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ORIGINAL PAPER

Estimating relative population size included within protected areas

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Abstract We propose and test a method to determine the proportion of specific animal national populations included in a network of protected areas. The proposed method further allows identifying the best potential new sites to be included in the network, to reach target population sizes and so to test whether a network extension is realistic or not. We used data from the French Breeding Bird Survey (BBS) and spatial interpolation models known as kriging to predict the relative abundance of species at a national scale. We applied the proposed methodology to Special Protection Areas (SPAs) and a sample of 20 bird species concerned by the Directive of the Council of the European Community on the Conservation of Wild Birds. We estimated which relative part of the national population is included within the boundaries of all national SPAs. Our results suggest that the current SPA network is probably not efficient to ensure favorable national conservation status for the most widespread species, but allows reaching a 5% target value for more localized ones. Consequently, we discuss the limit of such a protected area network to ensure the global conservation of widespread species, and therefore the need for other large-scale conservation measures.

Keywords Breeding Bird Survey · Kriging · Population size · Protected area assessment · Spatial interpolation · Special Protection Areas

Introduction

Defining and testing methods to identify or evaluate protected area networks are major issues in conservation biology (Warmw et al. 2004). The main raised questions concern the location and number of protected areas (Hoctor et al. 2000; Hannon and Schmiegelow

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2002; Thiollay 2002), their shape, size and composition (Kunin 1997; Hooker et al. 1999; Wielgus 2002; Halpern 2003; Donnelly and Marzluff 2004), but also the ways to maximize species abundance or richness in the network (Church et al. 2000; Polasky et al. 2000). Evaluating the effectiveness of protected areas is generally a neglected issue (Lü et al. 2003), and several authors noticed that numerous evaluation methods are largely descriptive and conceptual (Hockings 1998). Moreover, the few empirical studies conducted mainly focused on a limited number of sites (e.g., Caro 2001; Guillemain et al. 2001; Fabricius et al. 2003; Warmw et al. 2004).

Among existing protected area networks in Europe, Nature Reserves and National Parks are certainly the most renowned among public or politic audience, though a more recent network was created following the application of two directives of the European Union: the Habitat Directive (92/43/EC) and the Bird Directive (79/409/EC). The main aim of this program, called ''Natura 2000,'' is to form a coherent European ecological network for the protection of biodversity, by the special management of Special Areas of Conservation (SACs, for the protection of habitats listed in Annex I and of species listed in Annex II of the Habitat Directive) and Special Protection Areas (SPAs, for the protection of bird species listed in the Bird Directive). The European Commission requires Member States to report on the efficiency of their network to ensure a favorable conservation status for the habitats and species concerned. Some studies have already shown that this program is largely incomplete, not only for the SACs (Dimitrakopoulos et al. 2004) but also for the SPAs (Madsen et al. 1998) and also proposed better management to ensure the conservation of biodiversity in SACs (Muller 2002). A first step in the evaluation of this network should be to estimate the proportion of all concerned habitats or species that are effectively included within the boundaries of the designated areas, and eventually compare these estimates with target values considered to be efficient to ensure a global favorable conservation status. This work can be quite straightforward for rare or localized habitats or species, by direct census of all individuals within the populations. However, this is not the case for more widespread species for example, for which quantitative data on local abundances over a country are generally missing.

Here we develop a simple method that allows determining the proportion of a national species population that is effectively included in a network of protected areas. The aim of this paper is therefore to address two main objectives: (i) to present a method to estimate which part of a national population is included in protected areas; (ii) to identify solutions in order to improve the current network. We illustrate this by evaluating which proportion of national population sizes are included in French SPAs for 20 fairly common breeding species in France but representing a conservation stake at a European scale (concerned by the Bird Directive). Though we chose to test the method for the case of bird species and SPAs, the same work could have been carried out for any other animal populations and for other types of protected areas, whenever data on local relative abundances can be obtained through large scale national monitoring programs.

Methods

Measuring species local relative abundances

Data on birds were collected in continental France during the spring 2001–2004 campaigns of the national Breeding Bird Survey (BBS) currently operating in France (Julliard and

Jiguet 2002). Observers were asked to prospect squares (2 × 2 km, i.e., 4 km^2) selected from a national grid (comprising 136,477 possible squares). Random selection was stratified by observer density. Each observer received a locality, and the square to prospect was randomly selected within a 10 km-radius circle around this locality (i.e., c.80 possible squares). Each square was monitored twice in the spring before and after the 8th of May, with 4–6 weeks between sampling events. Each observer realized 10 point counts within a given square and counted every detected species on each point during 5 min exactly, beginning the survey at dawn. Point counts were at least 400 m apart from each other, and spread homogeneously within the available habitats. Each point was sampled in the same order during the first and the second sampling session. Data were available for 983 squares across the country (Fig. 1). Although local density in surveyed squares was not constant, each prospected square was surveyed with the same standardized method, so that the observation effort was identical in each surveyed square, though with potential heterogeneities in observers' abilities to detect the different species (Julliard and Jiguet 2002).

Fig. 1 Location of all 938 squares that have been surveyed by the French Breeding Bird Survey during the period 2001–2004 and that have been used in this study

SPAs and bird species studied

The SPAs network considered in this study includes all 136 sites already designated in continental France in December 2004 (excluding Corsica), which represent a total of 11,739 km^2, thus 2.18% of the national continental territory. We retained five species listed in Annex I of the EU Bird Directive and 15 more species concerned by the Directive, either as game birds (5 species) or long-distance migrants (10 species). All are well encountered by the national breeding bird survey, detected on at least 30 different squares, with at least 50 individuals detected across sites each year. The species listed in Annex I are (mean number of individuals detected in 2001–2004 for all 983 squares/number of squares where the species has been detected over the period, are given in brackets): Woodlark *Lullula arborea* (803/344), Tawny Pipit *Anthus campestris* (68/35), Red-backed shrike *Lanius collurio* (360/270), Dartford warbler *Sylvia undata* (100/52) and Ortolan Bunting *Emberiza hortulana* (57/31). Game species are: Grey Partridge *Perdix perdix* (413/126), Stock Dove *Columba oenas* (168/106), Turtle Dove *Streptopelia turtur* (2878/736), Skylark *Alauda arvensis* (5020/647) and Mistle Thrush *Turdus viscivorus* (1296/543). Long-distance migrants considered here are Common Cuckoo *Cuculus canorus* (3299/776), Hoopoe *Upupa epops* (706/348), Barn Swallow *Hirundo rustica* (7455/878), Tree Pipit *Anthus trivialis* (923/410), Whinchat *Saxicola rubetra* (203/135), Willow Warbler *Phylloscopus trochilus* (574/318), Western Bonelli's Warbler *Phylloscopus bonelli* (390/169), Wood Warbler *Phylloscopus sibilatrix* (136/108), Garden Warbler *Sylvia borin* (766/451) and Spotted Flycatcher *Muscicapa striata* (82/92).

Modeling relative abundances of species at a national scale

For each monitored square in each year, the maximum number of individuals detected on each point count during either the first or the second sampling session was calculated for each species, then the maxima were summed for each square, the sum being retained as a measure of local relative abundance. We further calculated a mean relative abundance for each square across the 4 years of survey, this value being used in further analyses. The fact that we are using relative abundance values, measured similarly on any surveyed square, and moreover that we used means of yearly relative abundances over 4 years of survey, make us confident that under-detection should be low on the surveyed sites. To summary, a species would be detected in a square if it is detected on 1 of the 10 point counts done twice a year during 4 years. With 5 min duration for each point count, a single square is potentially sampled during 400 min over 4 years to detect any species.

We used geostatistical models known as ordinary kriging models (Johnston et al. 2001), that account for spatial autocorrelation in order to predict values for each of the 136,477 squares. A comparison of the values measured on each point (e.g., species abundance measured in one BBS square) with the prediction obtained for the same square, allows to measure the prediction error at this point. Kriging models thus provide prediction error measurements that can be used for model selection in a simple way when compared to other predictive methods (Boone and Krohn 1999; Anderson et al. 2003), and the overall power of the interpolation model can be computed. We compared statistics on prediction errors for different number of neighbors considered to obtain estimates, and further retained the best number of neighbors as 8. Semivariograms used in this study were exponential models (Johnston et al. 2001) and we performed the different kriging models with the Geostatistical Analyst implement of ArcGIS 8.1 (ESRI 1999). Semivariograms showing the modeled spatial autocorrelation for four of the 20 species considered in this study are presented in Fig. 2.

Comparing relative abundances in SPAs with national totals

For each species, we obtained predictions of relative abundance for each of the 136,477 squares of the national grid. Using the Geographical Information System, we superimposed the 153 SPAs' layer to the national grid, and obtained for each square a ''coefficient in SPA,'' i.e., the proportion of the 4 km^2 that is really included in the Special Protected Area network (this coefficient ranging from 0 = 0% to 1 = 100% in SPA). For each square and each species, the relative abundance in SPA was calculated by multiplying the local predicted relative abundance by the square's ''coefficient in SPA.'' For each species, we further summed all relative abundances in SPAs on one hand and in all squares on the other hand, calculated the ratio between the two sums, and compared the obtained ratios with the target values.

For the 24 BBS surveyed squares that are located within a SPA, we compared predicted and observed values of relative abundance for each of the five species listed in Annex I of the Bird Directive. We used Wilcoxon signed-rank tests for paired data to compare log-transformed data on abundance for each of the five species considered. This analysis was

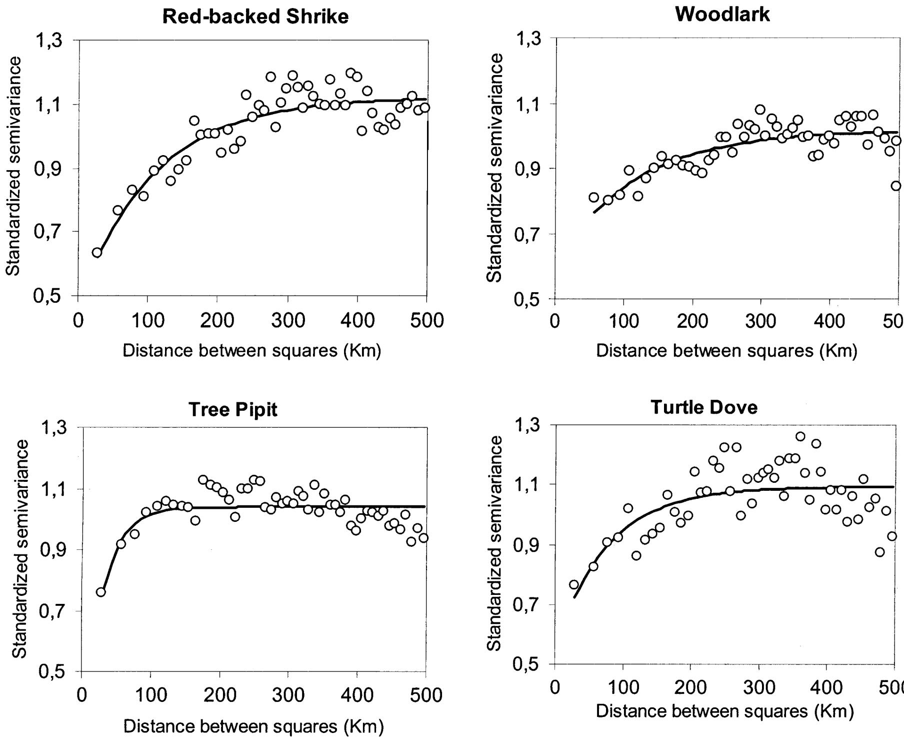

Fig. 2 Semivariograms showing the modeled spatial autocorrelation for four of the 20 species considered in this study. Graphs present standardized semivariance according to the distance between surveyed sites in kilometers (exponential models). Ranges for these models vary from 105 km to 320 km

conducted with the package SPLUS 2000 (Mathsoft 1999). The mean distance between a square surveyed in a SPA and the closest surveyed square (outside the SPA) used to interpolate values within the SPA was 9.0±5.4 km (range 2–30 km). This distance is far smaller than the ranges of spatial autocorrelation found for the 20 species studied here, so that we are confident in the quality of the selected interpolation method to predict values within SPAs.

Proposing size and location of SPAs to be created

We aimed at identifying the minimal area to be further protected to improve the existing network, in order to increase the relative population size of the considered species in SPAs. After excluding all squares which centers were included in existing SPAs' boundaries, we ranked all other squares according to the relative abundance of the considered species. However, some of these squares were partly in a SPA (i.e., coefficient in SPA < 1). For these squares we multiplied the predicted relative abundance by (1 − coefficient in SPA) before sorting the complete set. We computed the best cumulative relative abundances until the threshold was crossed, thus identifying a minimal area to be designated as SPAs to reach enlarged target values. This analysis was conducted for three species listed in Annex I of the Bird Directive, with the least relative abundance already included in the SPAs' network.

Results

Relative population size included in the protected area network

The Tawny Pipit, the Dartford Warbler and the Ortolan Bunting were all most southerly-distributed species, with strongholds localized around the Mediterranean Sea (Fig. 3). The Red-backed Shrike and the Woodlark, two insectivorous farmland species, were fairly widespread over the country (Fig. 3). For each species, the kriging model was validated thanks to the cross-validation statistics. For the three more localized species (Tawny Pipit, Dartford Warbler and Ortolan Bunting), more than 5% of the national population sizes were included within the SPA network (Table 1). For the two more widespread species (Red-backed shrike and Woodlark), less than 2.5% of the national populations were included within the boundaries of the SPA network (Table 1). Among the 15 more species considered here, none reached the 5% target national population size included in the protected area network (Table 1).

For the 24 squares located within a SPA and surveyed in 2003, we found no difference between predicted and observed values of relative abundances for the Tawny Pipit ($Z = 0.50$, $P = 0.62$), the Dartford Warbler ($Z = -0.11$, $P = 0.91$) and the Ortolan Bunting ($Z = -0.63$, $P = 0.53$), but predictions were significantly higher than observed relative abundances for the Woodlark ($Z = 2.17$, $P = 0.030$) and the Red-backed Shrike ($Z = 2.20$, $P = 0.028$).

Identifying potential new SPAs to reach targets

Quantitatively, we noted contrasted results among species. For the Dartford Warbler, it would be necessary to increase the area of the current SPA network by ca. 6% to double the

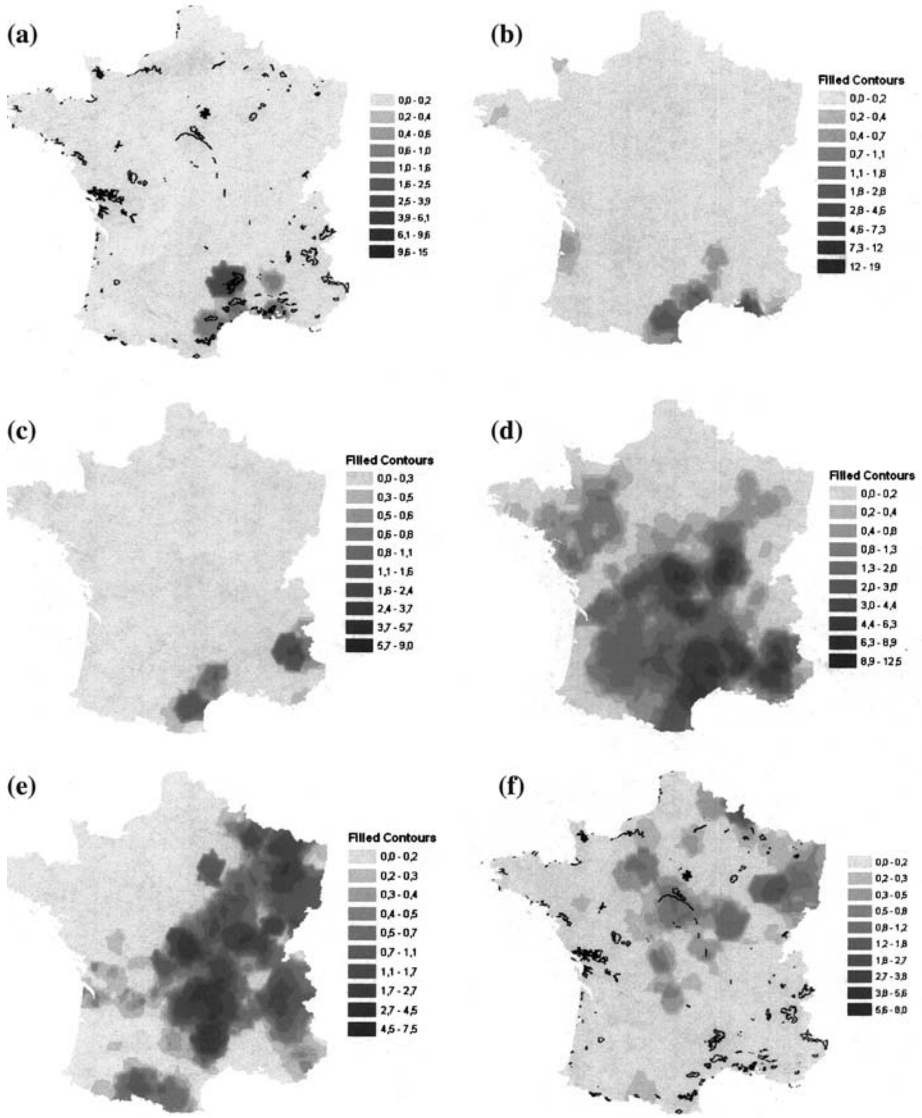

Fig. 3 Maps of modeled relative abundances for six breeding birds concerned by the Bird Directive (graphic output of ordinary kriging models). The Darker the gray color, the higher the relative abundance. (**a**) Tawny Pipit *Anthus campestris*; (**b**) Dartford Warbler *Sylvia undata*; (**c**) Ortolan Bunting *Emberiza hortulana*; (**d**) Woodlark *Lullulla arborea*, (**e**) Red-backed Shrike *Lanius collurio*; (**f**) Wood Warbler *Phylloscopus sibilatrix*. SPAs' boundaries are shown as black lines for Tawny Pipit (**a**) and Wood Warbler (**f**), the two species with the largest and one of the smallest relative abundance in SPAs, respectively, according to our results

population size included in SPAs (i.e., to reach a target α of roughly 10%). At the opposite, it would be necessary to increase by more than 20% the area covered by current SPAs (+21.6% for Red-backed Shrike, +23.5% for Woodlark) to double the population sizes of the two other species in SPAs (i.e., to reach a target a of roughly 5%). These relationships

Table 1 Proportion of national population sizes included in Special Protection Areas in France for 20 species concerned by the Directive of the Council of the European Community on the Conservation of Wild Birds. The species long-term trends are also given for the period 1989–2003, when available (in bold when statistically significantly different from stability, Julliard and Jiguet 2005)

Species		% in SPAs	Long-term trend (1989–2003)
Annex I Bird Directive			
Woodlark	*Lullula arborea*	2.47	+37%
Tawny Pipit	*Anthus campestris*	11.06	–
Red-backed Shrike	*Lanius collurio*	2.06	−5%
Dartford Warbler	*Sylvia undata*	5.39	–
Ortolan Bunting	*Emberiza hortulana*	8.65	–
Game species			
Grey Partridge	*Perdix perdix*	0.68	**−54%**
Stock Dove	*Columba oenas*	1.01	**−57%**
Turtle Dove	*Streptopelia turtur*	1.66	+9%
Skylark	*Alauda arvensis*	2.15	**−16%**
Mistle Thrush	*Turdus viscivorus*	1.87	−15%
Trans-Saharan migrants			
Common Cuckoo	*Cuculus canorus*	1.94	**−28%**
Hoopoe	*Upupa epops*	2.70	**−56%**
Barn Swallow	*Hirundo rustica*	1.63	**−30%**
Tree Pipit	*Anthus trivialis*	2.20	**−44%**
Whinchat	*Saxicola rubetra*	3.38	**−60%**
Willow Warbler	*Phylloscopus trochilus*	1.07	**−57%**
Western Bonelli's Warbler	*Phylloscopus bonelli*	4.84	**−58%**
Wood Warbler	*Phylloscopus sibilatrix*	0.76	**−79%**
Garden Warbler	*Sylvia borin*	1.35	−14%
Spotted Flycatcher	*Muscicapa striata*	1.53	**−59%**

can be extrapolated to check what area is needed to reach an hypothetic conservation target α. Figure 4 presents a graph where percentage of the national area to be designated as SPA is plotted according to varying target values α, for the three species cited above. Graphically, species can be classified in two classes: species for which hypothetic conservation goals (α) can be reached easily by creating a fairly small surface of new SPAs (e.g., Dartford Warbler), and species for which it would be necessary to designate large new surfaces in order to increase the protected population sizes (Red-backed Shrike and Woodlark). This investigation can also be done spatially by mapping the squares to include

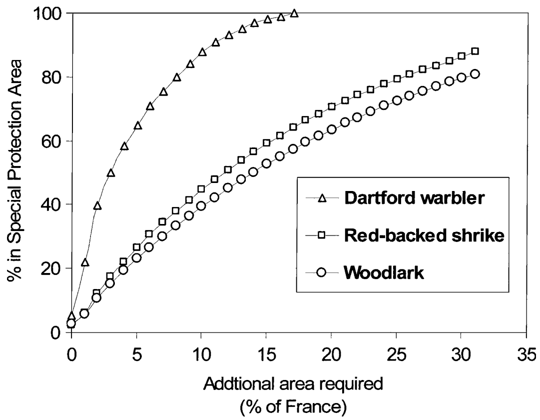

Fig. 4 Relationship between the target value α and the additional area required to include this part of the national population in SPAs, for three species listed in Annex I of the Bird Directive

preferentially in the SPA network in order to reach a target α of 5% (illustrated for the Red-backed Shrike in Fig. 5). Despite large surfaces are required, the sites are quite spatially localized and overlapping for the two species.

Discussion

Whereas large-scale monitoring programs are usually used to assess temporal trends in biological diversity (Yoccoz et al. 2001), spatial trends in such data are rarely studied (but see Nichols et al. 1998; Doherty et al. 2003; Jiguet and Julliard 2006). Thanks to a well-designed BBS and spatial interpolation tools, this method allowed us to evaluate a protected area network at a national scale and for several species. This method is not only descriptive and quantitative but further allows testing the opportunity of enlarging the current network spatially. Whereas several studies only used the probability of occurrence of the species in order to improve reserve networks (see Li et al. 1997; Manel et al. 1999, 2001; Fleishman et al. 2001; Wilson et al. 2005), we further considered the relative abundance of the species. The relative abundances predicted here certainly underestimate the true abundance, as all individuals are not detected during the BBS sampling sessions. For example, about 8,000 individuals of Woodlark are predicted by kriging, whereas the French population size was estimated between 50,000 and 500,000 pairs in 1997 (Labidoire 1999). However, our goal was to compare relative abundances inside versus outside protected areas, so to evaluate which part of a national population, and not how many pairs, was included in the protected area network. We also suspect some SPAs of the network to hold larger population sizes of the target species than the neighboring wider countryside, so that predicted values there might under-estimate local values in protected areas. This could be true as some SPAs should have been designated in order to encompass local strongholds of the species, so should have been placed on sites where their abundances are higher than average, though as all SPAs are not holding strongholds of all

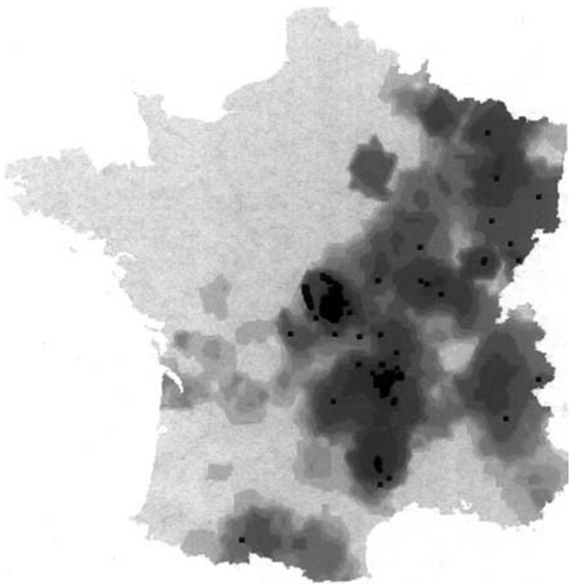

Fig. 5 Identified BBS squares (in black) that would allow reaching a 5% target value of national population size of Red-backed Shrike included within a putative extended SPA network

concerned species, this would not be a general rule. However, by comparing predicted and observed values for BBS squares included in SPAs boundaries, we failed to find a significant under-estimation of relative abundances for the five species considered there, a result that further strengthens the accuracy of the tested method. On the contrary, the interpolation models predicted higher abundances than observed values for two of the five species, which means that by using spatial autocorrelation the model was able to correct for eventual under-detection of such species there when compared to adjacent sites.

Using the interpolation models, we also demonstrated how the proposed methodology could help with identifying the best sites to be eventually designated as future SPA to reach higher target values of protected population sizes. By selecting the minimum number of squares (i.e., 4 km^2 areas) that maximizes the proportion of national populations to be protected, we did not consider any spatial constraint for the network efficiency. For example, the size of the selected patches should not be too small, and different aggregates of squares should not be too distant, if we consider that the viability of the national populations depends on the viability of the protected meta-populations, beyond also evident administrative and political constraints linked to the designation of many fragmented protected areas. However, the use of kriging models that account for and use the spatial autocorrelation led to the identification of largely aggregated sites, and protecting such best aggregates could ensure a favorable conservation status for the specific meta-populations, as long as these sites might act as sources for less healthy populations.

Though the method proposed here was tested at a species-specific level, further developments should consider a multi-specific approach, as some of the possible future SPAs certainly contribute to improve the status of more than one species. Such developments should aim at maximizing proportion of each species-specific national population to be included in protected areas, and also at minimizing the total area to be necessarily protected to reach these multi-specific targets. This should enable to identify the optimal protected area network. Anyway, there is always an uncertainty associated with the use of predicted species distribution (Wilson et al. 2005). So even if propositions of the best sites to include in SPAs can be quickly done, a further on-ground inspection is required. Creating new protected areas is often costly in time and money (Pressey and Tully 1994), and our propositions based on predicted relative abundances can be considered as a gain in such costs. It appeared to be a light way firstly to evaluate if protecting a significant part of the national population of a species is realizable and suitable.

As we shown in this study, the French SPA network should cover 20% more area if we just need to protect 10% of one of the most widespread species considered. This result alone illustrates the inefficiency of such a protected area policy for ensuring favorable conservation status for non-rare or non-localized species. However, their fate could be improved by implementing large-scale conservation measures in the wider countryside, and agri-environmental schemes might be obliging relevant tools for such species (Vickery et al. 2004; but see Kleijn et al. 2001). A general ''upstream conservation policy'' has to be set up: common species have to be preserved before they become rare and localized. Anyway, designing protected areas is often the most effective solution for threatened or localized species and is crucial to face with increasing pressures on environment (Groombridge and Jenkins 2002).

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