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Exposure of European biodiversity to changes in human-induced pressures

M.B. Araújo^{a,*}, D. Nogués-Bravo^a, I. Reginster^b, M. Rounsevell^b, R.J. Whittaker^c

^a Department of Biodiversity and Evolutionary Biology, National Museum of Natural Sciences, CSIC, C/Gutiérrez Abascal 2, 28006 Madrid, Spain

^b Département de Géographie, Université Catholique de Louvain, Place Louis Pasteur 3, B-1348, Louvain-la-Neuve, Belgium

^c Biodiversity Research Group, Oxford University Centre for the Environment, South Parks Road, Oxford OX1 3QY, UK

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ABSTRACT

There is increasing evidence that areas with high concentrations of species tend to have high concentrations of human activities. Would this tendency be altered with projected environmental changes? We investigate this possibility using the most extensive available dataset on species distributions in Europe, providing data for 3143 species. Observed land uses were utilised to generate three indicators of anthropogenic 'pressure' for 1971–2000: urbanisation, cropland and grassland use intensities. Storylines describing alternative development pathways were used to create four land-use scenarios for 2021–2050. Exposure of biodiversity to human activities was measured as changes in the degree of pressure recorded in hypothetical reserve networks selected to maximize the representation of plant, breeding bird, mammal, amphibian, and reptile species. In all socio-economic scenarios there was a tendency for increasing urbanisation and decreasing cropland intensities within selected conservation areas. Pressures arising from increasing grassland use were variable across scenarios and taxa. Our results challenge the idea that a single development strategy might provide reduced impacts in all regions and taxonomic groups. We show that impacts are likely to be complex and that tradeoffs might exist among development strategies. Nevertheless, results are contingent on the data, scale, and type of analyses conducted and further research is required to assess the impacts of alternative human-development scenarios.

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1. Introduction

There is increasing evidence from analyses of large geographic regions that areas with high concentrations of species also tend to have high concentrations of humans (e.g. Balmford et al., 2001; Araújo, 2003; Real et al., 2003; Diniz-Filho et al., 2006; Vázquez and Gaston, 2006; O'Dea et al., 2006; Luck, 2007), and therefore of human activities. Authors have speculated that the correlation between people and biodiversity may arise because humans and other living organisms depend on

processes that are driven or perhaps delimited by contemporary available energy (e.g. Balmford et al., 2001; Luck, 2007). Alternative mechanisms for this relationship have been proposed (e.g. Fjeldså and Rahbek, 1998), but regardless of the explanation, one consequence of the coincidence between people and biodiversity is that pressures inflicted on important areas for biodiversity might be greater than expected by chance (Cincotta et al., 2000; Araújo et al., 2002; Chown et al., 2003; van Rensburg et al., 2004) and sometimes difficult to avoid (Araújo and Rahbek, 2007). As societies adapt to global

* Corresponding author.

E-mail address: maraujo@mncn.csic.es (M.B. Araújo).

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environmental changes the distribution of human activities may change and the current patterns of coincidence between human activities and native species may be altered. From a species' conservation perspective the important question is whether the socio-economic changes that follow from climate change will increase or decrease the level of anthropogenic pressures on biodiversity? We investigate this question by assessing changes in the exposure of biodiversity to a number of indicators of pressure between 1971–2000 and 2021–2050 (after 2050 projections of change become very uncertain and a very large number of simulations would be required to produce meaningful assessments). A range of socio-economic scenarios describing alternative development pathways for the future are used to generate the land use scenarios from which the indicators of anthropogenic 'pressure' (i.e. land use changes of likely negative impact) are calculated.

Whilst conservationists have historically focused on activities such as hunting, pollution, introduced species, habitat loss and fragmentation, as drivers of species loss, current scientific and popular concern has increasingly emphasised the possible implications of future climate change under a range of global-warming scenarios. Recent developments in modelling biotic responses to climate change have gained considerable attention (for debate see Hannah and Phillips, 2004; Ladle et al., 2004), yet it is also apparent that we currently have limited ability to forecast species changes in abundance or range (e.g. Araújo et al., 2005; Whittaker et al., 2005; Pearson et al., 2006). One reason for this is because, for many species, contemporary changes of climate have been of lesser significance compared to change in land use and agricultural practices (e.g. Sala et al., 2000). There are several difficulties with modelling the response of species to climate change (for reviews see Guisan and Thuiller, 2005; Araújo and Guisan, 2006; Heikkinen et al., 2006), not least of which is the problem of how to predict species responses to the combined effects of climate and land use changes (e.g. Zebisch et al., 2004; Reidsma et al., 2006). One reason is that knowledge of the synergistic effects of climate and land use on the distributions of species remains very incomplete (e.g. Thuiller et al., 2004; Luoto et al., 2007). An additional reason is that modelling land use changes is a challenge in itself (e.g. Busch, 2006). Human land use is affected by private and public choices and these are in turn affected by external factors such as climate change (e.g. Lambin et al., 2001). The complex nature of human decisions makes it difficult, if not impossible, to predict future land uses with any degree of confidence.

To address the difficulties in modelling land use change a range of 'what if scenarios' based on an interpretation of qualitative 'storylines' can be produced. These scenarios cannot be interpreted as predictions of future land uses. Instead they should be viewed as sets of coherent and internally consistent simulations based on plausible but necessarily simplified assumptions of how the future may develop. Here we use four hypothetical worlds to describe a range of possible future socio-economic development pathways and their corresponding land use changes. The approach is based on an interpretation and quantification of the narrative storylines of the Intergovernmental Panel on Climate Change (IPCC) Special Report on Emissions Scenarios (SRES) (Nakicenovic et al., 2000). The SRES storylines represent

alternative visions of the future evolution of society that reflect either economic or environmental objectives within a global or regional development strategy. In order to translate land uses into measures of pressure for biodiversity three indicators were used: (1) degree of urbanisation; (2) degree of cropland intensification; and (3) degree of intensity of grassland use. Finally, to avoid the difficulties with modelling dynamic responses of species to the combined effects of climate and land use changes (which would be a different study) a simple, but transparent, approach was adopted: we measured the magnitude of modelled changes in all three indicators of pressure within sets of important areas for biodiversity conservation in Europe. Even though this approach is simplistic and does not take into account the variety of species responses to environmental changes, it allows us to provide a first assessment of future land use changes for areas that are important for biodiversity conservation in Europe.

2. Data and methods

2.1. Mapping anthropogenic pressure

The four alternative SRES scenarios used here are as follows (Nakicenovic et al., 2000). A1FI (global economic) is a fossil fuel intensive world of rapid economic growth, low population growth and rapid introduction of new and more efficient technologies. Major underlying themes are economic and cultural convergence and capacity building, with a substantial reduction in regional differences in per capita income. The focus is on the accumulation of personal wealth rather than increasing environmental quality. Technological development still relies on fossil intensive energy sources. A2 (regional economic) is a very heterogeneous world. The underlying theme is that of strengthening regional cultural identities, with an emphasis on family values and local traditions, high population growth, and less concern for rapid economic development. B1 (global environmental) is a convergent world with rapid change in economic structures, "dematerialization" and introduction of clean technologies. The emphasis is on global solutions to environmental and social sustainability, including concerted efforts for rapid technology development and improving equity. B2 (regional environmental) is a world in which the emphasis is on local solutions to economic, social, and environmental sustainability. It is again a heterogeneous world with less rapid, and more diverse technological change but a strong emphasis on community initiatives and social innovation to find local, rather than global solutions. The land use scenarios used in this study were developed for the former 15 European Union member states, Switzerland and Norway, using input climate data from the Hadcm3 GCM (Mitchell et al., 2004). They are spatially explicit and were originally projected onto a 10' resolution grid (for more details see Kankaanpää and Carter, 2004b,a; Ewert et al., 2005; Reginster and Rounsevell, 2006; Rounsevell et al., 2005, 2006).

Baseline (i.e., current conditions) and scenarios of future land use changes for the A1FI SRES scenario are shown in Fig. 1. Degree of urbanisation was measured as the proportion of urban land within each 50 km grid cell. Degree of cropland

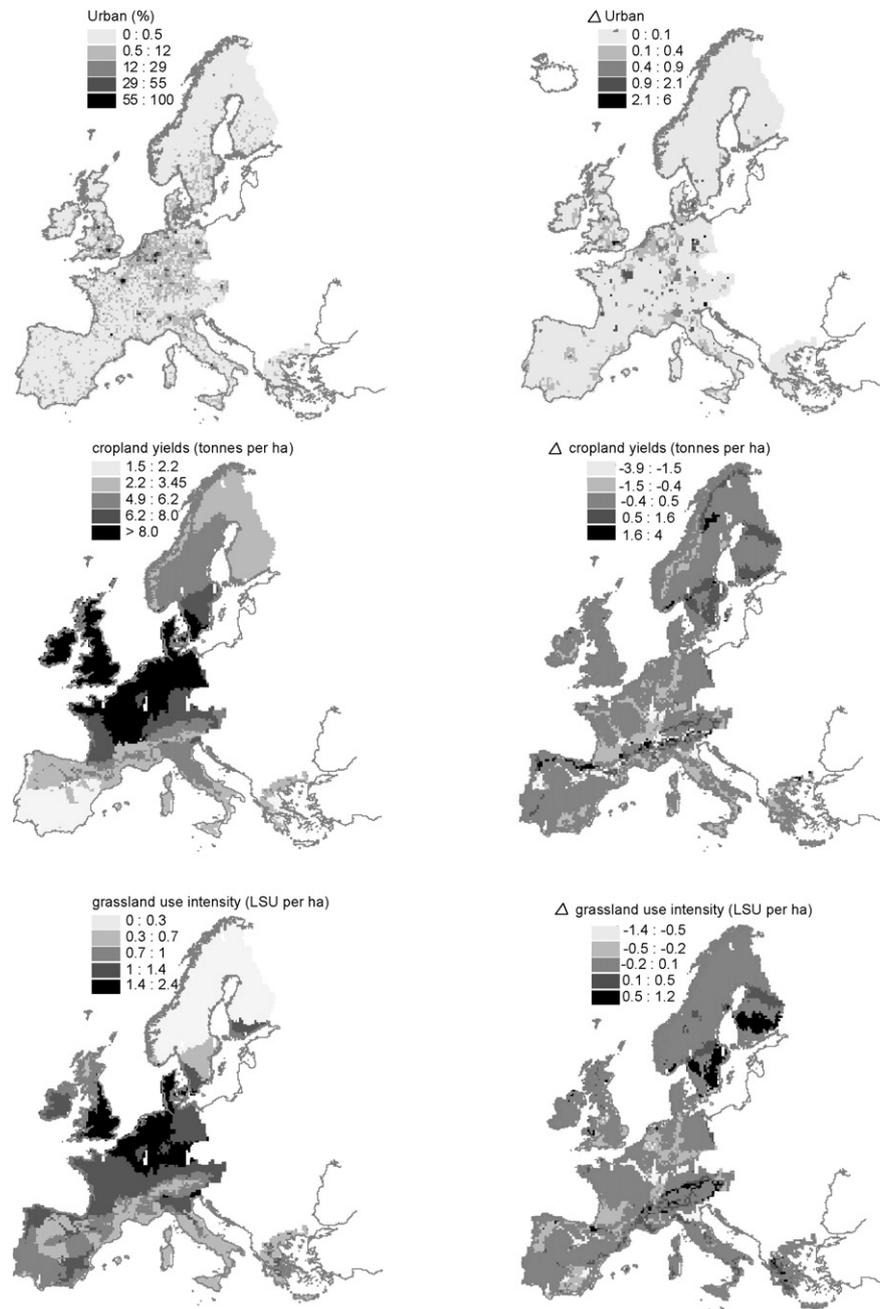


Fig. 1 – Map of land use intensity (1970–2000) and change (using an A1FI scenario, 2020–2050) in Europe using a 10' grid resolution. Land uses are aggregated in three indicators of pressure: urbanisation (%); cropland yields (tonnes per ha) land; grassland use intensity (LSU per ha).

intensification was measured using yields of winter wheat (tonnes per ha) as a proxy for intensification: higher yields were assumed to be commensurate with increased inputs (see also Donald et al., 2000). Degree of livestock stocking unit intensity (LSU per ha) was used as a measure of intensity of grassland use; this was calculated as cattle LSU + goat LSU + sheep LSU divided by the sum of grassland area, maize fodder and green fodder. The estimation of wheat yields and of stocking intensity was based on a spatial overlay of observed agricultural statistics from the Regio database for NUTS2 (common European classification of territorial units for

statistical purposes) regions in 1990 (Eurostat, 2000) with the Environmental Stratification (EnS) maps for Europe (Metzger et al., 2005). The EnS is a bioclimatic classification derived from the geographic patterns of climate. Maps of the EnS change, therefore, in response to the SRES-based climate change scenarios allowing new values of wheat yields and grassland intensity to be recomputed for each grid cell as a function of the known new EnS class and its baseline yield or stocking intensity value. Regio (Eurostat, 2000) contains data for the EU15 only and not for Switzerland and Norway and so, the agricultural statistics for these countries were derived from

national statistical databases. Full descriptions of the derivation of these statistics are given in Ewert et al. (2005) and Rounsevell et al. (2005). Individually, and collectively, these metrics are considered indices of anthropogenic pressure, i.e., changes in land use increasing the conflict between human activities and biodiversity attributes.

2.2. Mapping biodiversity value

Data for biodiversity include all known European species of breeding birds (Hagemeijer and Blair, 1997), mammals (Mitchell-Jones et al., 1999), herptiles (Gasc et al., 1997), and c. 20% of the European vascular flora (Jalas and Suominen, 1972–1996; Lahti and Lampinen, 1999). Details of the grid and data conversion are provided elsewhere (Williams et al., 2000). Biodiversity value can be measured in multiple ways. Here it was assessed as the relative contribution of European grid cells to achieving a representation goal for each taxonomic group considered; this is conceptually equivalent to measuring the option and bequest values of biodiversity (e.g. Humphries et al., 1995). In practice, this strategy amounted to selecting sets of complementarity areas (defined as a

property of sets of objects that exists when at least some of the objects [species] in one set [areas] differ from the objects [species] in another set [areas], Williams, 2001) that represented as many species as possible in a limited number of areas (also known as maximum coverage solution, Church et al., 1996). Here, 173 complementarity areas (~10% of the grid cells) were selected for each of the four taxonomic groups (Fig. 2). The number of areas selected is arbitrary, but follows the IUCN recommendation for countries to establish minimum conservation areas up to 10% of their total area (Soule and Sanjayan, 1998).

Complementarity areas were selected using the progressive rarity algorithm proposed by Margules et al. (1988). We used a heuristic technique adapted from the near-minimum-set algorithm of Margules et al. (1988). The algorithm starts by first selecting all areas with taxa that are equally or more restricted than the representation goal. For example, for a goal of representing each species at least once, it begins by selecting all areas that have species recorded in only one grid cell. Second, the algorithm follows a simple set of rules, applied iteratively to select areas richest in the rarest taxa. Initially, it selects grid cells with the greatest complementary

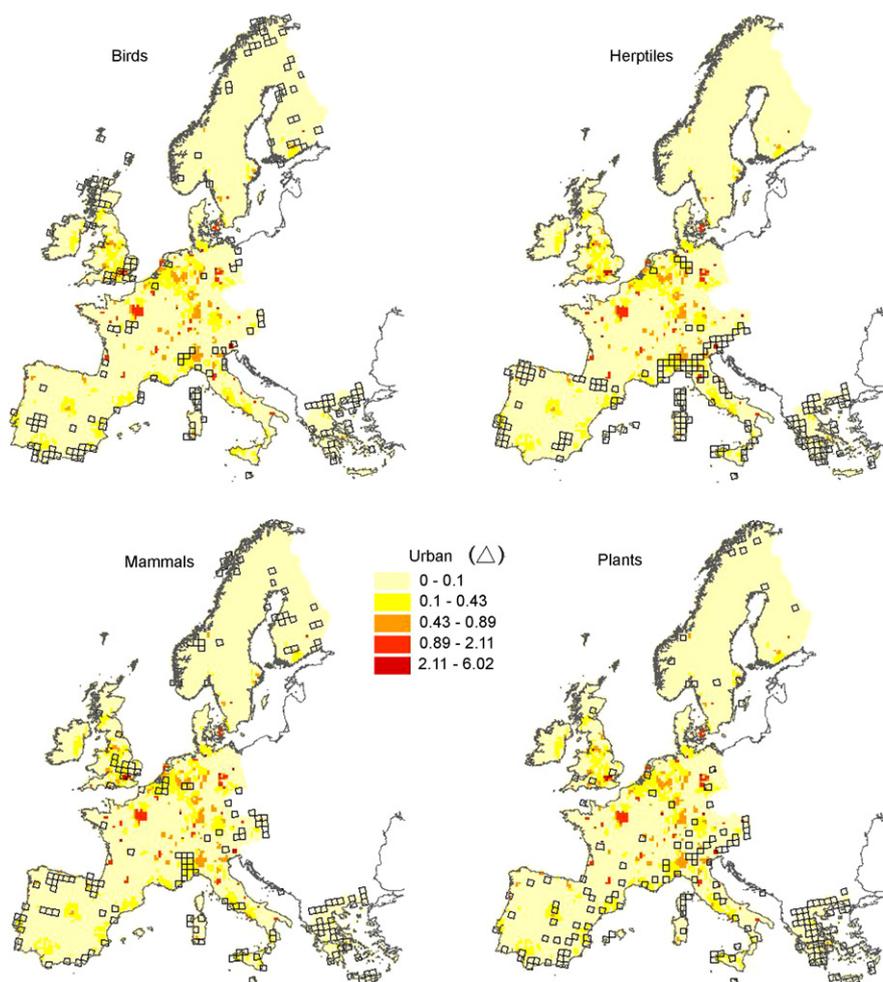


Fig. 2 – Complementarity areas for plants, breeding birds, mammals and combined amphibians and reptiles using a 50 km grid resolution. Complementarity areas overlapped on a map of urban land use change intensity (A1FI scenario) for 2020–2050. The complementarity-area solutions depicted on the maps are, for each taxon, just one out of 10 identified for this study (see Section 2).

richness in just the rarest taxa (ignoring other taxa). If there are ties, it proceeds by selecting areas among ties that are richest in the next rarest taxa. If there are still ties, it then selects those areas among ties with the lowest grid cell number (this is an arbitrary rule used in place of random choice among ties in order to ensure repeatability in tests). Finally these steps are repeated as necessary until the representation goal is achieved. A test is performed to reject any grid cell that in hindsight is redundant to the selected goal. Here, a modified version of this algorithm was used to provide an approximation to a maximum coverage problem (i.e. selecting n complementarity areas that maximise the representation of species). For this purpose, we repeated the process until the required number of areas was attained or exceeded. A final re-ordering of areas by complementary richness was made to provide an approximate solution to the maximum coverage problem (Williams et al., 2000).

As ties occur within the selection process, there are a number of alternative area selection solutions that would be equally effective in reaching the conservation goal (e.g. Hopkinson et al., 2001). To assess the sensitivity of our results to alternative solutions we selected 10 sets of complementarity areas for each one of the four taxonomic groups considered (by breaking ties at random instead of choosing areas with the lowest grid cell number—see above), i.e., we computed 40 solutions overall, and calculated the median anthropogenic pressure values. The median was used because it is less sensitive to outliers, or extreme values, than the mean (see also O’Dea et al., 2006).

All reserve selection analysis were performed using WORLDMAP (Williams, 1999).

2.3. Measuring human-induced pressures on biodiversity

In order to translate land uses into measures of pressure for biodiversity three indicators were used: (1) degree of urbanisation; (2) degree of cropland intensification; and (3) degree of intensity of grassland use. Changes in pressure between the two periods were measured as delta pressure (pressure in time t_2 – pressure in time t_1) recorded within selected complementarity areas. Differences between pressures recorded in the baseline and the different scenarios were tested with Friedman-rank tests for related samples. Dunnett t-tests were performed for ‘post hoc’ multiple comparisons. All statistical tests were carried out using SPSS.

3. Results

Levels of anthropogenic pressure recorded within complementarity areas were significantly different (Friedman test, $P < 0.001$) between the baseline and scenarios (Chi-square urban = 135.975; cropland = 42.625; grassland = 89.684). Urbanisation was projected to increase within the complementarity areas selected for all taxonomic groups (Fig. 1, Table 1), but this increase was significantly different from baseline and for all taxonomic groups only in the B2 world (regional environmental). Increased pressure from urbanization was less noticeable in the B1 world (global environmental). Cropland intensity was projected to decrease within complementarity areas selected for most taxonomic groups; except for birds, where both B-scenarios recorded slight expansions (not significant) of agricultural practices (Table 1). There was a consistent pattern of reduction of grasslands in the A2

Table 1 – Estimated median levels of anthropogenic pressure (urban land use [percentages], cropland yield [tonnes per ha] and grassland intensities [LSU per ha] within 50 km grid cells) and direction [+/-] of changes in the 10 complementarity-area solutions ($n = 173$) calculated for all four taxonomic groups considered within Europe 15 plus Norway and Switzerland

	Urban baseline	Urban A1FI	Urban A2	Urban B1	Urban B2
Plants	1.393	[+]1.460*	[+]1.471*	[+]1.438	[+]1.497**
Breeding birds	1.675	[+]1.735	[+]1.737	[+]1.711	[+]1.843*
Mammals	1.965	[+]2.030	[+]2.011	[+]2.023	[+]2.141**
Herptiles	1.354	[+]1.424	[+]1.444	[+]1.404	[+]1.565*
	Cropland baseline	Cropland A1FI	Cropland A2	Cropland B1	Cropland B2
Plants	3.016	[-]2.917**	[-]2.919**	[-]2.897**	[-]2.904**
Breeding birds	3.739	[-]3.724	[-]3.718	[+]3.744	[+]3.744
Mammals	3.461	[-]3.426	[-]3.404**	[-]3.430	[-]3.424
Herptiles	2.693	[-]2.560**	[-]2.566**	[-]2.521**	[-]2.563**
	Grassland baseline	Grassland A1FI	Grassland A2	Grassland B1	Grassland B2
Plants	0.721	[-]0.712**	[-]0.676**	[-]0.719	[-]0.711**
Breeding birds	0.718	[+]0.721	[-]0.679**	[+]0.737**	[+]0.730**
Mammals	0.741	[+]0.745	[-]0.717**	[+]0.755**	[+]0.753**
Herptiles	0.693	[+]0.696	[-]0.661	[-]0.679	[+]0.697

Five cases are compared: one baseline (1971–2000) and four scenarios (2021–2050, A1FI [global economic], A2 [regional economic], B1 [global environmental], and B2 [regional environmental]).

* $P < 0.05$ (two sided Dunnett t-tests treating baseline as control and comparing scenarios against it).

** $P < 0.01$ (two sided Dunnett t-tests treating baseline as control and comparing scenarios against it).

(regional economic) scenario, but this indicator of pressure was otherwise variable across taxa and the remaining scenarios (Table 1). For example, in the B1 and B2 worlds there was a significant increase in the area covered with grasslands within areas selected for birds and mammals and decreases in grassland for areas selected for herptiles in the B1 world; in contrast, grassland use intensity decreased within areas selected for plants in all scenarios.

4. Discussion

Our results illustrate some possible ‘what-if’ consequences for biodiversity of adopting different development pathways in the future. Pressures from urbanization are expected to increase in all scenarios because of the projected growth in human population and GDP (Fig. 1). However, a ‘green-world’ (B2) scenario, in which emphasis is given to local solutions for socio-economic problems, is projected to lead to the greatest increase in urban pressure due to the development of medium and small-sized human settlements. Cropland abandonment is projected to increase in all of our scenarios although the impacts are spatially diverse (Rounsevell et al., 2005; van Meijl et al., 2006); this is consistent with the post war trend in Western Europe (Rounsevell et al., 2002). For example, under scenarios where economies of proximity are prominent, agricultural changes within areas selected for birds are reduced; this is because selected areas for birds have, in comparison with other groups, a greater tendency to be located in Central Europe (Fig. 1 and Araújo et al., 2001), where agricultural yields are higher. Under these same scenarios, marginal land for cropland, such as that found in Southern Europe, is projected to witness a steady reduction in anthropogenic pressures, with potentially beneficial implications for those conservation areas that are located in the south (the case of herptiles, see Araújo et al., 2001).

It is important to note that the value of our estimates is mainly heuristic; they may help planners and decision-makers alike to formulate problems and choices on the basis of prospective impact studies, but do not provide predictions of future events. Furthermore, climate and land use scenarios are constantly being updated and the model outputs used in this paper will inevitably be challenged in the future. Nevertheless, there are important conclusions that can be drawn from this scenario-based study. The most important of which is that impacts of alternative development pathways on biodiversity are likely to be complex, and in cases may have apparently perverse outcomes. In other words, no single set of decisions should necessarily maximise benefit for all biodiversity in every region. If a fossil fuel intensive world of rapid economic growth, low population growth and rapid introduction of new and more efficient technologies (A1FI, global economic) comes about, this is likely to lead to sharper increases in temperature with greater projected climate-induced impacts on biodiversity (e.g. Berry et al., 2002; Thuiller et al., 2005; Araújo et al., 2006; Harrison et al., 2006). However, it may also have the consequence of reducing pressure from urbanisation in the wider countryside. In contrast, a world in which the emphasis is on local solutions to economic, social,

and environmental sustainability (B2, regional environmental) may drive human population density to increase and promote dispersed urbanisation. In this scenario, there is an increased pressure on species diversity at local level even though species are projected to be less affected by changes in the global climate system.

It is clear that purely climate-oriented assessments of the impacts of changes in greenhouse gas concentrations offer an incomplete picture of the likely impacts of global environmental changes on biodiversity (see also Sala et al., 2000; Pyke, 2004; Berry et al., 2006). However, combining scenarios of land use with climate change is not straightforward when the goal is to assess synergetic effects on biodiversity (but see del Barrio et al., 2006). In the present paper, we have developed a simple approach whereby the magnitude of land use changes within a number of important areas [complementarity areas] for biodiversity was measured. This is a suitable exploratory approach but one has to bear in mind that important areas for biodiversity are themselves likely to change their geographical position as a consequence of global environmental changes (Araújo et al., 2004; Pyke and Fisher, 2005; Williams et al., 2005). Given the coarse resolution of our data, this problem may not be as severe expected when dealing with grid cells of finer resolution. However, in the future, assessments of changes in human-biodiversity risks will benefit from considering at least two main improvements. First, they should attempt to incorporate modelled responses of species to climate and land use changes. There have been several attempts to model species responses to climate change (for reviews Pearson and Dawson, 2003; Guisan and Thuiller, 2005; Araújo and Guisan, 2006; Heikkinen et al., 2006), but the consequences of combining uncertainties related with the species responses to climate and land use change is still a matter of enquiry (e.g. Pearson et al., 2004; Thuiller et al., 2004; del Barrio et al., 2006; Luoto et al., 2006). Second, assessments should utilise probabilistic estimates of land use change. This is a potentially important innovation from previous scenario-based analysis, with early results from ensembles of model projections suggesting probable cascading effects on estimated species distributional shifts under environmental changes (see Araújo and New, 2007).

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- Miguel Araújo** is associate professor of research in the Spanish Research Council (CSIC). His research spans different issues of conservation biogeography, global change biology, and macroecology.
- David Nogués Bravo** is a postdoctoral fellow in the CSIC. His research focuses on the relationships between climate, species distributions and human impacts at large scales.
- Isabelle Reginster** is senior researcher in the University of Louvain. Her research includes the development of future land use change scenarios.
- Mark Rounsevell** is a Professor of Geography in the University of Edinburgh. His research includes the development of future land use change scenarios.
- Robert Whittaker** is Professor of Biogeography in the University of Oxford. His research includes conservation biogeography, large-scale patterns of diversity and island biogeography.