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Potential Impacts of Climate Change on Ecosystem Services in Europe: The Case of Pest Control by Vertebrates

EMILIO CIVANTOS, WILFRIED THUILLER, LUIGI MAIORANO, ANTOINE GUISAN, AND MIGUEL B. ARAÚJO

Global environmental changes threaten ecosystems and cause significant alterations to the supply of ecosystem services that are vital for human well-being. We provide an assessment of the potential impacts of climate change on the European diversity of vertebrates and their associated pest-control services. We modeled the distributions of the species that provide these services using forecasts from bioclimatic envelope models and then used the results to generate maps of potential species richness among vertebrate providers of pest-control services. We assessed how the potential richness of pest-control providers would change according to different climate and greenhouse emissions scenarios. We found that the potential richness of pest-control providers was likely to face substantial reductions, especially in southern European countries whose economies were highly dependent on agricultural yields. In much of central and northern Europe, where countries' economies were less dependent on agriculture, climate change was likely to benefit pest-control providers.

Keywords: biodiversity, ensemble forecasting, service-providing units, species distribution models

Ecosystem services are the benefits that humans derive from ecosystems. They include provisioning (e.g., food, timber, firewood), regulatory (e.g., climate regulation, water purification, pollination, pest control), supporting (e.g., soil retention and formation), and cultural (e.g., the use of natural areas for recreation) services. The services provided by ecosystems indirectly sustain humankind and directly support more than one billion people worldwide (Costanza et al. 1997, MA 2005, Turner et al. 2007). Biodiversity influences ecosystem services, because biodiversity has well-established effects on a number of ecosystem services mediated by ecosystem processes (Díaz et al. 2006). Biodiversity conservation is therefore part of the equation for sustaining human livelihoods (MA 2005, Naughton-Treves et al. 2005).

Global environmental changes threaten many ecosystems and are predicted to cause significant alterations to the supply of ecosystem services that are vital for human well-being (Schröter et al. 2005). A primary reason for concern over the current accelerated loss of species is the associated loss of ecological functions. Despite the link between ecosystem services and biodiversity conservation, it is still unclear how different aspects of biodiversity relate to ecosystem services (de Bello et al. 2010).

In the present article, we provide an assessment of the potential impacts of climate change on European vertebrates and on the ecosystem services they provide. A first step toward this goal is the identification of *ecosystem service providers* (Kremen 2005, Luck et al. 2009)—biodiversity components that are linked to and provide a specific ecosystem service. Service provision by multispecies functional groups has received particular attention (e.g., Kremen et al. 2002, Díaz et al. 2007) and was explored in detail by Kremen (2005). For the sake of simplicity, we focused on a single service: vertebrates' biological control of invertebrate and rodent pests in agricultural ecosystems. Although invertebrates such as spiders, ladybugs, mantis, flies, and wasps also provide an important natural control of pests (Naylor and Ehrlich 1997), we restricted our focus to pest-control services provided by vertebrates because comprehensive, spatially explicit data are lacking for most invertebrate groups. A second step involves modeling the potential impacts of climate change on the potential distributions

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of the vertebrate species that were identified as providing valued ecosystem services. The distributions of the vertebrate species providing pest-control services were modeled using ensemble forecasting of bioclimatic envelope models (Araújo and New 2007). Finally, modeled distributions of the target species were stacked in order to generate maps of potential species richness among service providers both in the baseline period and in the future.

Why natural pest control?

An important ecosystem service provided by biodiversity is natural pest control (Wilby and Thomas 2002). This is also one of the ecosystem services threatened by human activities (Naylor and Ehrlich 1997). Species competing with humans for food, timber, cotton, and other fibers are usually named *pests*, and they include a large range of herbivorous insects, rodents, fungi, snails, and nematodes. Pest species destroy 25%–50% of the world's crops, either before or after those crops are harvested (Pimentel et al. 1989).

The amount spent globally on pesticides is perhaps the best indicator of how much we value pest control: World pesticide expenditure reaches more than $30 billion annually, with insecticides alone accounting for nearly one-third of the total amount (Kiely et al. 2004). Naylor and Ehrlich (1997) added the cost of plant resistance-breeding programs to the amount spent on pesticides in their estimate, arguing that these costs serve as a proxy for the value of services formerly provided by natural enemies, which would result in a lower bound for the value of natural pest control of $54 billion. Furthermore, pesticide usage is expected to triple by 2050 (Tilman et al. 2001). Arthropod pests constitute a serious threat to food sustainability and the economy, annually destroying an estimated 37% of potential crops in the United States, even with pesticide use (Pimentel et al. 1992). Chemical pesticides, and the strategies by which they are applied to fight crop pests, can have harmful unintended consequences. Pests can develop resistance, which means that higher doses of pesticides must be applied or new chemicals must be periodically developed in order to achieve the same level of control (Naylor and Ehrlich 1997).

Resistance is now found in more than 500 insect and mite pests, over 100 weeds, and in about 150 plant pathogens (WRI 1994). Populations of the natural enemies of pests are also being decimated by heavy use of pesticides. Natural predators are often more susceptible to synthetic poisons than are the pests, because they do not have the same evolutionary experience with overcoming plant chemicals that pests have. Natural predators also have smaller population sizes than their prey, which makes them even more susceptible to the effects of mortality caused by chemicals (Abrams 2000). The result is that destruction of nonnatural predator populations leads to blooms in prey numbers, not only freeing target pests from natural controls but often promoting nonpest species to pest status. In California in the 1970s, for instance, 24 of the 25 most important agricultural pests had been elevated to that status by the overuse of pesticides (NRC 1989). Finally, exposure to pesticides and herbicides poses serious health risks to humans and many other types of organisms (Pimentel et al. 1992, Bassil et al. 2007).

Fortunately, approximately 99% of potential crop pests are controlled by natural enemies, including vertebrates, spiders, parasitic wasps and flies, lady bugs, and numerous other types of organisms (DeBach 1974). These natural biological control agents save farmers billions of dollars annually by protecting crops and reducing the need for chemical control (Naylor and Ehrlich 1997). However, in only a handful of studies have the socioeconomic impacts of the richness of the biological control agents been explored, and even then, they have been studied only indirectly. Most examples derive from agricultural ecosystems and are focused on the effects of predator richness on the biological control of insect pests—currently valued at $4.49 billion per year in the United States (Losey and Vaughan 2006) and more than $400 billion per year globally (Costanza et al. 1997).

Mapping changes in ecosystem service provision

In this study, the first step for mapping changes in ecosystem services was to identify the species that provide pest control. The second step was to model the distributions of those species in relation to different climate models and emissions scenarios. Finally, the third step was to investigate geographical patterns of change in aggregated measures (i.e., richness) of species.

Identification of ecosystem service providers. The species pool we considered included all terrestrial vertebrate species occurring in Europe, including 187 mammals (Mitchell-Jones et al. 1999), 445 breeding birds (Hagemeijer and Blair 1997), and 149 amphibians and reptiles (Gasc et al. 1997). These data have been extensively used in biogeographical and climate change studies (e.g., Araújo et al. 2006, 2011, Thuiller et al. 2011). We conducted an extensive literature review to select and classify species according to their functional contributions to the biological control of invertebrate and rodent pests in agricultural ecosystems (see supplemental table S1, available online at *http://dx.doi.org/10.1525/bio.2012.62.7.8*). Assignment of species to the invertebrate-pest-control or rodent-pest-control groups was determined using collected life-history traits and ecological indicators, such as feeding habitats and behavior, and their functional effect on the ecosystem (in this study, the impact of food habits on their ecosystem). The analyses were restricted to species that live in and obtain their resources from agricultural habitats. The selected species for each group were, therefore, species that feed mainly on agricultural land and whose prey species are either invertebrate or rodent species. Species sharing a particular trait linked to a given ecosystem service are referred to as *functional groups*. A functional group contributing to a given service provision is referred to as *service-providing unit* (SPU; Luck et al. 2009).

Overall, 110 species of European terrestrial vertebrates were grouped into two SPUs. The first SPU included

Table 1. European terrestrial vertebrates that provide pest control on invertebrates.

Mammals	Amphibians	Reptiles		Birds	
Erinaceus concolor	*Alytes obstetricans*	*Anguis fragilis*	*Bubulcus ibis*	*Upupa epops*	*Turdus viscivorus*
Erinaceus europaeus	*Alytes cisternasii*	*Chamaeleo chamaeleon*	*Falco naumanni*	*Melanocorypha calandra*	*Parus major*
Sorex araneus	*Bombina bombina*	*Hemidactylus turcicus*	*Falco subbuteo*	*Calandrella brachydactyla*	*Lanius collurio*
Sorex coronatus	*Pelobates syriacus*	*Phyllodactylus europaeus*	*Falco tinnunculus*	*Calandrella rufescens*	*Lanius minor*
Crocidura russula	*Pelobates cultripes*	*Tarentola mauritanica*	*Falco vespertinus*	*Galerida cristata*	*Lanius nubicus*
Crocidura suaveolens	*Pelobates fuscus*	*Lacerta vivipara*	*Burhinus oedicnemus*	*Galerida theklae*	*Lanius senator*
Suncus etruscus	*Pelodytes punctatus*	*Lacerta lepida*	*Glareola pratincola*	*Lullula arborea*	*Lanius excubitor*
Talpa europaea	*Bufo viridis*	*Lacerta agilis*	*Vanellus vanellus*	*Alauda arvensis*	*Pica pica*
Talpa occidentalis	*Bufo bufo*	*Podarcis hispanica*	*Cuculus canorus*	*Eremophila alpestris*	*Corvus corone*
Talpa romana	*Bufo calamita*	*Psammodromus algirus*	*Otus scops*	*Hirundo daurica*	*Corvus frugilegus*
Glis glis	*Rana perezi*	*Coluber viridiflavus*	*Athene noctua*	*Delichon urbica*	*Corvus monedula*
Mustela nivalis	*Rana arvalis*	*Coluber gemonensis*	*Caprimulgus europaeus*	*Anthus pratensis*	*Sturnus unicolor*
	Rana lessonae	*Coluber hippocrepis*	*Caprimulgus ruficollis*	*Oenanthe oenanthe*	*Sturnus vulgaris*
		Elaphe scalaris	*Apus apus*	*Saxicola rubetra*	*Emberiza citrinella*
		Elaphe longissima	*Apus melba*	*Saxicola torquata*	*Emberiza hortulana*
		Elaphe situla	*Apus pallidus*	*Turdus merula*	*Miliaria calandra*
		Malpolon monspessulanus	*Merops apiaster*	*Turdus pilaris*	

Table 2. European terrestrial vertebrates that provide pest control on rodents.

Mammals	Reptiles	Birds	
Erinaceus concolor	*Lacerta lepida*	*Ciconia ciconia*	*Buteo rufinus*
Erinaceus europaeus	*Coluber viridiflavus*	*Elanus caeruleus*	*Tyto alba*
Canis aureus	*Coluber gemonensis*	*Circus cyaneus*	*Otus scops*
Vulpes vulpes	*Coluber hippocrepis*	*Circus pygargus*	*Athene noctua*
Mustela nivalis	*Elaphe scalaris*	*Falco naumanni*	*Asio flammeus*
Mustela putorius	*Elaphe longissima*	*Falco tinnunculus*	*Lanius collurio*
Martes martes	*Elaphe situla*	*Milvus milvus*	*Lanius excubitor*
Meles meles	*Malpolon monspessulanus*	*Milvus migrans*	*Lanius senator*
	Vipera aspis	*Buteo buteo*	*Corvus corone*
	Vipera latastei	*Buteo lagopus*	*Corvus corax*

92 terrestrial vertebrate species that provide biological control on invertebrates (*SPUinv*; table 1). The second SPU included 38 terrestrial vertebrate species that provide biological control on rodents (*SPUrod*; table 2). Some species belonged to both SPUs.

Modeling changes in the distribution of ecosystem service providers. After the species were classified into an SPU, their distributions were modeled using bioclimatic envelope models. Several techniques exist to fit bioclimatic envelope models (e.g., Segurado and Araújo 2004, Elith et al. 2006). Projections often vary significantly (e.g., Araújo et al. 2006, Diniz-Filho et al. 2009), so we implemented models within the R-based BioMod package (*https://r-forge.r-project.org/R/?group_id=302*) for ensemble forecasting (Thuiller et al. 2009) and generated consensus projections by averaging them across all models. Specifically, we used seven alternative models, including two regression methods (the generalized additive model and the generalized linear model), two machine-learning methods (artificial neural networks and boosted regression trees), two classification methods (classification tree analysis and mixture discriminant analysis), and one surface range envelope. To convert projected probabilities of occurrence into presences and absences, we used *prevalence* (the ratio between the number of records and the total number of grid cells on the map—here, 2362 cells) as a threshold; this approach has been shown to be as good as the more familiar accuracy-based thresholds method (Liu et al. 2005). More details about the parameterization of the models are provided in Araújo and colleagues (2011) and Thuiller and colleagues (2011) and in the supplemental material of the present article.

Consensus projections were generated for three 30-year periods of time ending in 2020, 2050, and 2080 and using three general climatic models (GCMs), and four greenhouse emission scenarios (GESs; Nakicenovic et al. 2000). These GESs were chosen to capture a range of levels of greenhouse gas emissions predicted for the twenty-first century, including a pessimistic GES, two intermediate GESs, and a more moderate GES. The combinations of GCMs and GESs cover

a representative range of potential future climate conditions (Schröter et al. 2005).

Geographical patterns of change in service-providing units. After consensus projections from bioclimatic envelope models were obtained for every modeled species, we overlaid the potential distributions of the species with respect to SPU, time period, GCM, and GES. The resulting maps provide a spatial representation of the potential species richness within each pest-control SPU. The assumption was that the higher the number of species was for a given SPU, the greater was the expected natural control of pests. Our assumption was based on studies that showed that a more diverse natural-enemy community can enhance or potentially control a greater richness of pests on diverse crops (e.g., Cardinale et al. 2003, Perfecto et al. 2004). The difference between the projected future richness and the baseline richness was mapped and regressed against geographical coordinates (longitude and latitude) to assess geographical trends in the models.

Overall, the current modeled species richness was significantly lower than the projected richness, both in SPUinv and in SPUrod, for all future time periods and scenarios (table 2 and supplemental table S3). However, this relationship was not constant across geographical space, time periods, and emissions scenarios (figures 1 and 2; see figures for the remaining models in the supplemental material). The differences in species richness in both SPUinv and SPUrod were positively and significantly correlated with latitude, with the projections for 2020, 2050, and 2080 and the four GESs following the same pattern (figure 3 and figure 4; see all figures for the remaining models in the supplemental material). For all GESs, there were clear gains in the numbers of species in central and northern Europe and a loss of species in the southern European latitudes (figures 5 and 6; see the figures for the remaining models in the supplemental material). The differences in projected species richness between the future and the baseline period were higher for the pessimistic GES scenario and for 2050 and 2080 than those for the other levels of GES and for 2020. For 2080, latitudinal patterns in the differences in species richness were more marked for lower latitudes, for which there was a substantial loss of species in the Mediterranean countries.

Conclusions

Climatic conditions are known to determine the geographical distributions of species at broad spatial scales (see Thomas 2010 for a review). In this study, we used species–climate associations to assess the potential impact of climate change on vertebrate species that provide pest control in Europe. The major objective of our analyses was to assess how species range shifts and variation in species richness due to climate change could produce changes in the provision of an important ecosystem service in Europe: the natural biological control of invertebrates (SPUinv) and rodents (SPUrod) in agricultural lands.

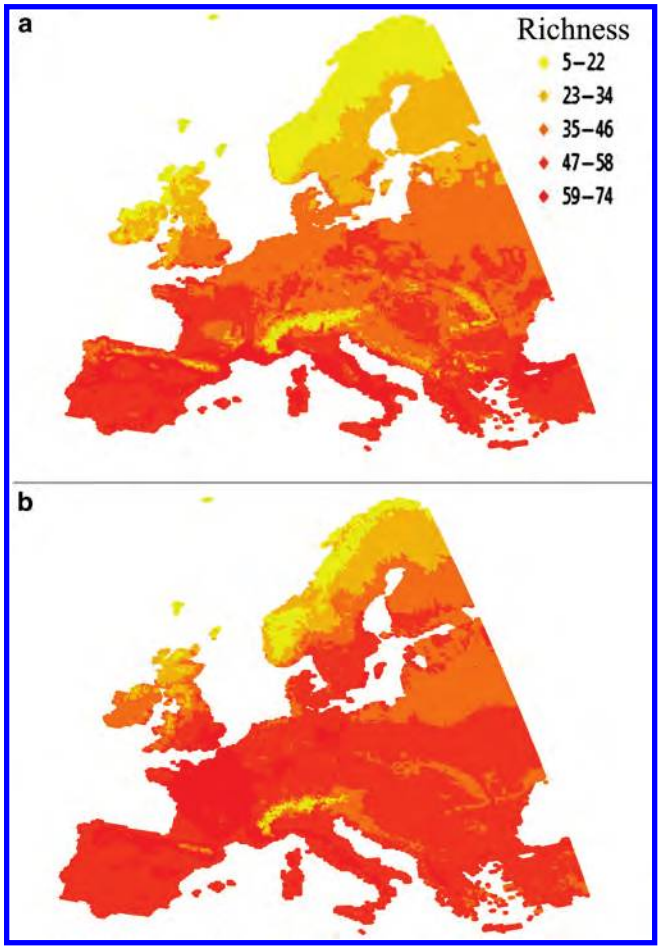

Figure 1. Projected variation in the total number of species (richness) providing pest control on invertebrates for the (a) baseline period and (b) future (2080) for the most pessimistic greenhouse gas emission scenario.

We found that the impacts of climate change on the two SPUs were similar for all scenarios. For both SPUs, there was a clear tendency for increased numbers of species in central and northern Europe and a loss of species in southern Europe, particularly in the Mediterranean countries. Because the results were qualitatively similar for all GESs and time periods, our discussion is focused on one of each of them: the pessimistic GES and 2080 (see the tables and figures for the remaining models in the supplemental material).

When climate change impacts on species providing pest-control services were compared across Europe, the Mediterranean region appeared most vulnerable. More precisely, the Iberian Peninsula (except the northern mountain ranges), southern France, Italy, and the eastern Mediterranean countries could suffer a substantial loss of richness among vertebrate species providing natural pest control for invertebrates and rodents. Potential biodiversity losses from climate changes in the Mediterranean have been reported for plant assemblages (e.g., Thuiller et al. 2005), amphibians and reptiles (e.g., Araújo et al. 2006), birds

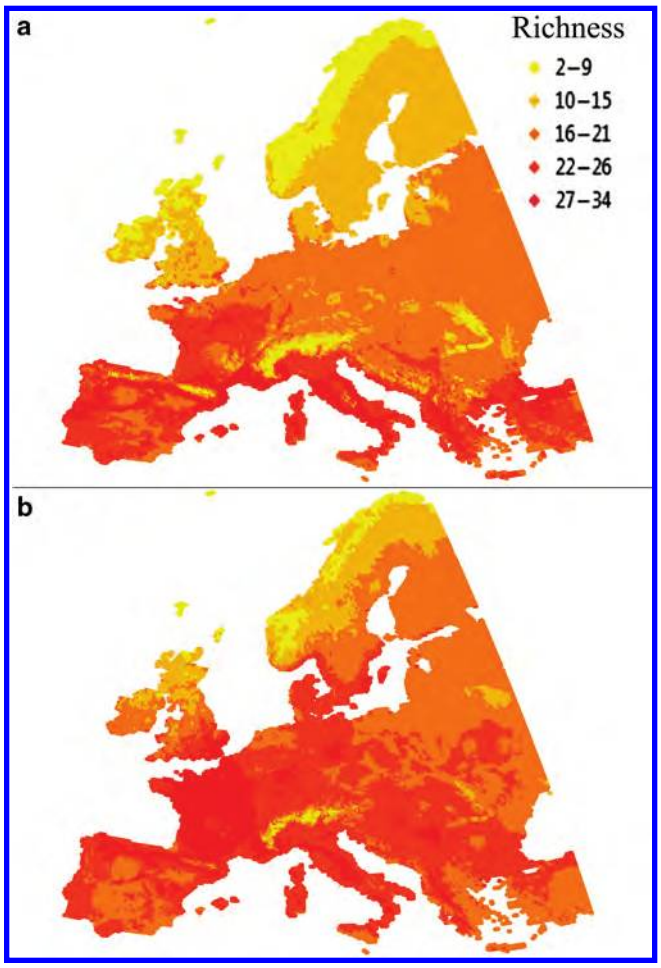

Figure 2. Projected variation in the total number of species (richness) providing pest control on rodents for the (a) baseline period and (b) future (2080) for the most pessimistic greenhouse gas emission scenario.

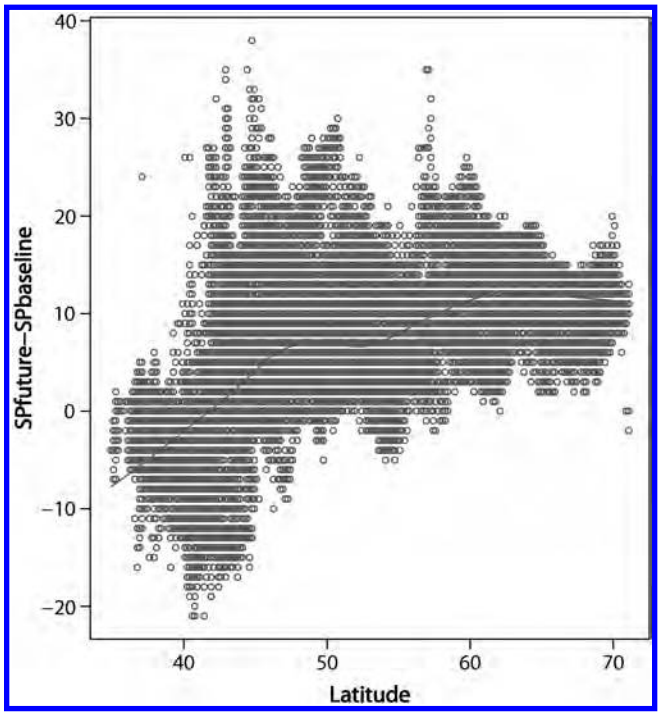

Figure 3. Differences in species richness of European terrestrial vertebrates that provide pest control on invertebrate between projected future (2080) richness (SPfuture) and baseline richness (SPbaseline) for the most pessimistic greenhouse gas emission scenario and latitudinal correlates of these differences.

(e.g., Huntley et al. 2008), and mammals (e.g., Maiorano et al. 2011). These losses are often associated with increased temperature and drought, which may push many species to environmental conditions that are beyond their critical tolerances. Simultaneously, warming in the cooler ranges of species in high latitudes and altitudes creates opportunities for colonization for some of these species.

Selected pest-control species in mountain regions also appear to be disproportionately affected by climate change—a pattern repeatedly shown for several groups of organisms (e.g., Araújo et al. 2011, Thuiller et al. 2011). In contrast, gains in potential species richness are projected to occur mainly in northern Europe (Denmark and southern Sweden), northern and central France, and the mountain ranges of the Alps and the Apennines. It is possible that species from regions around the Mediterranean will gain new climatically suitable areas in northern ecosystems as climate changes (Thuiller et al. 2005), although whether they would be able to colonize them as they become climatically suitable is uncertain. Rapid changes in species richness could have significant structural and functional effects on ecosystems. Reductions in vertebrate abundance and species richness can be expected to decrease natural pest control in agricultural crops (Cardinale et al. 2003, Perfecto et al. 2004). This is problematic because natural pest-control services are becoming increasingly important as invertebrate pests develop resistance to chemicals and as pesticide use is curbed by environmental regulations and consumer trends (Naylor and Ehrlich 1997).

Several previous studies have shown, for example, that birds can depress the abundance of at least some herbivorous insects. Atlegrim (1989) documented the effect of birds on plant performance: Leaf damage to the common bilberry (*Vaccinium myrtillus*) increased significantly in their absence. Perfecto and colleagues (2004) found that a high density and richness of birds can potentially prevent pest outbreaks in coffee farms. Perhaps the most noticeable example, however, comes from Dutch apple orchards: Mols and Visser (2007) reported that the emplacement of nesting boxes to attract great tits (*Parus major*) reduced the number of caterpillars and the amount of fruit damage and increased fruit yield. The increase in yield was striking, from 4.7 to 7.8 kilograms of apples per tree—a yield increase of 66%.

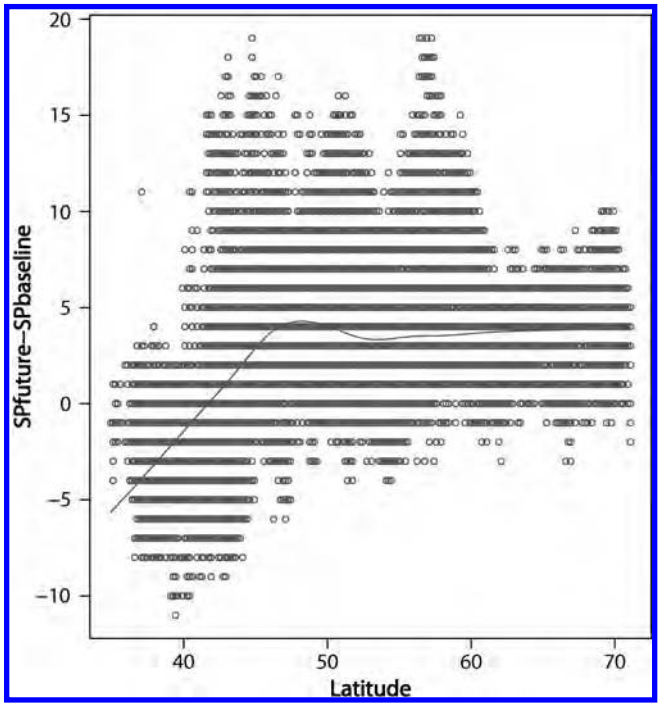

Figure 4. Differences in species richness of European terrestrial vertebrates that provide pest control on rodents between projected future (2080) richness (SPfuture) and baseline richness (SPbaseline) for the most pessimistic greenhouse gas emission scenario and latitudinal correlates of these differences.

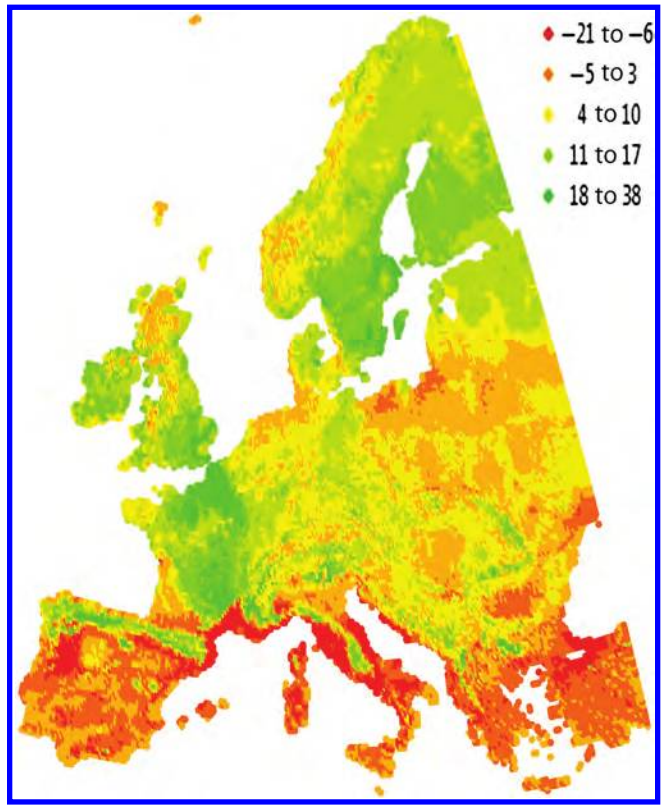

Figure 5. Gains and losses of species providing pest control on invertebrates for 2080 (under the pessimistic scenario) in respect to current conditions. Red represents losses, yellow represents a stable population, and green represents gains.

In few studies have the effects of rodents on agricultural production been assessed (e.g., Brown et al. 2007). Nonetheless, there are examples of rodent species' having strong effects on agricultural crops (Brown et al. 2007), natural plant communities (Ostfeld and Canham 1993), and newly established synthetic prairie gardens (Howe et al. 2002). Given the preponderance of rodents in the diets of many raptors (e.g., hawks and owls), it seems reasonable to assume that these birds benefit agriculture. Moreover, several raptor species readily occur in agricultural landscapes. Most investigations of predation by raptors on rodents are focused on the predators' potential role in cyclic population dynamics. From these studies, we know much about the predator–prey interactions of many raptor species and many rodent species (Korpimäki and Norrdahl 1991). Experimental studies have suggested that predators can drive cyclic population change of microtines because predator exclusion can reverse the decline phase in the microtine population cycle (Korpimäki and Norrdahl 1998). However, other studies have shown that predation can be a major cause of mortality of voles during the decline phase of their cycle (Norrdahl and Korpimäki 1995).

Avian and terrestrial predators of rodents may facilitate each other. Kotler and colleagues (1992) demonstrated predator facilitation between owls and desert diadema snakes (*Spalerosophis diadema*) feeding on gerbils (*Gerbillus allenbyi* and *Gerbillus pyramidum*): Owls drive gerbils to cover, the preferred habitat of the snake. Such effects lead to predation on rodents even though the raptor itself is not directly responsible for it. Similarly, Korpimäki and colleagues (1996) found predator facilitation between the least weasel (*Mustela nivalis*) and Eurasian kestrel (*Falco tinnunculus*), suggesting that "the assemblage of predators subsisting on rodent prey may contribute to the crash of the four-year vole cycle" (p. 30). To paraphrase Kotler and colleagues (1992), rodents shifting habitat to avoid an owl may wind up in the fangs of a snake (and vice versa).

Our results cannot be taken as precise forecasts for at least four important reasons. First, there are several uncertainties in climate change scenarios (Weaver and Zwiers 2000) and in the bioclimatic modeling itself (e.g., Thuiller et al. 2004, Araújo et al. 2005, Nenzém and Araújo 2011). These uncertainties can be somewhat mitigated by the use of ensemble forecasting, but the value of this approach is still contingent on model inputs, and these have known limitations. Second, projections from bioclimatic-envelope models estimate how the distribution and abundance of suitable environments for species will change in time, but they do not provide population-level estimates of persistence or extinction (e.g., Anderson et al. 2009), which may

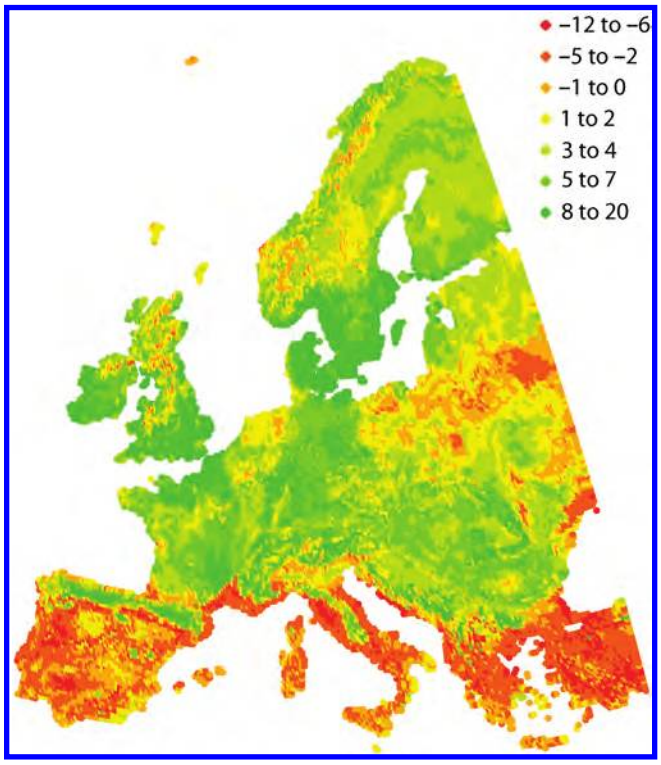

Figure 6. Gains and losses of species providing pest control on rodents for 2080 (under the pessimistic scenario) in respect to current conditions. Red represents losses, yellow represents a stable population, and green represents gains.

or may not linearly covary with changes in environmental suitability (e.g., Araújo et al. 2005). Third, the relatively coarse grid scale of our study may hide potential refuges for species and environmental heterogeneity that could enhance species survival, especially in mountain areas, where our estimation of changes could be overestimated (e.g., Randin et al. 2009). In particular, the extent of species losses may be overestimated because the plasticity of species and the survival of species in favorable microhabitats were not considered. Finally, the patterns of species richness and species losses at the southern European margins may be overestimated, because information on the distributions of species in northern Africa is missing, and there is the possibility that some southern European species are able to tolerate conditions that are more arid than those recorded in Europe and that some North African species will migrate northward, thus colonizing southern European ecosystems (Barbet-Massin et al. 2010, Maiorano et al. 2011). However, even if the numbers are overestimated, the patterns across the regions are likely to stand (e.g., the ranking of the regions in terms of their vulnerability to loss) and provide guidance in risk assessment.

Despite these uncertainties, our findings provide a first examination of the potential magnitude and the likely direction of climate change effects on this ecosystem service. The potential loss of richness of pest-control service providers could have important consequences in southern European countries with economies highly dependent on agricultural yields (EC 2010). The percentage of gross domestic product based on agriculture is generally higher in southern European countries (e.g., Spain, 2.7%; Greece, 3.3%; and Portugal, 2.4%) than in central Europe (e.g., Germany, 0.9%; Belgium, 0.7%; European Union average: 1.8%). In much of central and northern Europe, which has countries with economies that are relatively less dependent on agriculture, climate change is likely to benefit providers of pest-control services. However, any potential increases in natural pest-control services are also contingent on changes in land use and pollution that are not accounted for in our models. The societal importance of ecosystem services is often underappreciated and is only adequately valued when it is missing. However, the anticipation of changes improves our capacity to adapt and to make effective decisions. Therefore, it becomes increasingly urgent to improve our understanding of species and ecosystem responses to a changing environment.

Acknowledgments

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