and changes in predator abundance. In this case, change could therefore still be linked to moorland management, or non-moorland related issues such as management on adjacent in-bye fields which are utilised by some moorland breeding species, such as Curlew, Golden Plover and Twite (Robson 1999, Pearce-Higgins & Yalden 2003). Changes in populations of other species may be driven by even larger scale processes such as climate change, or potentially change away from the breeding grounds. There is recent evidence that declines in Ring Ouzel populations may be linked to increasing summer temperatures in the UK (Beale *et al.* 2006), whilst milder winters may have improved survival rates for both passerines, such as Wren and Stonechat, and waders, such as Golden Plover (Pearce-Higgins & Yalden 1997). Further research is therefore required, and is currently being undertaken by the RSPB, and others, to examine the factors associated with population trends across a range of upland regions, which should help to address this issue. However, for rapidly declining species such as Ring Ouzel, Dunlin and Twite (Sim *et al.* 2005), such analysis should be complemented by detailed autecological work. This has already been at least partially achieved for Golden Plover (Whittingham 1996, Pearce-Higgins 1999), Curlew (Robson 1999), Ring Ouzel (Burfield 2002) and Black Grouse (Haysom 2001), with detailed work currently underway on Twite (Andre Raines at the UEA, funded by EN). Although the current survey methodology is adequate (given the caveats discussed above) for large-scale monitoring of upland bird populations, this analysis suggests that more detailed survey work may be required, using an experimental set-up, to fully test the effectiveness of particular management regimes in maintaining and enhancing moorland bird populations. High quality habitat and management data are also required for such analyses to be most informative.

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APPENDIX I

Summary of population changes on areas of moorland resurveyed in 2005 (Shepherd 2005)

Access to four areas of moorland was restricted in 2004 due to access refusals; Meltham Moor (around SE 0808), Snailsden Moor (around SE1302), Broomhead, Bradfield and Derwent Moors (SK1996 – SK2288) and Hallam Moor (around SK2386). No survey work was conducted in Meltham Moor or Snailsden Moor in 2004, partial coverage was achieved around Broomhead, Bradfield and Derwent Moors with access refused to 17 km squares, and Hallam Moor was surveyed in 2004 through a combination of Brown & Shepherd survey and observation from footpaths (Carr & Middleton 2004). Presented below are the estimated population changes for each of these areas from 1990 and 2005 in terms of numbers of presumed pairs from the Brown & Shepherd surveys.

	Meltham Moor		Snailsden Moor		Broomhead etc		Hallam Moor		Percentage
	1990	2005	1990	2005	1990	2005	1990	2005	change
Oystercatcher			0	2					increase
Golden Plover	4	3	10	11	23	56	16	38	104%
Lapwing			6	5	17	16	7	13	13%
Curlew	4	4	10	18	22	51	18	40	109%
Dunlin			2	0					decrease
Snipe			0	1	5	14	3	9	200%
Common Sandpiper			4	3					-25%
Wheatear			1	0	8	1	3	1	-83%
Whinchat	3	2	0	4	8	9	8	1	-16%
Stonechat	0	3	0	1	0	11	0	1	increase
Ring Ouzel	2	0			1	10	5	8	125%
Twite	10	1	4	0	3	1	2	0	-89%
Linnet	0	6			0	14	0	5	increase
Reed Bunting			0	1	1	8	0	3	1100%

Population changes from these data broadly reflect those from 1990 to 2004. Thus, the big increases in populations of Curlew, Snipe and Reed Bunting detected by Carr & Middleton (2004), and declines in Wheatear and Twite are replicated in these data. However, changes in Golden Plover and Ring Ouzel populations appear to have been more favourable on these resurveyed squares than the declines detected by Carr & Middleton (2004), whilst declining Whinchat populations contrast with a Peak District-wide increase in abundance. These differences could reflect true local variation in population trends, or may also be subject to yearly differences in estimates of the same population, dependent upon weather conditions, surveyors etc. For example, estimates of Golden Plover abundance from Brown & Shepherd surveys can vary by over 50 % between years at the same time, purely as a result of differences in the timing of breeding in relation to survey visits (Pearce-Higgins & Yalden 2005).

APPENDIX II

An assessment of the accuracy of the moorland bird survey

Pearson's correlations between Brown & Shepherd survey counts across 165 1 km squares surveyed in 2000 (see Sim *et al.* 2005) and 2004 (Carr & Middleton 2004), to assess the degree of annual variability in count at the 1 km level. Squares with double zero counts, and hence which may represent unsuitable habitat, are included for the correlations on the left, but excluded on the right; the latter being most likely to be correct. Wren and Reed Bunting were not surveyed in 2000, and therefore not analysed. Statistically significant ($P < 0.05$) correlations are in bold.

	Correlations across all squares		Double zero counts excluded		
Species	r	r^2	n	r	r^2
Curlew	**0.37**	**0.14**	124	**0.32**	**0.10**
Golden Plover	**0.54**	**0.29**	97	**0.41**	**0.17**
Lapwing	**0.29**	**0.08**	35	0.16	0.03
Ring Ouzel	0.12	0.01	51	-0.01	0.00
Stonechat	0.13	0.02	21	**0.55**	**0.30**
Snipe	0.02	0.00	32	-0.12	0.01
Whinchat	**0.37**	**0.14**	22	**0.61**	**0.37**
Wheatear	**0.28**	**0.08**	46	0.24	0.06
Dunlin	**0.64**	**0.41**	24	**0.55**	**0.30**
Red Grouse	**0.47**	**0.22**	148	**0.43**	**0.18**
Skylark	**0.45**	**0.20**	133	**0.44**	**0.19**
Meadow Pipit	0.17	0.03	162	0.16	0.03

These crude correlations suggest that there is considerable variability in the abundance of most moorland species at the 1 km level as estimated from Brown & Shepherd (1993).

APPENDIX III

Production of a vegetation map from a satellite image

The Landsat 7 image capured in 7/4/2003 was trained using vegetation data gathered from 37, 2 km^2 moorland plots surveyed in the South Pennines in 2002; of which 18 were in the Peak District as part of another project (Pearce-Higgins, unpubl.). These plots were located within the South Pennines using a random stratified sample based on heather cover. Within each plot, points were located at 100 m intervals along transects spaced 200 m apart, with a doubled sampling intensity along 2, 1 km transects in each plot, yielding 120 points per plot. At each point, vegetation height (mean of 3 measures of maximum height within 5 cm of a vertically held cane held at arms length to the right, left and ahead of the observer), and species composition (estimated to the nearest 5 % along a 1m length of cane) were recorded (see Pearce-Higgins & Grant, 2006 for more details of vegetation survey methodology). Data were collected from a total of 4,583 points, of which 492 were excluded from the analysis, to provide a test data set with which to assess the accuracy of habitat predictions.

Manipulation of the satellite image was conducted in Idrisi Kilmanjaro v. 14 (Clarke Laboratories 2004), with bands 1-5 and 7 geo-referenced to OSGB on Transverse Mercator projection to within 1 pixel accuracy (< 30m), and shade corrected using a 50 m digital terrain model. These bands were used for analysis as they each have a resolution of 30 m, and have previously been used to map moorland vegetation composition and structure (Buchanan *et al.* 2005). The appropriate six reflectance values were extracted for each vegetation sample point, and used to build artificial neural networks (ANNs) to predict the arcsine-square root transformed proportion cover of each of 10 vegetation types which were sufficiently frequent (present at > 5% of points) for analysis to be conducted. These were Matt grass *Nardus stricta,* Purple Moor Grass *Molinia caerulea,* fine-leaved grasses, Hares-tail Cotton Grass *Eriophorum vaginatum,* Common Cotton Grass *E. angustifolium,* Heather *Calluna vulgaris,* Crowberry *Empetrum nigrum* and Bilberry *Vaccinium myrtillus*. We also estimated vegetation height using the same method. Recognising that the methods involved in model selection using ANNs are not generally agreed upon (Zhang 2000), we took a heuristic approach to modelling habitat variables based on satellite reflectance measures (which were first transformed to fall between zero and one). In each of 20 replicates, models were iterated using random starting weights in the range −0.25 to +0.25 and a decay constant of 5 × 10 −4 (Tso & Mather 2001; Venables & Ripley 2002). Iterations continued until convergence to six decimal places. This involved fitting numerous models of increasing complexity (a ‘bottom-up’ choice of the number of hidden neurones: Anders & Korn 1999) and using each model to predict the habitat variables in the test datasets. We then plotted the correlation coefficient between predicted and observed results for each model against the number of hidden neurones. As models increased in complexity, their predictive ability (assessed from the correlation coefficient between observed and predicted cover across the test data) also increased, until the models started to overfit and the correlation coefficient started to decline. Using simple calculus on the best-fit quadratic model describing the relationship between correlation coefficient and model complexity, we selected the optimum number of hidden layer neurones that balanced model complexity against overfitting.

The accuracy of model predictions were assessed by correlating predicted values against observed values for the 492 test data points. As data from individual sample points described the vegetation across only a small fraction of each satellite image pixel, to reduce the sample error due to potentially unrepresentative points, we averaged predicted and observed cover across the excluded points for each of the 37 plots (mean 13 points per plot). This provided a second measure of overall predictive ability that better reflects the accuracy of each prediction. For all but two of the 9 habitat variables, calculated r^2 between predicted and observed covers were over 0.5, and better than equivalent predictions produced using a GLM (cf. Buchanan *et al.* 2005). Predictions were best for Heather ($r^2 = 0.825$) and worst for Common Cotton Grass ($r^2 = 0.212$). Predictions of vegetation height were related to observed height by the following equation (height = 6.63 + 0.73 * predicted height). Models were used to predict vegetation composition and height for every pixel of the satellite image representing land within unenclosed moorland areas.

	Point scale		Plot scale	
Vegetation variable	GLM	ANN	GLM	ANN
Nardus stricta	0.322	0.403	0.653	0.731
Molinia caerulea	0.484	0.603	0.653	0.812
Fine-leaved grasses	0.332	0.438	0.572	0.708
Eriophorum vaginatum	0.341	0.391	0.592	0.713
Eriophorum angustifolium	0.207	0.303	0.363	0.460
Calluna vulgaris	0.577	0.623	0.86	0.908
Empetrum nigrum	0.467	0.508	0.777	0.859
Vaccinium myrtillus	0.28	0.383	0.644	0.795
Vegetation height		0.36		0.624

Pearsons correlations (r) between observed and predicted values across the test data at the point scale and the plot scale using a GLM approach (cf. Buchanan *et al.* 2005), and Artificial Neural Networks (ANN). The plot scale analysis provides a more accurate assessment of predictive ability by reducing the error associated with points in vegetation locations unrepresentative of the overall vegetation composition within which they are located.

Several of the predicted habitat variables were strongly correlated, presenting considerable challenges to data analysis. To minimise these effects, we combined several strongly correlated variables and chose to consider only those habitat variables that made up > 10 % of the total cover. Consequently, our final satellite derived habitat variables were reduced to four: Heather (*Calluna vulgaris*), Grass (consisting of *Molinia*, *Nardus* and fine-leaved Grasses), Cotton Grass (*Eriorophorum* spp.) and Non-heather Heath (*Vaccinium mytrillus* and *Empetrum nigrum*), together with vegetation height. The plot scale correlations (r) for grass was 0.798, cotton grass was 0.634 and non-heather heath was 0.728, The proportion cover of Heather and Grass remained strongly negatively correlated (r = -0.499).

Appendix IV

Comparison of fine-scale model predictions using models based on vegetation data from either ESA habitat maps or remote sensing

Species	Satellite vegetation data		ESA habitat data	
	AUC	*P*		
Curlew	**0.659**	**0.003**	0.5	1
Golden Plover	**0.845**	**0**	0.5	1
Lapwing	**0.723**	**0.003**	**0.737**	**0.002**
Reed Bunting	**0.899**	**0**	nc	nc
Ring Ouzel	**0.861**	**0**	**0.752**	**0.001**
Stonechat	**0.69**	**0.011**	nc	nc
Snipe	**0.867**	**0**	0.5	1
Whinchat	**0.778**	**0**	nc	nc
Wheatear	**0.762**	**0.001**	0.546	0.290
Wren	**0.787**	**0**	0.5	1
Dunlin	nc	nc	nc	nc

Accuracy of predictions of habitat suitability based on fine-scale model predictions, applied to a random subset of the data excluded from the model building process (see text). Models for Reed Bunting, Stonechat, Whinchat and Dunlin failed to converge with the ESA habitat data, whilst Dunlin only failed to converge with the satellite derived vegetation data (although a model with just linear terms was able to converge – Table 3.4). Because of the poor predictions with the ESA data for species where convergence was possible, we did not pursue these models further, concentrating instead on using the satellite vegetation data.