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Can common species benefit from protected areas?

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ABSTRACT

We studied the role played by the French protected area network on 100 common bird species at the population and community level. The long-term trend of each species was calculated over 15 years (1989–2003). We then used 418 plots monitored by the French Breeding Bird Survey over 2001–2005 either inside or outside the national protected area network (including Nature Reserves, National Parks and Special Protected Areas) to compare each species' average density inside and outside the protected areas. We then tested if the relative species densities in protected areas were related to the species long-term trends. At the community level, we assessed the average proportions of the most severe long-term declining species inside and outside protected areas as well as their temporal stability. We found that several species, which are mostly dependent on human activities, had both higher densities in unprotected areas and exhibited a negative long-term trend. However, for most species, we found that the more a species has declined over the 15 years, the higher its density in protected areas. At the community level, declining common species were found in higher proportion and exhibited greater temporal stability in protected areas. Our results emphasize that many common but declining species could benefit from protected areas and that large-scale monitoring programs provide highly valuable quantitative tools for extensive protected area assessments.

1. Introduction

The protected area approach is a classic conservation policy, defended on the grounds that protected areas conserve species or habitats that are under threat elsewhere (Fabricius et al., 2003). This assumption has already been tested by looking at how protected areas covered the spatial distribution of a particular protected species (De Klerk et al., 2004), of some taxonomic groups (Pawar et al., 2007) or of global biodiversity (Rodrigues et al., 2004).

In the context of current global changes, protected areas should ideally represent spatial aggregates allowing the preservation of a large sample of biodiversity into the future (Gaston et al., 2006; Strange et al., 2006). Surprisingly, little

attention has been paid to the potential role of protected areas in protecting the most familiar species. Yet, the ultimate extinction of species is just the tip of the iceberg of global wildlife erosion: many common species are now showing declines at large temporal and spatial scales (Conrad et al., 2006; Donald et al., 2006).

The decline of common species threatens to disrupt vital ecosystem processes (Sekercioglu, 2006) and results in the loss of good candidates helping biodiversity adaptation to global changes (Luck et al., 2003). Common species also provide useful surrogate indicators of ecosystem function and health (Gregory et al., 2005). Monitoring the fate of common species should therefore be a worthwhile strategy to ensure long-term and holistic biodiversity conservation planning (Balmford et al., 2003).

In this respect, one may ask if and how populations of common species located in protected areas are different from elsewhere. Such comparisons are largely absent from conservation studies, although advances in data availability and large-scale surveys should enable such evaluation. Indeed, large-scale bird surveys were shown to be a promising tool for measuring the suitability of a protected area network to contain populations of certain species (Godet et al., 2006).

Ideally, to demonstrate that protected areas have real positive effects, one needs to show that the status of species has improved more (or deteriorated less) on the protected areas than elsewhere. Nevertheless, assessing if a protected area network, which is motivated by the protection of rare and threatened species, is also able to support populations of many declining common species is a first necessary step (Dunk et al., 2006).

In this paper, we used data on the 100 most common bird species in France, to test the prediction that declining species could benefit from protected areas in space and time. We first compared each species'density inside and outside the protected area network. Then, we assessed whether the temporal stability of the most rapidly declining species were enhanced in protected areas.

2. Methods

2.1. Data collection

Data on bird species came from the breeding bird survey (BBS) running in France, using a standardized monitoring methodology (Julliard and Jiguet, 2002). Each observer provided a locality, and a 2×2 km plot to be prospected was randomly selected within a 10 km-radius of this location (i.e. among 80 possible plots). Such random selection ensured the survey of varied habitats (including farmland, forest, suburbs and cities). Each plot was monitored in two annual visits during spring, one before and one after the 8th of May, with 4–6 weeks between the two visits. To be validated, the count must be repeated on approximately the same date each year (±7 days from April to mid-June) and the same time of day (±15 min within 1–4 h after sunrise). In each plot, the observer carries out 10 evenly distributed five minute point counts, during which every individual from species that are heard or seen is recorded. For a given point count and a given species, the maximum counts of the two annual visits is retained. Points counts are 300 m apart and sampled in the same order by the same observer each year.

For each point count, the surroundings within a 100 m radius were classified by observers as belonging to one of a standardized list of habitats. This list was organized into a standardized land use description, inspired by the one developed for the British BBS (Crick, 1992), and covered 18 habitat classes.

2.2. Site selection

Out of the entire set of BBS plots monitored by this monitoring program ($n = 1205$), we selected the 180 plots located in the French protected area network and monitored for at least two years during 2001–2005. The plots were located either in National Parks ($n = 24$), Nature Reserves ($n = 17$), or in Special Protected Areas ($n = 109$). Nature Reserves (NR) are small strictly protected areas, with very few human activities permitted, and generally managed for the protection of localized threatened species or habitats. National Parks (NP) are large protected areas, mostly located in mountains where traditional land use is preferred and human disturbance is strongly restricted. Special Protected Areas (SPA) were designated by a more recent network following the application of the Bird Directive of the European Union (79/409/EC). The main goal of this program, called Natura 2000, is to form a coherent Europe-wide ecological network for the protection of bird species, listed in the Bird Directive.

We sought to compare BBS plots located inside and outside the protected area network according to the geographical location of each plot. Therefore, we further selected neighbouring BBS plots located in the vicinity of the 180 protected BBS plots, which were monitored for at least two years between 2001 and 2005, but located in areas with no protection status. To perform this selection, we retained all BBS plots ($n = 238$) that fell within a fixed 15 km radius of each protected BBS plot, from the global BBS data set of the 1205 surveyed plots. Note that plots in both protected and unprotected areas were subject to the same standardized monitoring scheme.

2.3. Species selection and classification

We first wanted to test if declining common species had higher densities in protected areas. We thus focused on the 100 most common species monitored by the French BBS (Table 1). Each species long-term trend was estimated by fitting a Poisson regression with site and year effects on species abundance. The latter was provided using a 15 year time series from 1989 to 2003 from a previous version of the French BBS (Julliard and Jiguet, 2005). The trend was not available for 16 species because their abundance was too low to produce reliable estimates (Table 1).

We then considered, in a single group, the 25 most rapidly declining species (i.e., species with the lowest long-term temporal trends) and five common species which are listed in the Appendix I of the EU bird directive (79/409/EC). For control purposes, we built a second group with the 30 species exhibiting the highest temporal trends (Table 1). The 60 selected species were all frequently encountered in the national breeding bird survey, detected on at least 30 different BBS plots, with at least 50 individuals detected across plots each year.

2.4. Population and community parameter estimates

At the population level, we averaged species abundance at each point count over the monitoring period. During 2001–2005, a given plot was monitored on average 3.18 years. In all analyses, the effect of the number of years a given plot was monitored was not significant.

At the community level, we first estimated the number of species belonging to each group (declining and control) on each BBS plot and each year. These estimations were based on capture–recapture methods developed for animal

Table 1 – Each species' long-term trend (noted na if not available) and relative density in protected areas

Species		Trend	Relative density	Criteria
Eurasian Sparrowhawk	Accipiter nisus	−0.69	0.06 **	Declining
Long-tailed Tit	Aegithalos caudatus	−0.06	−0.01 ***	–
Skylark	Alauda arvensis	−0.16	0.07 **	–
Red-legged Partridge	Alectoris rufa	−0.28	0.11 **	–
Tawny Pipit	Anthus campestris	na	0.00 *	Annexe 1
Meadow Pipit	Anthus pratensis	−0.41	0.36 ***	Declining
Tree Pipit	Anthus trivialis	−0.44	0.27 ns	Declining
Common Swift	Apus apus	−0.12	−0.23 ***	–
Eurasian Thick-knee	Burhinus oedicnemus	na	0.10 ***	–
Common Buzzard	Buteo buteo	−0.17	−0.08 **	–
Eurasian Linnet	Carduelis cannabina	−0.52	0.09 ***	Declining
European Goldfinch	Carduelis carduelis	0.19	−0.14 ***	Non-declining
European Greenfinch	Carduelis chloris	−0.31	−0.24 ***	Declining
Short-toed Tree-Creeper	Certhia brachydactyla	0.33	−0.05 ***	Non-declining
Cetti's Warbler	Cettia cetti	2.27	−0.22 *	Non-declining
Western Marsh-Harrier	Circus aeruginosus	na	0.45 **	–
Hen Harrier	Circus cyaneus	na	−0.52 ***	–
Montagu's Harrier	Circus pygargus	na	0.33 **	–
Zitting Cisticola	Cisticola juncidis	na	0.35 ***	–
Hawfinch	Coccothraustes coccothraustes	0.71	−0.10 ***	Non-declining
Common Pigeon	Columba livia	na	−0.55 **	–
Stock Dove	Columba oenas	−0.57	0.14 ns	Declining
Common Wood-Pigeon	Columba palumbus	0.45	−0.23 *	Non-declining
Common raven	Corvus corax	na	0.76 ns	–
Carrion Crow	Corvus corone	−0.04	−0.10 ***	–
Rook	Corvus frugilegus	0.01	−0.51 **	Non-declining
Eurasian Jackdaw	Corvus monedula	−0.42	0.00 ns	Declining
Common Quail	Coturnix coturnix	0.08	0.22 ns	Non-declining
Common Cuckoo	Cuculus canorus	−0.28	0.10 ns	–
Northern House-Martin	Delichon urbica	na	−0.12 ns	–
Great Spotted Woodpecker	Dendrocopos major	0.22	0.04 *	Non-declining
Lesser Spotted Woodpecker	Dendrocopos minor	−0.73	−0.04 ns	Declining
Black Woodpecker	Dryocopus martius	0.66	−0.16 ns	Non-declining
Cirl Bunting	Emberiza cirlus	0.54	−0.25 ns	Non-declining
Yellowhammer	Emberiza citrinella	−0.31	0.01 ns	Declining
Ortolan Bunting	Emberiza hortulana	na	0.03 ns	Annexe 1
European Robin	Erithacus rubecula	0.82	−0.07 ns	Non-declining
Common Kestrel	Falco tinnunculus	−0.28	−0.06 ns	–
Chaffinch	Fringilla coelebs	−0.11	−0.03 ns	–
Crested Lark	Galerida cristata	na	0.29 ns	–
Eurasian Jay	Garrulus glandarius	−0.05	−0.11 ns	–
Melodious Warbler	Hippolais polyglotta	−0.2	−0.07 ns	–
Barn Swallow	Hirundo rustica	−0.3	−0.16 ns	Declining
Eurasian Wryneck	Jynx torquilla	−0.47	0.01 ns	–
Red-backed Shrike	Lanius collurio	−0.05	0.29 ns	Annexe 1
Red Crossbill	Loxia curvirostra	na	0.51 ns	–
Wood Lark	Lullula arborea	0.37	0.14 ns	Annexe 1
Common Nightingale	Luscinia megarhynchos	−0.16	−0.08 ns	–
European Bee-eater	Merops apiaster	1.32	0.02 ns	Non-declining
Corn Bunting	Miliaria calandra	−0.27	0.10 ns	–
Black Kite	Milvus migrans	0.24	0.28 ns	Non-declining
White Wagtail	Motacilla alba	0.15	−0.05 ns	Non-declining
Yellow Wagtail	Motacilla flava	0.71	0.17 ns	Non-declining
Spotted Flycatcher	Muscicapa striata	−0.59	0.16 ns	Declining
Northern Wheatear	Oenanthe oenanthe	na	0.70 ns	–
Eurasian Golden-Oriole	Oriolus oriolus	−0.17	−0.17 ns	–
Coal Tit	Parus ater	−0.38	0.58 ns	Declining
European Blue Tit	Parus caeruleus	0.07	−0.14 ns	Non-declining
Crested Tit	Parus cristatus	−0.29	0.38 ns	–
Great Tit	Parus major	0.01	−0.13 ns	Non-declining
Willow Tit	Parus montanus	−0.5	0.61 *	Declining
Marsh Tit	Parus palustris	−0.6	0.11 ns	Declining
House Sparrow	Passer domesticus	−0.11	−0.24 ns	–

Table 1 – continued

Species		Trend	Relative density	Criteria
Eurasian Tree Sparrow	*Passer montanus*	−0.37	0.03 ns	Declining
Grey Partridge	*Perdix perdix*	−0.54	−0.49 ns	Declining
Common Pheasant	*Phasianus colchicus*	0.4	0.20 ns	Non-declining
Black Redstart	*Phoenicurus ochruros*	−0.06	0.03 ns	–
Common Redstart	*Phoenicurus phoenicurus*	−0.21	0.23 ns	–
Western Bonelli's Warbler	*Phylloscopus bonelli*	−0.58	0.31 *	Declining
Common Chiffchaff	*Phylloscopus collybita*	−0.12	−0.08 ns	–
Wood Warbler	*Phylloscopus sibilatrix*	−0.79	0.51 ns	Declining
Willow Warbler	*Phylloscopus trochilus*	−0.57	0.22 ns	Declining
Black-billed Magpie	*Pica pica*	−0.62	−0.28 *	Declining
Eurasian Green Woodpecker	*Picus viridis*	0.25	−0.05 ns	Non-declining
Hedge Accentor	*Prunella modularis*	−0.14	-0.09 *	–
Red-billed Chough	*Pyrrhocorax pyrrhocorax*	−0.45	0.31 ns	Declining
Firecrest	*Regulus ignicapillus*	−0.18	0.09 *	–
Goldcrest	*Regulus regulus*	0.03	0.22 ns	Non-declining
Bank Swallow	*Riparia riparia*	na	0.13 ns	–
Whinchat	*Saxicola rubetra*	−0.6	0.52 **	Declining
Common Stonechat	*Saxicola torquata*	0.71	0.05 *	Non-declining
European Serin	*Serinus serinus*	−0.37	−0.09 ns	Declining
Wood Nuthatch	*Sitta europaea*	−0.49	0.10 ns	Declining
Collared-Dove	*Streptopelia decaocto*	2.17	−0.34 **	Non-declining
European Turtle-Dove	*Streptopelia turtur*	0.09	−0.08 *	Non-declining
Common Starling	*Sturnus vulgaris*	−0.09	−0.28 *	–
Blackcap	*Sylvia atricapilla*	0.05	−0.11 **	Non-declining
Garden Warbler	*Sylvia borin*	−0.14	−0.03 ns	–
Subalpine Warbler	*Sylvia cantillans*	−0.17	0.01 *	–
Greater Whitethroat	*Sylvia communis*	0.04	0.10 ***	Non-declining
Lesser Whitethroat	*Sylvia curruca*	0.15	−0.33 **	Non-declining
Sardinian Warbler	*Sylvia melanocephala*	0.6	0.11 **	Non-declining
Dartford Warbler	*Sylvia undata*	na	0.36 **	Annexe 1
Winter Wren	*Troglodytes troglodytes*	0.3	−0.12 **	Non-declining
Common Blackbird	*Turdus merula*	0.18	−0.15 *	Non-declining
Song Thrush	*Turdus philomelos*	0.69	−0.03 **	Non-declining
Fieldfare	*Turdus pilaris*	na	−0.65 ***	–
Mistle Thrush	*Turdus viscivorus*	−0.15	0.15 ***	–
Eurasian Hoopoe	*Upupa epops*	−0.56	0.07 **	Declining
Northern Lapwing	*Vanellus vanellus*	−0.21	0.41 ***	–

Among the 100 studied species, the 30 most declining and the 30 least declining species were respectively grouped in a declining and in a control group. The relative difference in protected areas is tested using a generalized mixed model accounting for spatial dependence between plots and using habitat-adjusted estimates of each species abundance (ns: $P > 0.05$; *: $P < 0.05$; **: $P < 0.01$; ***: $P < 0.001$).

communities using the freeware program COMDYN (Hines et al., 1999). This program considered the 10 point counts within a BBS plot as sampling replicates of the local community within the geographical unit (i.e. the $4\ km^2$ cell of the BBS plot). The presence–absence matrix of detected species over the ten sampling units was processed each year respectively for each group (declining and control) to estimate the number of species belonging to the declining group $(\widehat{S}_d)$ and to the control group $(\widehat{S}_C)$ each year. We then calculated the relative number of species belonging to declining species detected on each plot each years (i.e., $\widehat{S}_d/(\widehat{S}_d + \widehat{S}_C)$). This proportion was arcsin transformed and averaged over the study period in each plot.

Secondly, we studied the temporal stability of each group by studying changes in year-to-year community composition. This was calculated as the rate of local turnover, defined as the proportion of species at time $t + 1$ estimated to be locally new since time t (Nichols et al., 1998; Boulinier et al., 2001). This parameter accounted for heterogeneity in detectability among species and was calculated for each

group (declining and control) on each BBS plot for each time interval.

2.5. Data analysis

At the population level, our aim was to compare each species abundance recorded in protected and unprotected plots. As a part of the difference in species abundance may have resulted from systematic differences between habitats in protected and unprotected areas, all statistical analyses were performed controlling for this effect. We thus first used, for each species, a general linear model (GLM) assuming poisson error with the species abundance as the dependent variable, and plot and habitat as factors (i.e., using the habitat documented in the field by observers in each point count, among the 18 habitat classes). This first analysis was conducted using the 4180 point counts (i.e., 418 plots of 10 point counts with 180 and 238 plots respectively monitored in protected and in unprotected areas). We used the habitat-adjusted abundance estimates at the plot level provided by this first statistical model for further analysis.

We then tested for the effect of the BBS type (protected or not) using spatial generalized mixed models (GLMM) which accounted for spatial location of each plot. Such models were performed in two steps. First, we studied the shape of the semivariograms drawn from generalized linear models (GLM) using habitat-adjusted abundance estimates of each species as the dependent variable and type of plot (i.e. protected or not) as a factor. This semivariogram analysis was performed to obtain parameters describing the spatial autocorrelation structure of the data (Fortin and Dale, 2005).

Second, we performed a complete General Linear Mixed Model (GLMM) allowing for spatial dependence in the errors. This mixed model incorporated the spatial correlation structure (given by semivariogram analysis) and each sample coordinates as random effect (Lin and Zhang, 1999). These spatial mixed models were performed using logarithm $(x + 1)$ transformed data of the habitat-adjusted abundance estimates. Therefore, this first analysis provided a comparison of each species density between protected and unprotected plots, adjusted to habitat, inside and outside protected area networks, accounting for the spatial dependence of each sample.

The same model was used to investigate more specific effects, using the type of protection as a factor (i.e., no protection, NR, NP, SPA). Finally, to describe the link between vulnerability and difference in population densities in protected and unprotected samples, we further tested the correlation between difference in species relative densities in protected and unprotected areas and species long-term trends (when available).

At the community level, the capture–recapture models provided reliable estimates for a given group (declining or control) only if at least six species were detected by the observer in this group. Respectively 125 and 158 plots fulfilled this condition among protected and unprotected plots. As community parameters were available at the plot level (i.e., not at the point count level), we aggregated habitat information to account for difference in habitat among plots in the statistical analysis. Each plot was thus designated as belonging to a broad habitat classification according to the number of points monitored in the different habitat classes. Among the 283 plots considered, 136 plots had at least six point counts in the same habitat and were thus classified either as farmland ($n = 44$); forest ($n = 47$); semi-natural grasslands ($n = 29$); and urban samples ($n = 16$). The other 147 plots had only five or fewer point counts in the same habitat type and were grouped into a single heterogenous habitat class.

As for abundance comparisons, we used GLMMs accounting for spatial dependence between samples and controlling statistical analysis for the habitat type. These models were run separately with the proportion of declining species and turnover (either of the vulnerable or the control group) as the dependent variable, habitat type (farmland, forest, semi-natural, urban and heterogenous) and type of plot (protected or not) as factors. Finally, to determine more specific effect of each protected area type, the same models were also run using type of protection as factor (i.e. no protection, NR, NP, SPA).

3. Results

Among the 100 species considered, 45 had higher densities in protected plots (Table 1). Taking all species together, the differences in species density between protected and unprotected plots were negatively correlated with long-term trends. We tested this correlation excluding the collared dove, *Streptotelia decaocto*, (a species that has been naturally colonizing Europe since the 1950s): we found that the more rapidly declining a species, the higher its relative density in protected area ($R^2 = 11\%$, $F_{1,82} = 10.80$, $P = 0.001$, Fig. 1). Among the 30 most declining species, 20 species had higher densities in protected plots, whereas among the control group, only 10 species had higher densities in protected areas (Table 1).

At the community level, the relative number of vulnerable species was 7% higher in protected plots than in unprotected ones (respectively 37.7 ± 0.84% se, 35.1 ± 0.88% se; GLMM: $F_{1,277} = 4.25$; $P = 0.04$; Fig. 2a). More specifically, the proportion of declining species increased respectively in SPAs (36.1 ± 1.0%), NRs (38.3 ± 1.8%) and NPs (42.2 ± 2.3%).

No difference in species turnover of the control group was detected among protected and unprotected plots (GLMM: $F_{1,277} = 0.21$, $P = 0.64$; Fig. 2b). Conversely, the local turnover of declining species was 28.6% higher in unprotected than in protected plots (GLMM: $F_{1,277} = 4.20$, $P = 0.04$; Fig. 2b). Turnover of declining species in protected plots increased respectively from PNs (7.8 ± 1.4% se), SPAs (14.1 ± 1.4%) to RNs (16.8 ± 2.3%).

4. Discussion

4.1. A protected area assessment based on common species

In this study, we assessed the relative abundance and temporal stability of declining and non-declining common species,
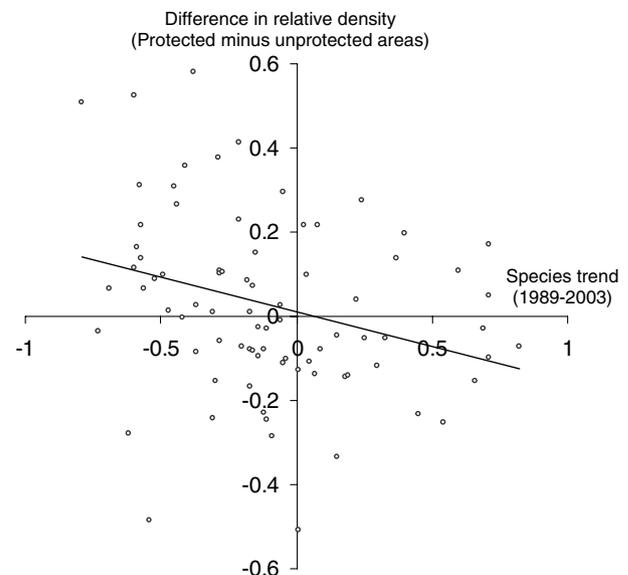
Fig. 1 – Relationship between differences in relative species density measured in protected and unprotected areas and the long-term trend (1989–2003) of the species.

a Composition
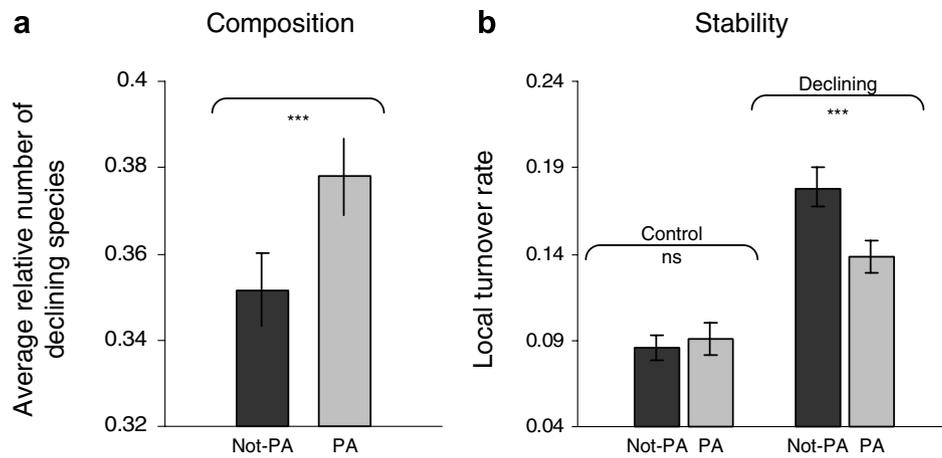
b Stability

Fig. 2 – Comparison of (a) average relative number of declining species and (b) temporal stability of declining and control species measured in protected (*n* = 125) and unprotected samples (*n* = 158), respectively noted PA and Not-PA. The mean comparison were performed using general linear mixed model controlling for the difference among sample habitat type and accounting for spatial dependence between samples (ns: *p* > 0.05, ***: *p* < 0.001).

both in protected and in non-protected areas. We showed that declining common species (i.e., species with negative long-term trend) had higher relative densities in protected areas and exhibited higher temporal stability in these areas. These differences could simply be caused by the fact that protected areas were specifically established in places that included more abundant and more stable declining species than elsewhere. In this case, we showed that protected areas which were mostly designed to protect rare or threatened species (or remarkable habitats) can also include higher densities of common declining species although they were not designed for this purpose. Alternatively, several common species may have benefited from protected areas following specific management actions, although the latter were not motivated by these species. In both cases, our results suggest that protected areas could now constitute important spatial refuges for declining common species and can achieve holistic biodiversity conservation goals (Simberloff, 1998).

4.2. A non-random segregation of species in protected areas

Interestingly, we showed that the more severe the long-term decline of a species, the higher its current density in protected areas. But beyond this general pattern, a close inspection of the differences in species densities revealed more specific results. Indeed, several species with both a lower relative density in protected areas and a negative long-term trend (e.g., *Hirundo rustica*, *Perdrix perdrix*) were cavity nesters on human settlements, or dependent on voluntary feeding by human for game management (Buner et al., 2005). This result stresses that some species need to rely more on areas where people live and work than on protected areas anyway (Miller and Hobbs, 2002).

Furthermore, our results revealed that species were not randomly distributed among protected and non-protected areas according to their habitat preference. Indeed, one can rank species from the more to the less habitat special-

ized according to the variation in their proportions across different habitat classes (Gregory et al., 2005; Julliard et al., 2006; Devictor et al., 2007). For instance, species with skewed abundance in specific habitats (i.e., that can be considered to be habitat-specialized in France) had significantly higher densities in protected samples than elsewhere (e.g., farmland birds: *Alauda arvensis*, *Carduelis cannabina*, *Saxricola rubetra*, *Anthus pratensis*, *Alectoris rufa*, *Vanellus vanellus*; Woodland birds: *Phylloscopus bonelli*, *Parus montanus*, *Turdus viscivorus*). Conversely, several habitat-generalist species (showing very low variation in their densities across habitats in France) had higher densities in non-protected areas (*Turdus merula*, *Parus major*, *Parus caeruleus*, *Columba palumbus*).

Specialization is an expected evolutionary response to habitat stability (Futuyma and Moreno, 1988). The pattern we observed may thus be explained by the higher habitat quality and/or stability in protected areas. In this respect, Canova (2006) showed that the process of habitat change likely explained the difference in species richness between protected and unprotected areas. In our study, we could not measure the dynamic of the landscape within and outside the protected areas. However, the difference in landscape disturbance in protected versus non-protected areas probably contributes to explain the segregation of species according to their habitat specialization.

4.3. Higher declining species richness and stability in protected areas

Shifting the response variables from individual species to groups of species that share a common attribute (i.e. declining or not) reinforced previous findings. We showed that the proportion of most declining species was higher in protected areas. Not only do declining species have higher relative densities in protected areas: a given set of species recorded in protected areas was also composed, on average, of more declining species.

We were also able to show more specific differences between protected areas: the relative number of declining species increased respectively from Special Protected Areas, through Nature Reserves to National Parks. Such difference among protected areas is both a question of land use and scale. Indeed, NPs protect rural lifestyles, landscapes and traditional practices which encompass many habitats and different kinds of land-use. Therefore, more common species are likely to benefit from such areas. In contrast, SPAs which have more specific management for specific endangered species (or habitats) may be highly beneficial to some species, but are unable to enclose many different species. To refine the assessment of each protected area type, comparing species-specific temporal trends in each protected area will be a promising step forward. We could not perform this analysis because the latter will require a longer time series than 5 years to get relevant estimates.

Finally, we found that species turnover within the declining group was lower in protected areas. This result was not found for the control group, suggesting that we found a true difference in declining species stability in time between protected and non-protected areas.

The latter result can be explained because species with low local densities are expected to experience higher level of local extinction (Hanski, 1999) and that declining species had lower densities outside protected areas. Therefore, at the community level, a declining species present in a given year in a non-protected plot had, on average, a lower probability of still being present in this plot the following year. These latter results concerning community stability suggest that protected areas had a stabilizing effect on declining species, probably by buffering the negative effects of habitat degradation occurring elsewhere.

4.4. Monitoring programs as valuable tools for extensive protected area assessment

Most attempts to quantify protected area efficiency have centred around estimates of species richness or have documented distribution patterns of targeted species as snapshots in time (but see Claudet et al., 2006). Our results suggest that large-scale monitoring programs can also be useful for extensive assessment in providing quantitative measurements of the fate of many common species in different sites.

However, using data coming from wildlife monitoring programs raises the problem of specific bias induced by the way data are collected in the field. Indeed, counts of species (or individuals) are the result of two processes: a biological process (true presence or absence of a species or individuals), and a methodological filter (species or individuals that are truly there need to be detected in order to be counted) (Thompson, 2002). If variation in detectability among species (or individuals) is not adequately accounted for, potentially most of the variation in the presence of a given species (or its abundance) result from variation in the detectability of species (or individuals), regardless of the variation in the true species presence (or abundance).

Many authors have stressed that this problem, which is often ignored, should be carefully addressed (Gu and Swihart, 2004). In this respect, several methodological tools were specifically developed to take variation in detectability of species and individuals into account (Boulinier et al., 1998; Royle et al., 2005). In our particular case, grouping species within a declining and a control group could have led to a systematic over-estimation of the richness of one group (if, for instance, declining species were on average harder to detect). We therefore explicitly accounted for such difference among species, using capture–recapture techniques when studying community richness and dynamics. In doing so, it was acceptable to consider that the community was closed during a monitoring session so that each point-count can be considered as spatial replicates of the same community of a given plot (Nichols et al., 1998).

In contrast, concerning relative abundance, it was not possible to use capture–recapture or distance sampling to correct abundance estimate. Indeed, point counts were not replicated in time (populations must be closed between temporal replicates to run capture–recapture algorithms) and only two classes of distance were filled by observers so that distance sampling was not applicable. However, our qualitative results concerning difference in species density would be weakened by imperfect detectability only if there was a systematic interaction between detectability of individuals of declining species and protected areas. All plots were monitored with the same standardized protocol and relative abundances were compared controlling for the difference in habitats. Therefore, even if we obviously did not compare true abundances, we see neither statistical nor ecological evidence for how imperfect detectability could induce strong bias in our results and alter our principal findings (note that results based on relative abundance were in accordance with results based on community parameters which accounted for imperfect detectability among species).

Protected area assessments based on the hot-spot and the red-listed species concepts have generally concluded that reserves alone are not adequate for nature conservation (Rodrigues et al., 2004; Virkkala and Rajasärkkä, 2007). Assessment of protected areas based on many common species should now also enable the investigation of impacts of human activities on biodiversity in a dynamic way at a wide array of scales. Such investigations represent great opportunities for measuring what happens to familiar species from human-dominated areas to more strictly protected areas, instead of restricting analyses to what happens to threatened species in strictly protected areas.

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