Scaling in Ecology and Biodiversity Conservation

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Preface

Human actions, motivated by social and economic driving forces, generate various pressures on biodiversity, such as habitat loss and fragmentation, climate change, land use related disturbance patterns, or species invasions that have an impact on biodiversity from the genetic to the ecosystem level. Each of these factors acts at characteristic scales, and the scales of social and economic demands, of environmental pressures, of biodiversity impacts, of scientific analysis, and of governmental responses do not necessarily match. However, management of the living world will be effective only if we understand how problems and solutions change with scale.

SCALES (http://www.scales-project.net), a research project lasting for five years from May 2009 to July 2014, was seeking for ways to build the issue of scale into policy and decision-making and biodiversity management. It has greatly advanced our knowledge of how anthropogenic and natural processes interact across scales and affect biodiversity and it has evaluated in a very practical way how this knowledge can be used to improve the scale-sensitivity and effectiveness of policy instruments for conservation and sustainable use of biodiversity.

During the project we have especially emphasized approaches that utilize existing biodiversity databases as they are the most widely available information in applied biodiversity conservation. We also tried to integrate the most appropriate assessment tools and policy instruments into a coherent framework to support biodiversity conservation across spatial and temporal scales. While the guidelines, practical solutions and special tools are presented as a special web based portal at a central place, the SCALETOOL (http://scales.ckff.si/scaletool/), the scientific outcome is widely spread over the scientific literature in regional and international journals.

With the SCALES book we want to bundle the main results of SCALES in a comprehensive manner and present it in a way that is usable not only for pure scientists but also for people making decisions in administration, management, policy or even business and NGOs; to people who are more interested in the "practical" side of this issue.

Yrjö Haila, Tampere

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CHAPTER I

Introduction

Scaling in ecology and biodiversity conservation: An introduction

Klaus Henle, Vesna Grobelnik, Simon G. Potts, Anna V. Scott, William E. Kunin, Richard M. Gunton, Yiannis G. Matsinos, Jukka Similä, John D. Pantis, Reinhard Klenke, Josef Settele, Lyubomir Penev

Biological systems are complicated. To understand how our management of the environment affects them, we need to frame our questions carefully. We can capture some of that complexity by thinking of a range of different types of "scales" – spatial scales, temporal scales and scales of biological and human organisation. Biodiversity conservation and management will only be effective if we understand how problems and solutions depend on these scales. At one scale it may be changes in climate, while at others it may be habitat loss and fragmentation or disturbance that need to be addressed (Henle et al. 2010).

The natural environment is heterogeneous in space and time, and these variations matter greatly for the species of living things. Crucially, the variation occurs across a wide range of spatial scales: climates vary across global and continental scales (from tropical to polar), but also at much finer scales from the bottom to the top of a mountain range, or even between the south-facing and north-facing side of a boulder. Different disturbance events may affect those environments at different scales as well, from a single hoof-print to the ploughing of a field, the track of a hurricane or the shifting of agricultural practices across continents due to policy shifts. Indeed, the intensity and spatial heterogeneity of direct drivers of biodiversity, which are associated with land use or land cover, change strongly with scale (Tzanopoulos et al. 2013). Drivers also vary at all kinds of temporal resolutions; for example, temperatures and moisture change greatly over the seasons but they may also shift more quickly as weather fronts pass, or between day and night. Different organisms will be affected by these differences at different scales of space and time. Small annual plants and snails experience conditions within a few square centimetres over a few weeks, whereas large birds or large carnivores may forage over many square kilometres and multiple years. Other organisms have intermediate mobility and life spans and therefore integrate environmental conditions over intermediate scales (Figure 1).

Human societies are also organised along different scales. We have established different administrative levels from the municipality, through the state to countries, and, in Europe, the European Union, to address governance at appropriate scales in human societies. This applies also for biodiversity conservation, in which the EU sets the framework that Member States have to implement. However, the spatial scales of ecological processes and those of social organizations responsible for management of the processes do not necessarily match each other. In addition, drivers of biodiversity change may operate at multiple levels, which do not match the levels of administration. Furthermore, governance – a term broader than administration – is increasingly taking place through networks with no single centre, but several complementary and competing centres, which have different functions as well as means of power and influence. As a consequence of this, while exploring the relationship between ecological and human systems, we need to distinguish not only ecological scales but also governance scales and need to match them better.

While the issue of scales takes an increasingly prominent role in ecology (Schneider 2001), biodiversity measures and policies still often do not match relevant ecological scales and thus may be unsuccessful – or even may have negative effects (Cumming et al. 2006, Henle et al. 2010). This book provides an introduction to issues of scaling in ecology and biodiversity conservation and summarizes some of the key results of the project SCALES (Securing the Conservation of biodiversity across Administrative Levels and spatial, temporal, and Ecological Scales). SCALES was initiated to improve our understanding of the relevance of scale for biodiversity conservation and to reduce the mismatches in human responses to changing biodiversity. Much of the material in this book concerns issues of spatial scaling and how they can be addressed effectively at different administrative levels. We focus on

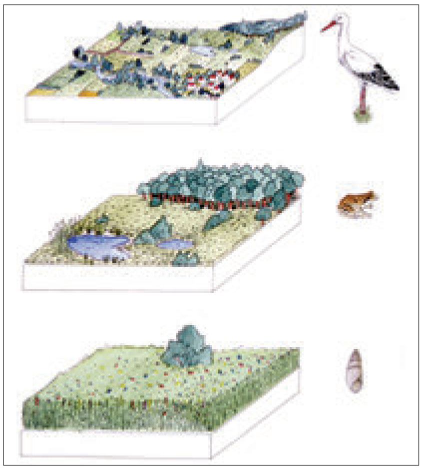

Figure 1. Difference of spatial scale at which species operate (Henle and Kaule 1991)

the relevance of scale for networks of protected areas, connectivity, and biodiversity monitoring and the key policies that address these issues. The book is divided into five main chapters: Scaling of anthropogenic and natural drivers of biodiversity; Scaling of biodiversity patterns and processes; Methods and tools; Scaling in policies and management; and Case studies and integration.

We begin with an overview of the notion of "scale" (Gunton et al. 2014a this book), which introduces key concepts to clarify the diverse uses of the word "scale" in what follows. Chapter I also explains what is typically meant by a number of scale-related terms, such as "scale-dependent", "scale-sensitive" and "scale-invariant", and verbs such as "upscaling" and "downscaling". In addition, a SCALES lexicon puts the key English terms in the context of corresponding terms in seven other European languages plus Chinese, which are represented among the SCALES project partners.

Scaling of anthropogenic and natural drivers of biodiversity

Chapter II focuses on the scale-sensitivity of drivers and the fundamental role they play in shaping patterns of biodiversity within the landscape. A driver of environmental change can be considered as any natural or human-induced instigator of functional or structural ecosystem change (Nelson et al. 2006). Drivers can have direct effects, e.g. land use, plant nutrients, diseases and climate, or an indirect (diffuse) effect, e.g. demographic, economic, socio-political and cultural drivers. The chapter starts with a discussion of the conceptual framework and typology of drivers (Marty et al. 2014 this book). Drivers can operate differently across different administrative and temporal scales and this can cause so-called "scale mismatches". Tzanopoulos et al. (2014 this book) show that direct drivers, such as deforestation, agricultural conversion and wetland loss, demonstrate the highest levels of scale sensitivity. They discuss the relevance of their findings for policy and biodiversity management. Touloumis and Pantis (2014a this book) and Scott et al. (2014 this book) explore the effects of these scale-sensitivities on the patterns and intensity of fragmentation and habitat loss. They show that these effects vary across scales, with significant implications for networks of protected sites, as exemplified by the European Natura 2000 system, where actual habitat losses and fragmentation can be higher than predicted at a regional or national scale.

Drivers can also be used for the prediction of future environmental changes. Lehsten and Scott (2014 this book) present a tool for modelling climate change, land use change and CO_2 increases until 2050 based upon existing environmental data and drivers. Such innovative methods help us to advance the incorporation of large-scale driver data into informative predictions that enable better decision-making across spatial and temporal scales, and administrative levels.

Scaling of biodiversity patterns and processes

Chapter III considers ecological scales. We examine four different scales of biological organisation: genes, populations, communities and ecosystems. All four of these levels can be important foci for conservation goals. Maintaining genetic richness within a species can be a goal in itself; moreover, the diversity of alleles within a population may circumscribe its ability to cope with current or future conditions. The importance of managing populations for conservation is more widely appreciated, especially when it involves high-profile "flagship" species, such as wolves or red kites. However, only a minority of the millions of species on Earth get that sort of attention. Conservation of these species depends on wider community-level biodiversity conservation: the preservation of whole sets of species that live together in a given environment. Beyond the conservation of biodiversity for its own sake, there is increasing interest in the services that it provides to our agriculture, environmental quality and well-being (Millennium Ecosystem Assessment 2005).

All four of these biological scales can be closely linked to certain spatial scales. The genetic diversity of a species at a coarse (national or global) scale might be considered in terms of the number of genetic variants (alleles) that its various populations contain across its whole range. Arenas et al. (2014 this book) show that the amount of genetic variability retained when species lose, expand, or shift their range strongly depends on the speed of these range changes and the mobility of the species. Populations must also be considered at multiple scales. For example, the dispersal power and the generation length of species differ by orders of magnitude among species. Besides genetic variability, dispersal among subpopulations and minimum area requirement are two key factors that determine the viability or extinction risk of populations. Pe'er et al. (2014a this book) show how life-history traits can be used to extrapolate minimum area requirements from case studies on particular species to unstudied species. They further discuss the implications for the conservation of multiple species.

Populations are typically aggregated in space across a wide range of scales. These spatially complex population patterns add up to produce the scaling of species richness at the community level. It has long been known that one tends to find fewer species in a small area than a larger one (because of smaller sample sizes, but also because of the effects of environmental variation and spatial distances), but the shape of this "species-area relationship" has been an issue of debate for decades. Storch et al. (2014 this book) provide an overview of recent progress in understanding the roles of spatial species turnover ("β diversity") in scaling biodiversity and Bommarco and Marini (2014 this book) explore how land use change influences species community change at different spatial scales.

The biodiversity of our ecosystems is of interest in itself, but there has also been increasing interest in recent years in the ecosystem services they provide to our economy and well-being. As with other aspects of biodiversity, these services too are scale-dependent. Steffan-Dewenter et al. (2014 this book) show that there is reason to think that biodiverse natural ecosystems may provide better services in some respects than would be provided by species-poor communities. Therefore, policy and management practices should consider the complex spatial and temporal interactions between semi-natural habitats, cropped areas and conservation interventions.

Conservation biology is full of trade-offs, as effective conservation requires both the extensive sampling/representation of natural diversity (which often requires widely scattered efforts) and the maintenance/preservation of such populations (which may require concentrating efforts in a few large sites). The best approaches to tackling these trade-offs seem likely to be different at different scales. Moreover, the ideal configurations of conservation effort for different aspects of biodiversity (genes, species, communities) and the services it provides may themselves be different. Ultimately, while these four processes are inter-related, the steps we need to take to preserve each may be rather different. The final paper of this chapter (Marsh et al. 2014 this book) turns to the issue of how those different goals may come together or conflict.

Methods and tools

The increasing recognition of the importance of scale-specific issues in biodiversity conservation has also created a demand for improved and novel methods. Chapter IV presents selected tools and methods that were developed or improved in the SCALES project to address scaling issues. Because of limited resources, biodiversity conservation has to set priorities. Schmeller et al. (2014 this book) illustrate a GIS-tool that facilitates the use of a method to assess the national responsibility for the conservation of species in Europe and elsewhere in the world and to use this information for setting priorities. Yu et al. (2014 this book) developed a GIS toolbox that implements advanced spatiotemporal analysis and mapping functions for environmental data in a geostatistical context and Gunton et al. (2014b this book) a method that allows downscaling of microclimatic data from coarse-grained meteorological data. These tools can greatly expand our ability to model current and predict future distributions of species under environmental change. For predicting likely impacts of climate change, it is also essential to assess connectivity. Pe'er et al. (2014b this book) provide a short introduction to methods for such assessment, stressing the importance to distinguish between structural and functional connectivity.

Scaling in policies and management

In Chapter V, we turn to policy issues. The seven essays of this chapter concern policy issues relevant both for protected areas and areas outside protected sites. The system of protected areas, often considered as the core of nature conservation policy, involves many issues, such as site-selection, protected area management and financing, which all have scale implications. The protected area policy in Europe aims to match ecological with societal scales in the sense that the current backbone of the system, namely Natura 2000, is based on European ecological needs, while national and local governments may establish additional protected areas based on national or local ecological needs.

Mathevet et al. (2014 this book) outline the experiences from the implementation of Natura 2000 and the significant changes of site-selection policy these experiences stimulated. Institutional structures for participation have been made stronger and new voluntary approaches for site-selection are emerging. In truly voluntary schemes, the landowners have power to make initiatives and final decisions, which may distort systematic selection of sites to be protected and, as a consequence, affect the effectiveness of biodiversity policy. Hence, a key challenge is to combine the strengths of voluntarism and systematic decision-making. Policy experiments on new forms of site-selection, for example in France, seek for solutions where large-scale ecological knowledge is used to support local-level decision-making. Along the same lines, Grodzinska-Jurczak et al. (2014 this book) argue that the governance of protected areas needs to be considered in relation to economic, political, cultural and historical contexts. Management should respect land use issues and nature conservation while assuring participation in decision-making, and just and fair distribution of conservation costs and benefits.

The benefits and costs of nature conservation are not equally shared. While benefits can be acknowledged at national or even global level, costs – negative economic consequences – are often carried at the level of a local economy. Santos et al. (2014 this book) advance ecological fiscal transfer (EFT) schemes as a potential solution to this problem. EFTs redistribute public revenues from national and regional governments to local governments on the basis of ecological or conservation based indicators. They can take into account spillover benefits and can also offset opportunity costs (e.g. resulting from land-use restrictions) and/or local public expenditure on conservation activities. This instrument is still new and adopted only in one EU Member State, namely Portugal.

The issue of nature conservation outside protected areas has become increasingly important because the space available for protected areas is limited. Green infrastructure policy, currently in development in the EU, aims to address this. Green infrastructure (GI) is the network of natural and semi-natural areas, features and green spaces in rural and urban, terrestrial, freshwater, coastal, and marine areas, which together enhance ecosystem health and resilience, contribute to biodiversity conservation and benefit human populations through the maintenance and enhancement of ecosystem services. Kettunen et al. (2014 this book) explore the scale related aspects and challenges of GI and draw preliminary conclusions on the integration of this concept into the EU 2020 biodiversity policy.

In human-dominated landscapes, as in most parts of Europe, biodiversity conservation requires appropriate management at the right scale. Lengyel et al. (2014 this book) analyse at which scales biodiversity management is carried out and to what extent this matches relevant ecological scales. They identify mismatches and develop recommendations for reducing them.

To assess the success of management and whether policies achieve their goals, monitoring of changes is essential. Kosztyi et al. (2014 this book) assess the effectiveness and limitations of monitoring programmes to detect the status and trends in biodiversity in light of the policy instruments that guide monitoring. They outline how recent scientific advances can be used to improve monitoring practices and provide recommendations for how the new methods can be integrated in national and European policy instruments that guide biodiversity monitoring. While goals are set at the EU level, the key level for actual monitoring is the national one, where monitoring is a combination of state- and NGO-funded schemes and carried out by varying proportions of volunteers and professionals. McConville et al. (2014 this book) evaluate to which extent the existing national monitoring institutions are capable of informing about the progress of achieving the EU 2020 targets.

Case studies and integration

In the sixth chapter of this book we turn our focus to some case-study countries. More specifically, our aim is to apply and integrate selected knowledge, tools, datasets and methods developed within the SCALES project, in order to test their applicability. The testing was carried out within five EU countries selected on the basis of biogeographic, socio-economic and conservation history criteria (Finland, France, Greece, Poland, UK) and in Taiwan.

To carry out assessments at larger spatial scales, issue of data availability and standardization become increasingly important. Differences in database definitions, structures, software and availability have made the process of data standardization a necessary action in order to create usable datasets. Touloumis and Pantis (2014b this book) assess the availability, quality, and suitability of ecological datasets in Europe for standardization and provide recommendations to facilitate the smooth dissemination and application of ecological data throughout the scientific community. Despite considerable biodiversity data being available, gaps in the knowledge about the distributions of species, which is essential for planning networks of protected areas and assessing connectivity, often still exist. Using an example from Taiwan, Lin et al. (2014 this book) outline an approach for optimizing data collection to reduce the effort required and how this helps to improve our understanding of the relationship of species to their environment.

Today, policy for establishing nature reserves is not oriented towards large, isolated areas but towards connecting sites into networks of protected areas. In this context, protected areas act as nodes in a web, among which a flow of individuals should be assured to minimize the risk of extinction for species. Additionally, in a constantly changing world, where climate and land-use transitions lead to corresponding alterations in species distributions and community structures, proper tools and methods to evaluate site connectivity in the future are of great importance. Papanikolaou et al. (2014 this book) present a methodological framework to evaluate the efficiency of a protected area network in view of global changes, by assessing connectivity at different scales. They show that climate change is likely to severely impact connectivity for grassland species.

Connectivity is not only important for networks of protected areas but also to set appropriate priorities for the conservation of species outside protected areas. Arponen et al. (2014 this book) illustrate this for the management of semi-natural grasslands within agri-environmental schemes in southern Finland. They argue that the implementation of agri-environmental schemes has not, until now, taken into account habitat connectivity, and as a result they have not been highly effective in biodiversity conservation.

A more general and comparative view of regional connectivity is provided by Klenke et al. (2014 this book). They combine results from previous papers with results of a telemetry study of brown bears. This paper also links the results of ecological research to EU policies and the social science based case studies that follow in this book chapter.

The perspectives of researchers and practitioners from different countries regarding the potential of policy instruments to improve ecological connectivity are of crucial importance. Through an empirical investigation of such perspectives, Paloniemi et al. (2014 this book) identify instruments that were evaluated positively by respondents, such as the establishment of conservation networks, and those that were criticized as not successful, such as the integration of connectivity measures into land use planning and development policies. This brings our attention to the fact that any effort to improve ecological connectivity could be futile unless integrated into a wider framework that smoothly and efficiently coordinates land-use, balancing conservation, social and economic factors.

By exploring the perspectives of stakeholders on legitimacy, Cent et al. (2014 this book) find that more attention should be given to both the formal and informal aspects of participation at all administrative levels. Most crucially, their empirical work revealed that issues of transparency, democracy and equity primarily explain the social construction of legitimacy, as expected from social science theories. However, current conservation policies ignore this in most cases and rather focus on ecological criteria in site selection and market-based instruments in supporting management practices.

Conclusions

As can be seen from the diversity of topics covered by the contributions to this book, scaling issues play an important role in conservation biology. The contributions also show that there are no simple solutions that cover all the different aspects that should be considered when attempting to secure the conservation of biodiversity at different ecological scales and administrative levels. Moreover, there are trade-offs between different goals in biodiversity conservation. The contributions to this book show promising directions for how such trade-offs could be accounted for by considering spatial and temporal scales. They also show promising directions for future research and for improving the scale-sensitivity of management and scale-effectiveness of biodiversity policies. To facilitate access to the recommendations for management and policy as well as to the databases and methodological tools developed in the SCALES project, a web-based resource, the SCALETOOL was created. Henle et al. (2014a this book) briefly outline the structure and main features of this resource.

In a concluding chapter, Henle et al. (2014b this book) summarize some of the key messages of the book in terms of ecological connectivity, multi-scale policy and the coherence of networks of protected areas. We hope that this book facilitates improving the conservation of biodiversity across all kinds of spatial, temporal, ecological and administrative scales.

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The meaning of "scale"

Richard M. Gunton, Reinhard A. Klenke, Riikka Paloniemi, Yoni Gavish, Charles J. Marsh, William E. Kunin, Klaus Henle

This chapter provides an overview of concepts of "scale" as used throughout this book, looking first at usage in English and then drawing upon some other languages represented in the SCALES project. It shows how "scale" is used primarily in a spatial sense, referring to various relationships among things of differing sizes. There are scales of patterns, processes, studies and analyses (Figure 1). There is also an analogous set of meanings about time: we may speak of short and long time-scales, the time-scale (period) of fluctuating signals, and fine and coarse time-scales of sampling, for example. We also consider the analogous use of "scale" to refer to a level of focus in some hierarchy of organisation.

What is scale?

"Scale" is an important concept in ecology and conservation because our science is concerned with abstract entities, such as populations, communities and ecosystems, that are localised in patchy, fluctuating environments spread across the globe and cannot directly be visualised. This requires ecologists to ask scale-based questions at the outset of any investigation, such as, "How big is this community?" or "How much of this ecosystem do I need to observe, and for how long, in order to make reliable inferences about it?" Ecological data are then collected, typically from sample units of some chosen size collected at points in space and time according to an arbitrary sampling protocol, which has some influence on the inferences that will be made and may even produce artefacts. "Scale" is also an important concept in conservation science and policy-making, because threats to our natural environment occur across the globe according to the spatio-temporal patterns of physical, biological and anthropogenic forces, while human governance is organised hierarchically by localities and has its own timeframes of response to perceived problems. Conservationists thus grapple with the challenges of making strategic, efficient interventions with limited resources, and monitoring their effects in cost-effective ways. All of these problems raise questions of scale.

"Scale" primarily relates to divisions of continuous space and time, but the word is also used in the context of hierarchies of discrete units of study. Ecologists may compare an analysis "at the scale of populations" with one "at the scale of species"; a field survey might have "scales" of fields, quadrats and individual point-samples (Figure 2), or of time-points and daily aggregated samples; geographers might talk about "scales" of suburbs, cities, regions and states, and political scientists may consider governance at local, regional, national, European and global "scales". Only sometimes do these designations have an obvious spatial or temporal dimension; the "scale" of species or of point samples is not obviously a spatial or temporal scale, for example. "Scale" can therefore also mean "level in a hierarchy" (Wu et al. 2006), and indeed the SCALES project considered "scale" in a broad sense encompassing all the meanings mentioned so far. The project's title refers to "administrative levels" and "spatial, temporal and ecological scales", and we may surmise that "administrative levels" and "ecological scales" are levels in hierarchies of organisation: political organisation in the first case, and biological in the second.

Spatial and temporal scales

In the realm of spatial and temporal scales, it is important to distinguish two contrasting concepts. "Scale" can be used to mean the overall *extent* of some region of space or time, and also the

Figure 1. A cultivated landscape in northern Germany, showing an extent of perhaps 100ha. Various grains might be detected in the patterns of the different habitat elements seen here, and these could suggest appropriate spatial resolutions for a sampling scheme and subsequent analysis.

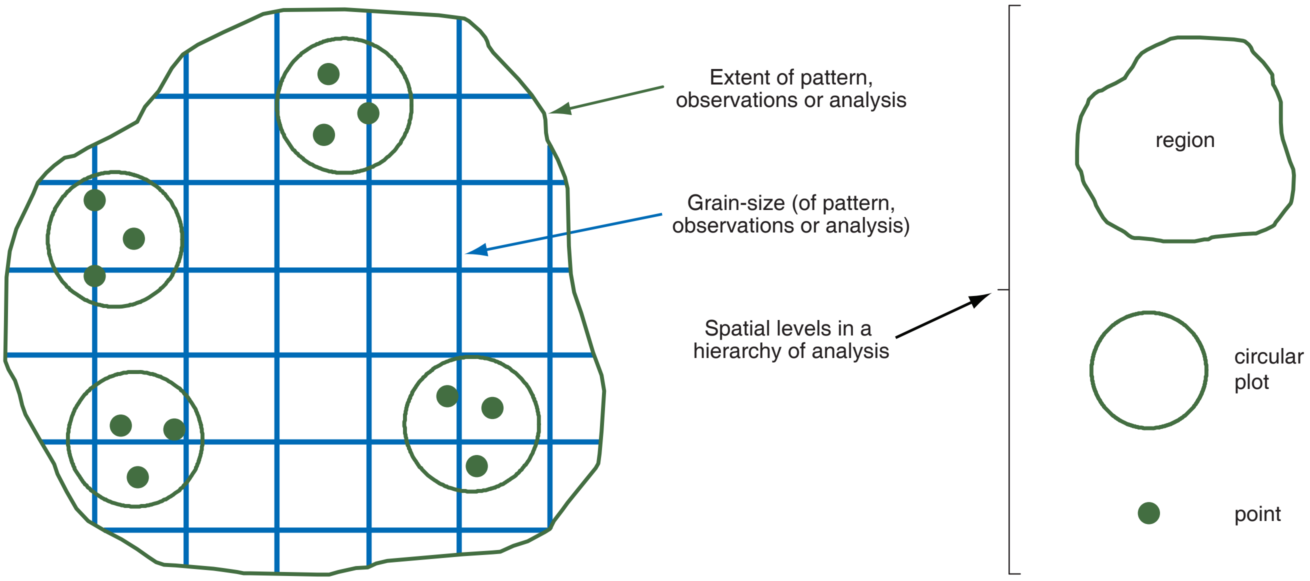

Figure 2. Basic scale concepts illustrated with a spatial scenario. "Extent" and "grain" are illustrated for a simple case of contiguous coverage. Scale as "level" is illustrated here with an example of three discrete, spatially-nested units at which some inferences might be made (e.g. for mean vegetation height). A non-spatial example could be for three levels of organisation: suppose that biomass densities were averaged over a single grass species, over all grass species and over all herbaceous vegetation within the circular quadrats.

size of the smallest unit in a pattern or analysis, sometimes called (spatially) the grain-size (Turner 1989). A closely related concept to grain-size is resolution, which may simply be its inverse, representing the density of information per unit space or time. It is meaningful to speak of "grain" and "resolution" when a continuous region is either intrinsically heterogeneous, or else digitised for analysis, both of which are common in ecological science.

"Extent" is a cumulative measure, and its units should normally be those of space (length, area or volume) or time. "Grain-size" is an average measure, based upon some discrete motif that is repeated over an extent, such as an individual, a quadrat, a sampling-period or a pixel. The units of grain-size should also be those of space or time. The related quantity "resolution" is reciprocally related to grain-size, so its units should be the inverse of space or time (items per unit space or time). "Resolution" is normally a property of an analysis (with uniform grain-size) rather than of a natural pattern, whereas "grain" can refer to either.

A range of other terms may be used for these basic scale concepts in specific contexts. For example, the notions of extent and grain may function quite differently when applied to patterns and processes, to observations and to analyses, and more-specific terms are sometimes used in these cases (Dungan et al. 2002). Moreover, sometimes it is difficult to decide which basic concept applies, especially in complex studies considering a hierarchy of non-contiguous spatial or temporal units. To help resolve this, additional terms may be needed, such as "focus" and "coverage". Such terms have not been commonly used in the SCALES project or this book, however, so they are not discussed further here. A helpful synthesis of scale concepts is given by Kienberger et al. (2013).

Table 1. Terms and basic concepts of "scale" as used in the SCALES project. The "synonyms and hyponyms" column contains both spatial and temporal terms related to extent and grain.

Key term	Definition	Synonyms and hyponyms (related to meaning of key term)	Examples of usage
extent	total size of a region in space or time	size, range, coverage, period[a], duration[a]	"The true extent of this population is unknown." "The extent of our study area was 10 km^2." "a broad-scale intervention"[c]
grain-size	size of the smallest unit of information in an analysis or pattern	grain, support, focus, periodicity, lag[a]; resolution[b], frequency[a b]	"a fine-grained pattern" "a coarse-grained (low-resolution) analysis"
level	vertical position in a hierarchy	rank	"At the level of individuals, we found great variability." "National governments can enact policies at the regional level."

[a] These terms are normally used for temporal scales.
[b] These terms refer to the inverse of grain-size.
[c] We recommend avoiding "large-scale" and "small-scale" because of the confusing way that a "large-scale" map tends to represent a smaller extent than a "small-scale" one.

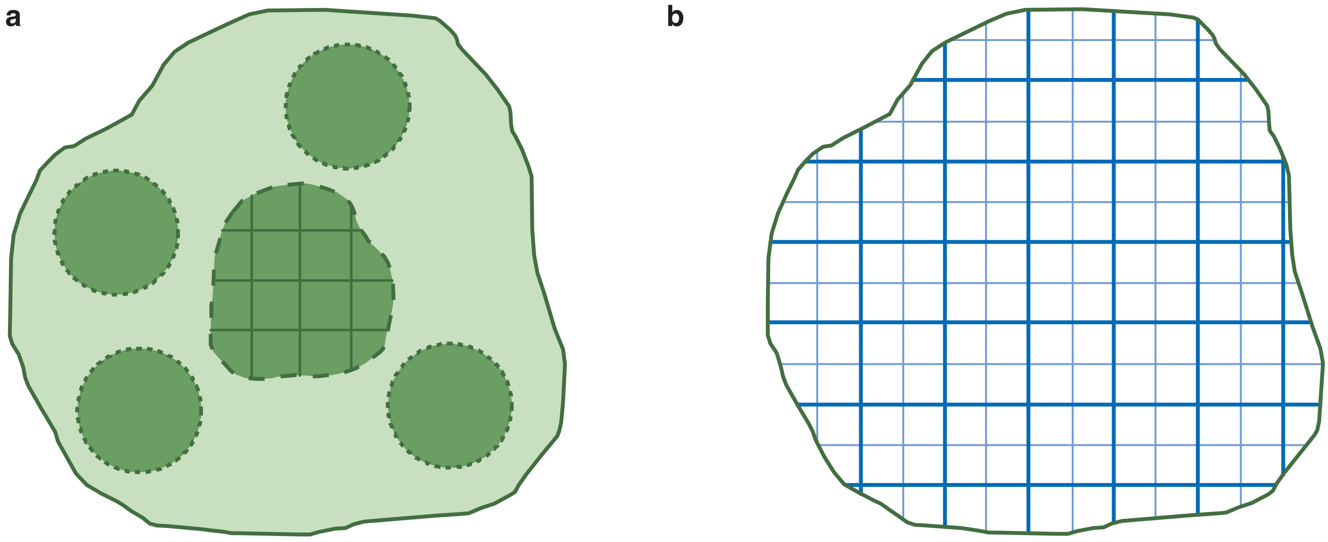

Figure 3. Various types of upscaling and downscaling. (a) Two forms of upscaling: (i) extrapolating data from a single small region (dashed enclosure) to predict some quantity in a target region of larger extent (solid enclosure), or (ii) upscaling from a number of samples (dotted enclosures) that have incomplete coverage of the target region. In some situations, corresponding forms of downscaling might be useful. (b) Downscaling by resolution: relating information at a coarser resolution (dark blue grid) to that at a finer resolution (pale blue grid). In some situations, a corresponding form of upscaling might be useful.

Table 1 presents the three basic meanings of the English word "scale" as found in geographical and ecological science. These are further illustrated in Figure 2.

An important theme in the SCALES project (Henle et al. 2010) is that of matching the scales at which conservation policies can be developed with the scales at which relevant ecological processes occur, and to encourage the enactment of suitable policies at appropriate scales. This may be a question of paying attention to multiple co-occurring processes and finding ways to match their spatial and/or temporal resolutions, so that conservation actions vary across space or time in accordance with the situations they are designed to improve. It may also concern spatial and/or temporal extents, so that conservation actions extend over enough space or time to achieve the intended effect in an efficient way. Failing in any of these ways may constitute a *scale-mismatch* between situation and response. A solution may involve *scaling*.

To scale or not to scale

"Scale" also functions as a verb in English. Many properties of a system are *scale-dependent*, and so we may want to *scale up* or *scale down* a quantity so that it applies to a different extent or resolution (normally spatial, as illustrated in Figure 3). *Upscaling* is often a form of extrapolation to a larger extent or coverage (Figure 3a), and a number of tools produced by the SCALES project provide methods for estimating quantities, such as species richness, over areas (extents or coverages) greater than those from which data are available. *Downscaling* is often a form of extrapolation or projection to an increased resolution (Figure 3b) (e.g. Gunton et al. 2014 this book, on microclimate).

Thermodynamics provides a helpful distinction here (Redlich 1970). When the extent of a pattern, study or analysis changes, its *extensive properties* can be expected to change with it: these are quantities that can simply be summed, such as population size, total carbon stock and water runoff. In a homogeneous system – or at spatial extents large enough with respect to the grain of heterogeneity – extensive properties scale linearly with space or time, being proportional to extent. Thus we can also say that a pair of variables *scale with* each other (Table 2), implying proportionality. By contrast, *intensive properties* need not vary when the extent varies, because they are averages or ratios with respect to extent. These typically include population density (per unit space), productivity (per unit time and space) and rainfall (per unit time). In a homogeneous system, intensive properties remain constant with re-

Table 2. Verbs related to scaling. Related terms that are not verbs are marked as nouns (n.) or adjectives (adj.).

Key term	Definition	**Synonyms and hyponyms** (related to meaning of key term)	Related terms	Examples of usage
to scale (transitive verb)	to transform a quantity so that it applies to a different scale	upscale (scale up), downscale (scale down), rescale	upscaling (n.), downscaling (n.), scalable (adj.)	"We can scale up our estimate from quadrats to grid cells."
to scale (intransitive verb)	to vary according to a mathematical law	be proportional		"Abundance scales with area, species richness with the logarithm of area."

Table 3. The key "scale" terms in English, with suggested translations into eight other languages.

Key English term	Part of speech	German	Polish	Hungarian	Finnish	Greek	French	Spanish	Chinese
scale	noun	Skala; Maßstab; Größenordnung	skala	skála; lépték	skaala; mittakaava	κλίμακα	échelle	escala	尺度
grain	noun	Rastergröße; Maschenweite; Zellgröße	ziarno	szemcseméret	rae	κόκκος	grain	grano	粒度
resolution	noun	Auflösung	rozdzielczość	felbontás	erotuskyky	ανάλυση	résolution	resolución	解析度
extent	noun	Ausdehnung	zasięg	kiterjedés	laajuus	έκταση	aire; étendue	extensión	範圍
level	noun	Ebene; Hierarchiestufe	poziom	szint	taso	επίπεδο	niveau	nivel	層級; 階層
to scale [something]	verb (transitive)	skalieren	skalować	skálázik	skaalata; muuntaa mittakaavaa	προβολή από μία κλίμακα σε άλλη κλίμακα	mettre à l'échelle	escalar	變尺度; 轉換尺度; 定訂尺度
to scale [together]	verb (intransitive)	proportional sein	odnieść	skálázódik	olla (mittakaavan mukaisessa) suhteessa	μεταβολή σύμφωνα με κάποιο μαθηματικό κανόνα.	se proportionner	variar en proporción	依數學定律改變

spect to extent. We may say that they are *scale-invariant*.

Other properties, such as the richness of species or alleles in a region, vary with extent but are not generally proportional to it. These may be termed *scale-specific*. These properties raise the most challenging questions of *scaling*: what is their relationship with extent? Perhaps a transformation of axes will provide a linear relationship, but often the problem is better solved when we have an understanding of underlying ecological processes. The study of allometry, concerned with relationships among different growth measures in organisms and with *cross-scale* self-similarity of spatial patterns, has shown that some such relationships can be considered as natural scaling laws (e.g. power laws for species–area curves: Storch et al. 2012). Others are artefacts caused by an unintended interaction between the grain-size of patterns and the resolution of methods used to study them. Both the detection and study of power laws and the analysis of artefacts to get knowledge of their effects can be aim of scale-related studies.

A lexicon for scale

Consultation with members of the SCALES project enabled us to compile a table showing how the basic scale terms in English may be translated into other languages represented in the project (Table 3).

It is apparent that most of these languages have a noun that is cognate with the English word "scale". This may be traced back to the Latin word *scala*, but in a number of cases it appears to have been taken as a loanword from the English "scale". Most languages also have a verb related to their "scale" noun.

Meanings of some of the words in Table 3 do not map exactly onto those of the English words given in the first column, so the table merely gives an indication of approximate correspondences. Many additional scale terms in these languages were revealed by our consultation, more of which are featured on the SCALETOOL web site (http://scales.ckff.si/scaletool/).

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CHAPTER II

Scaling of anthropogenic and natural drivers of biodiversity

Conceptual framework and typology of drivers

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Introduction

Climate change, land use change and biodiversity decline can be considered as the major global changes. Their effects are intertwined and operate in practically every region of the world, from local to global scale. The need for the conservation of biodiversity is universally acknowledged (Spangenberg 2007) and it has been the focus of extensive research by academic and non-academic scholars. Despite the repeated calls for action, and the targets agreed on and signed up to in the Convention on Biological Diversity (CBD), biodiversity continues to be under constant threat. Apart from the hotspots identified by Myers et al. (2000) land use changes and the related modifications of natural habitats generate threats for biodiversity at a global level. The conservation of biodiversity is an extremely complex task that needs to take into account ecological, economic and social parameters as well as their interactions. Much of the ineffective conservation action of the past could be attributed to the traditional approach in ecological research that separated humans from nature and did not include the effects of their actions on ecological systems (MEA 2005).

Biodiversity loss alters the resilience of ecosystems which has profound implications on the services that those ecosystems provide to humans (Chapin et al. 2000). Responding to this, scientists are increasingly focusing their research on the analysis of Socio-Ecological Systems (Berkes et al. 2003). A fundamental step in the analysis of Socio-Ecological Systems is the identification of the drivers of change that determine the status of ecosystems and their socio-ecological resilience (Folke et al. 2007). In different Socio-Ecological-Systems, drivers of change do not operate exactly in the same way and their possible associations are manifold. However a more limited number of territorial contexts of anthropogenic drivers can be identified. A more precise knowledge about those contexts can be of interest for conservation policies.

Different meanings have been attached to the concept of "driver of change" (Maxim et al. 2009) depending on the conceptual framework of analysis. Despite the multiplicity of meanings, most researchers agree that an analysis of drivers is imperative in order to derive policies that could alter the impact and intensity of human activities on ecosystems and therefore contribute towards biodiversity conservation. However, understanding drivers of cange, their impacts and their relationship to decision making constitutes a major challenge for science and policy makers. The challenge is not only related to the context of the analysis i.e. identification and description of all relevant social-economic-cultural and environmental drivers, but it goes further to the understanding of the effects of those drivers on biodiversity through habitat quality and suitability.

Drivers of change, more particularly anthropogenic ones, operate at various scales which do not always fit the scales that organisms or even ecosystems function. Furthermore, policies and policy instruments are elaborated over multiple scales (i.e., administrative units) which do not always comply with the scales of anthropogenic processes and their related impact on biodiversity. Therefore, the conservation of biodiversity requires the concurrent analysis of phenomena at multiple scales as well as interactions among them. To make things more complicated, the way drivers operate or appear over multiple scales is non-linear. Indeed, moving across scales, the intensity of a considered driver can change, as well as its spatial distribution (i.e., its evenness). Thus their impact on biodiversity and its conservation is scale sensitive and scale is a fundamental dimension for analysing the way drivers of change may affect the conditions of the maintenance of biodiversity. Furthermore, in the context of the European Union, the same drivers may affect biodiversity in different ways. Agricultural crisis, for instance, may represent a possibility for the increase of little human-driven ecosystems in regions where intensive farming is strongly developed; on the contrary, agricultural abandonment may be a threat in other contexts where semi-natural habitats need agro-pastoral practices to be maintained (Fonderflick et al. 2010).

In the SCALES project, one of the objectives was to identify a coherent set of drivers of change that affect biodiversity conservation, taking into account their scale sensitivity. In this chapter we describe the method applied to identify and select those drivers.

Methodological issues

The first part of the work was a compilation of a list of the most important drivers that affect biodiversity and the identification of relevant indicators (Tzanopoulos et al. 2013). This was achieved through an extensive literature review. The review included published work from scientific journals, databases and reports on the drivers of environmental change and the impacts of policies on anthropogenic processes. This review has revealed a number of issues that

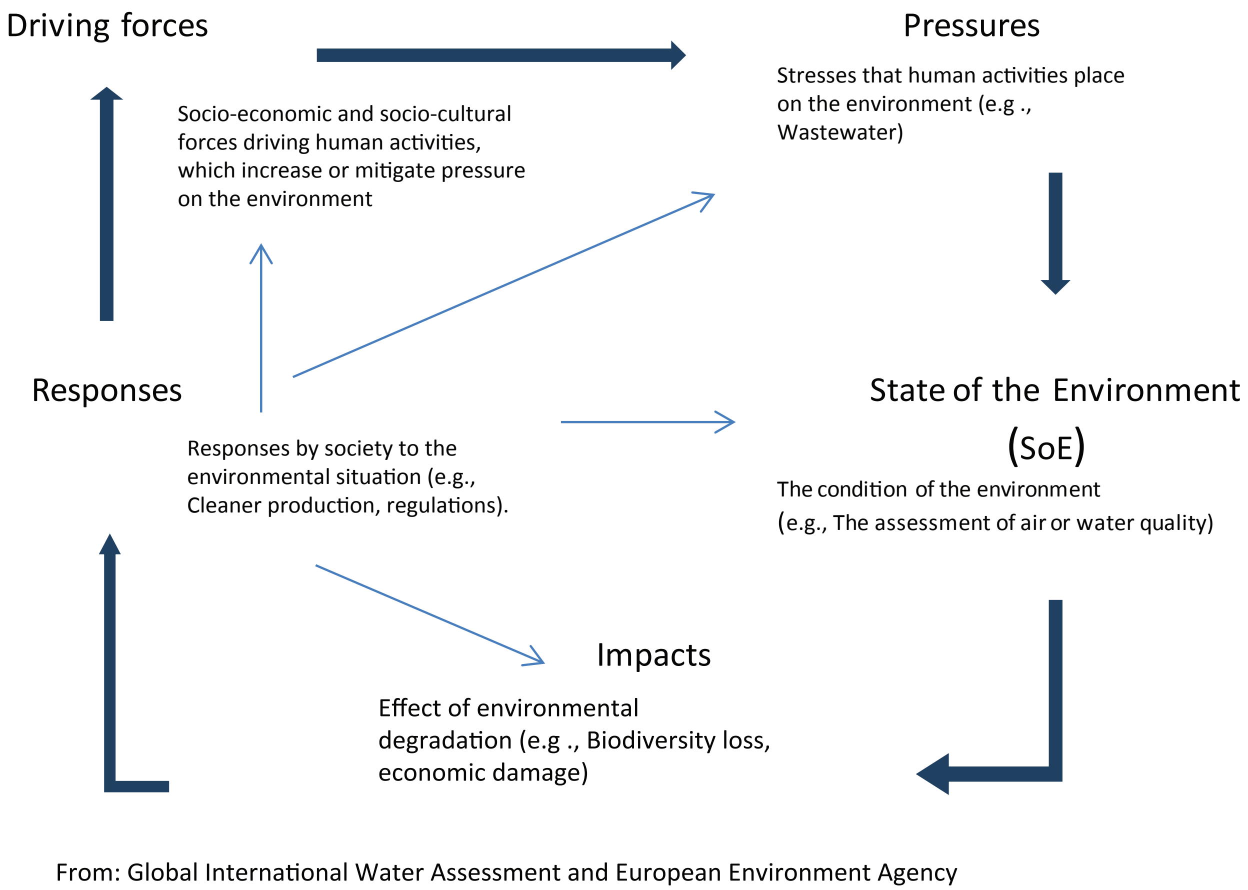

Figure 1. The DPSIR framework used by the EEA (Smeets et al. 1999).

needed to be addressed before the causal relationships between drivers and biodiversity can be examined.

Firstly, the framework of such an investigation had to be defined. The relationship among policies, drivers of change, anthropogenic processes and impacts on biodiversity was clarified and organized into a conceptual framework (Tzanopoulos et al. 2013). The literature review has shown that the boundaries between policies, drivers and processes are still unclear. For example, policies such as the Common Agricultural Policy or national forest policies are often regarded as drivers of environmental changes. On the other hand, many scholars argue that policies themselves are not drivers of change but a response of society to regulate anthropogenic processes. The DPSIR (Driver-Pressure-State-Impact-Response) framework, that was developed by the Organisation for Economic Co-operation and Development (OECD InterFutures Study Team 1979) and adopted by the European Environmental Agency (EEA) (Gabrielsen and Bosch 2003), provides a conceptual organization and reporting tool of the broad causal relationships between drivers, impacts and responses and thus clearly separates and defines the role of policies as means to influence and regulate drivers of change (Figure 1). Despite the criticism that this framework has received over the years (see Maxim et al. 2009) it is generally accepted that it can capture and communicate effectively the relationships among drivers and policy development. According to this framework, political bodies (EU, member states, regional or local governments) elaborate policies in order to affect drivers of change and to avoid or mitigate negative impacts on biodiversity and ecosystems. Thus, policy development requires a sound understanding of drivers of change and their characteristics. Drivers of change operate in non-linear ways at various spatial, temporal and administrative scales and the efficiency of a policy to achieve its targets depends to a large extent on the degree of addressing scale issues during the design process. Furthermore, policies are implemented and adapted to multiple governance contexts and levels (Bache and Flinders 2004).

Building on the above considerations, in the SCALES project we consider policies as regulators and not as drivers of change. Thus, we took into account the drivers of change that relate to the anthropogenic and natural processes which affect biodiversity, rather than policies *per* se. Since scale sensitivity of drivers has to be taken into account in order to design scale-sensitive policies within multi-level governance, the aims of the present work was to provide a typology of drivers described by quantitative indicators available at various scales..

The second issue that became apparent at the early stages of this review was that a diverse set of terms have been previously used to define drivers of change. This inconsistency in the terminology does not allow for a systematic comparison of the numerous studies on drivers of environmental change (for an extensive discussion on the inconsistency of terminology and the impacts of such a lack of clarity on the definition of drivers of change, see Anastasopoulou et al. 2009).

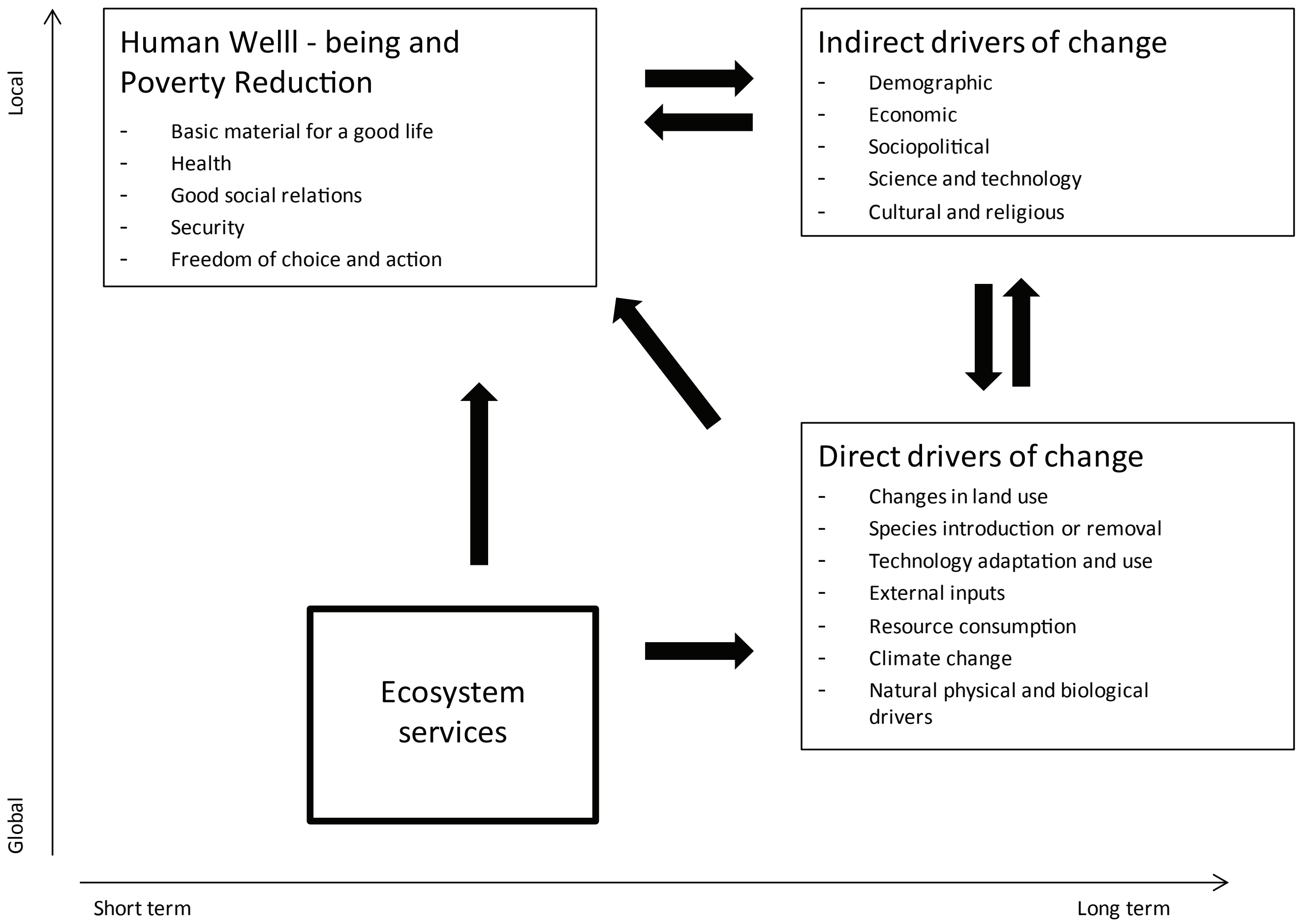

Figure 2. The MEA Framework linking indirect and direct drivers to human well-being.

The following two main approaches for defining drivers of change have been identified and are often used among researchers and practitioners:

- The **Millennium Ecosystem Assessment** (MEA 2005, Figure 2) approach which uses the terms "direct" and "indirect" drivers;
- The EEA approach which is linked to the **Driver-Pressure-State-Impact-Response** (DPSIR, see Figure 1) framework and uses the terms "driving forces" and "pressures".

According to the MEA approach the definition of a driver is "any natural or human induced factor that directly or indirectly causes a change in an ecosystem". The drivers are further divided into two categories:

- **Direct drivers** which have a direct impact on biodiversity;
- **Indirect drivers** whose impacts are more diffuse.

According to the EEA approach, the drivers are divided into "driving forces" and "pressures". However, the meanings of these two categories are almost identical to that of MEA with "driving forces" corresponding to indirect drivers of change and "pressures" corresponding to direct drivers only. The EEA approach is linked to the DPSIR framework, and though this framework is a relevant tool for structuring communication between scientists and end users, it fails to deal efficiently with the relationships between complex environmental and socio-economic systems; the framework rarely takes into consideration social and political aspects and is mainly based on the economic and environmental relationships only (Maxim et al. 2009). Additionally, "pressures" is not a neutral term. It carries an implicit value and puts the emphasis on the negative impacts of human activity on ecological systems. Taking these points into consideration, and to avoid further confusion, we concluded that for the SCALES project it would be more appropriate to adopt the MEA approach.

Furthermore, one should note the difference between the terms direct and indirect drivers vs. drivers related to anthropogenic and natural processes. Drivers related to anthropogenic processes can be both direct (e.g. urbanisation) and indirect (e.g. GDP), although most of them are indirect, while most drivers related to natural processes refer to direct drivers.

The third issue that needed to be addressed was directly related to topic of "scale". Since one of the objectives in SCALES was to develop a scale sensitive typology of drivers, we also carried out an extensive literature review on the relationship among drivers and scales and relevant categorisations. Since we adopted the MEA approach to define drivers and their categories, we paid particular attention to the relevant scale issues as described in the MEA approach. According to the MEA Multi-scale Assessments, there are two aspects of scales that must be taken into consideration for the categorisation of the drivers:

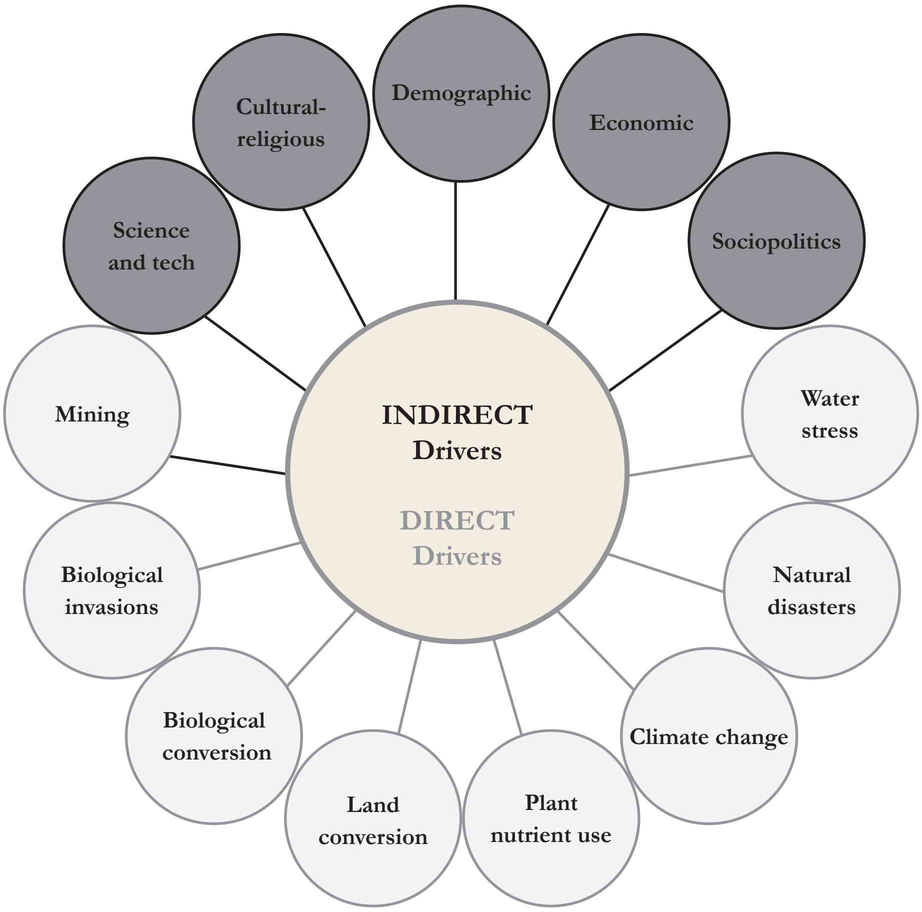

Figure 3. Common indirect and direct drivers of biodiversity change.

- The scale at which the driver operates;
- The scale at which humans can change the driver.

The MEA Multi-scale Assessments uses the second aspect and it categorises the drivers as:

- **Exogenous**: "Drivers that cannot be altered by a decision-maker at a certain scale, but influence his/her decisions, are called exogenous drivers";
- **Endogenous**: "Defined as the drivers that the decision-maker at a particular scale can influence" (MEA 2003).

Despite the interest in this categorisation it is not easily applicable since the assignment of a driver in one of the two categories is not always possible (e.g., urbanization may seem an endogenous driver at regional level, but it may be determined by decisions at national level). Therefore, our approach is based on the first aspect of scales, the scale at which a driver operates.

Identifying common drivers and associated indicators

Major indirect drivers used in many assessments are often divided into five categories: demographic, economic, socio-political, science and technology, and cultural and religious (Figure 3, Burgi et al. 2004). On the other hand, the direct drivers, are primarily physical, chemical and biological. They include, but are not restricted to: land conversion, plant nutrient use, water stress, pollution, mining, as well as biological invasions, climate change and natural disasters (Figure 3, Salafsky et al. 2008, Spangenberg 2007, Forester and Machlis 1996).

Taken together, these drivers underpin broad concepts that can be addressed in multiple ways. They can be described by indicators in order to simplify the complexity by focusing on relevant aspects for which data is available (Smeets et al. 1999). For example, Land Conversion is a direct driver with high complexity given its multiple and often contrasting impact on biodiversity; indicators that describe and enable the quantitative assessment of Land Conversion (such as Deforestation, Afforestation, Reforestation, Desertification, Agricultural Conversion, Urbanisation) can simplify the complexity of this driver and facilitate monitoring and policy development to address its impacts. Thus, indicators allow the supply of relevant information, and set a basis for dialogue. This is particularly true with regard to policy development or priority setting. As environmental indicators are of high relevance for international cooperation, policy implementation and monitoring, their number and diversity have been growing in recent years (Piorr 2003). For drivers of biodiversity change, several international organizations have been interested, directly or indirectly, in the definition of indicators related to the issue. From this perspective, we propose a non-exhaustive list of relevant indicators based on an extensive literature review. It is presented in Table 1 and Table 2 (see Tzanopoulos et al. 2014 this book) for indirect drivers and direct drivers respectively.

Among common sources of indicators are the United Nations Commission on Sustainable Development, the Food and Agriculture Organization of the United Nations, the United Nations Millennium Development Goals, the Organisation for Economic Cooperation and Development, the European Environment Agency, the European Commission and especially the Eurostat database, as well as EU funded projects such like ESPON (European Observation Network for Territorial Development and Cohesion). As a result, the established list is substantial with more than a hundred indicators. Nevertheless, the availability of data at lower levels, a prerequisite for carrying out studies dealing with the perception of drivers across administrative levels, has significantly reduced the number of indicators which are available for practical application. For instance, EUROSTAT provides data on land under irrigation at NUTS[1] level 3, whereas no data is available at that scale for land under organic farming. Even though data availability was low at NUTS level 3, the methodology elaborated allowed the selection of 27 indicators used to investigate to what extent drivers were scale sensitive (Tzanopoulos et al 2013).

Discussion

Once a framework for the identification of drivers of biodiversity change is adopted, the issue of data availability is still to be dealt with. If the main EU data-bases allow getting information at all NUTS levels for direct and indirect drivers, data are still scarce for some categories of indirect drivers. Our work addressed the issue of indirect drivers like socio-political factors, cultural-religious factors or science and technology with the available indicators. If (or when) future research infrastructure provides data from long term surveys on the ways European citizens relate and care for nature and biodiversity, it will allow scientists to deal with indirect cultural drivers in a more satisfying way.

If socio-economic data are selected as drivers, they are supposed to have an effect on biodiversity loss and natural habitats change. At that stage, based on the available literature, we selected a set of drivers whose effect is considered important in conservation science. However, policy-makers may want, from a local to EU perspective, to know more precisely what type of impact a driver may have on biodiversity and habitat suitability. As precise studies can be carried out only on a limited number of species or habitats and necessarily through a limited number of field studies, two alternative ways exist to link a driver to an effect. The first one is modelling, as done, for instance, by Green et al. (2005) who provided a theoretical framework widely used and discussed. The other solution is to collect expert advice and opinion through a wide survey using internet-based collaborative tools, allowing a high number of persons interested in sharing their knowledge on biodiversity to select habitats and geographical regions and to provide their expertise. These methods are based on the principles of crowdsourcing and public participation (Seltzer and Mahmoudi 2013).

Even though all the effects of the selected drivers are not precisely identified throughout the EU, analysing and mapping them at various scales provides information of interest both for science and policy. First (see Tzanopoulos et al. 2014 this book) it allowed an understanding of how and to what extent those drivers were scale-sensitive. Other information may be given by the way the drivers overlap in the EU territories. These different patterns of overlapping may define biodiversity management from local to regional contexts. This is particularly important at a time when a challenge in EU policies is to integrate different sectorial policies and to "ecologize" them.

The EU territory is structured by member state borders and, at lower scales, by administrative scales. It was not possible to conduct a transboundary analysis of the scale sensitivity of drivers of biodiversity change in the SCALES project. However, maps (see Tzanopoulos et al. 2014 this book) show that drivers are highly sensitive to state or administrative borders. Furthermore, regional administrative or State borders may cross homogeneous landscapes. But different administrations may adopt different policies and set different priorities (Henle et al. 2010). As emphasised by several intellectual schools in management of biodiversity (McGinnis 1999), policies should try to minimize the contrasts due to political or administrative borders.

Conclusion

Based on an extensive literature review, the conceptual framework adopted in the SCALES project for analysing the scale sensitivity of anthropogenic drivers of biodiversity change allowed for selecting a set of direct and indirect drivers and, for each of them, an indicator available at different scales. This conceptual framework was also a way for clarifying notions and terms (pressures, drivers, impacts) often used in the literature in a confusing way. In order to build a robust characterisation of the regional signature of anthropogenic pressures on biodiversity, decision makers and environment managers may use a set of indicators without confusing drivers of change and policy response.

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[1] NUTS stands for Nomenclature of territorial units for statistics. It is a hierarchical system for dividing up the economic territory of the EU for the purpose of collecting statistics for socio-economic analysis as well as framing EU regional policies (http://epp.eurostat.ec.europa.eu/portal/page/portal/nuts_nomenclature/introduction)

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Scaling of drivers of change across administrative levels

Joseph Tzanopoulos, Raphaëlle Mouttet, Aurelien Letourneau, Ioannis N. Vogiatzakis, Simon G. Potts, Klaus Henle, Raphaël Mathevet, Pascal Marty

Introduction

The conservation of biodiversity is an extremely complex task that needs to take into account ecological, economic and social parameters as well as their interactions. A number of analytical frameworks have been developed in order to facilitate interdisciplinary approaches to biodiversity conservation, such as scenario analysis, sustainability appraisal, and analysis of socio-ecological systems. A fundamental step in all these approaches is the identification of drivers of change which determine the status of ecosystems and their socio-ecological resilience. However, understanding drivers of change, their impacts and their relationship to decision making constitutes a major challenge for scientists and policy makers. The challenge is not only related to the context of the analysis i.e. identification and description of all relevant social-economic-cultural and environmental drivers, but it goes further to the choice of the appropriate dimensions and quantifiable organization of the analysis, in other words "the scale" of the analysis.

Drivers of change operate at various scales, which do not always match the scales that are relevant for organisms or ecosystems functions. Furthermore, policies and their instruments are elaborated over multiple scales (e.g. administrative units), which do not always match the scales of anthropogenic processes and their related impact on biodiversity. In addition, the way drivers operate or appear over multiple scales often is non-linear (Cash et al. 2006). Indeed, as we move across scales, the intensity as well as the spatial distribution (i.e. its evenness) of a particular driver may change. Thus drivers' impact on biodiversity and its conservation is scale-sensitive, and it is necessary to analyze and describe the way drivers of change operate over multiple scales by measuring their scale-sensitivity in order to better support policy making at the European scale (Tzanopoulos et al. 2013).

In this chapter we present first a conceptual framework to measure the scale-sensitivity of drivers of change over multiple scales of analysis and second, we define a typology of scale-sensitivity that classifies and summarizes the behaviour of drivers across multiple administrative levels.

Thus we discuss how this new typology of drivers can inform policy making for biodiversity conservation.

Identification of drivers of change and their indicators

There is a large number of concepts and terms that have been used to define drivers of change resulting in an inconsistency across terminologies. In our analysis we have followed the Millennium Ecosystem Assessment (MEA 2005) approach that defines drivers as "*any natural or human induced factor that directly or indirectly causes a change in an ecosystem*" and which subsequently divides drivers into two categories: (1) 'direct drivers', which have a direct impact on ecosystems, and (2) 'indirect drivers' whose impacts are more diffuse. Direct drivers are primarily physical, chemical and biological. They include, but are not restricted to, land conversion, plant nutrient use, water stress, pollution, mining, as well as biological invasions, climate change and natural disasters. On the other hand, major indirect drivers are often divided into five categories: demographic, economic, socio-political, science and technology, and cultural and religious.

We started our analysis by compiling a list of the most important drivers that affect biodiversity and subsequently we identified relevant indicators. This was achieved through an extensive literature review. The review included published work from scientific journals, databases and reports on the drivers of environmental change and the impacts of policies on anthropogenic processes. A subset of 27 indicators (out of 94 initially identified) was selected (Table 1) to be included in our analysis, based on the criteria of data availability for all four levels of the European NUTS nomenclature. The term NUTS stands for "Nomenclature of territorial units for statistics" where NUTS 0 indicates countries, NUTS 1 major socio-economic regions, NUTS 2 basic regions and NUTS 3 small regions (see http://epp.eurostat.ec.europa.eu/portal/page/portal/nuts_nomenclature/introduction).

We then compiled all indicator data into a GIS database and a set of four maps per indicator was produced, one for each administrative level (NUTS 0, NUTS 1, NUTS 2 and NUTS 3). The values of the indicators that have been used to generate the multiple maps were standardised by area (extent of each NUT).

Development of a tool for quantifying and assessing scale sensitivity

In order to quantify the scale-sensitivity of indicators we used two key metrics across administrative levels,

Table 1. List of indicators used in the analysis.

Indicator	Unit	Data Source
Afforestation	% of total area	CLC change 1990-2000
Age structure (×3)	% of population within each age class	Eurostat (decade 2000-2010)
Agricultural conversion	% of total area	CLC change 1990-2000
Arable area	% of total area	CLC 2000
Deforestation	% of total area	CLC change 1990-2000
Employment in agriculture	% of total active population	Eurostat (decade 2000-2010)
Employment in industry	% of total active population	Eurostat (decade 2000-2010)
Employment in services	% of total active population	Eurostat (decade 2000-2010)
Farm size	Hectares	Eurostat (decade 2000-2010)
Farm standard gross margin	European size units / utilised agricultural area	Eurostat (decade 2000-2010)
Farmers training level	% of farmers with full agricultural training	Eurostat (decade 2000-2010)
Forest area	% of total area	CLC 2000
Gross Domestic Product (GDP)	Purchasing power standard per inhabitant	Eurostat (decade 2000-2010)
Irrigation	% of Utilised agricultural area	Eurostat (decade 2000-2010)
Livestock density	Livestock units/ Utilised agricultural area	Eurostat (decade 2000-2010)
Mortality	Number of deaths per 1000 inhabitant	Eurostat (decade 2000-2010)
Pasture area	% of total area	CLC 2000
Permanent crop area	% of total area	CLC 2000
Population density	Number of inhabitants per km^2	Eurostat (decade 2000-2010)
Tourism infrastructure	Number of beds in hotels per km^2	Eurostat (decade 2000-2010)
Unemployment	Unemployment rate	Eurostat (decade 2000-2010)
Urbanization	% of total area	CLC change 1990-2000
Utilised agricultural area	% of total area	Eurostat (decade 2000-2010)
Wetland loss	% of total area	CLC change 1990-2000
Women in active population	% of total active population	Eurostat (decade 2000-2010)

"change in intensity" and "evenness", to express the homogeneity of the indicator's spatial pattern. The use of these metrics for assessing scale sensitivity is visually explained in Figure 1.

Change in intensity (I) was assessed by measuring the relative change in the median of an indicator at a given administrative level compared to NUTS level 0. Intensity is equal to zero at NUTS level 0 and can either be positive or negative for other levels. Intensity measures how low or high values of an indicator are over- or under-represented within regions from one NUTS level compared to NUTS level 0. A negative value of intensity stands for an over-representation of low values of the indicator whereas a positive one stands for an over-representation of high values.

Evenness was measured using Shannon's Evenness Index (E) (Hill 1973) which is derived from Shannon's diversity index. Evenness is a measure of how similar the values of an indicator are for different regions within a larger unit. The Shannon's Evenness Index is constrained between zero and one and the closer evenness is to 1, the more the regions within a NUTS level are similar in terms of the values of the indicator.

The values of these two metrics for all 27 indicators and across all different administrative levels were then plotted on a two-axes graph (Figure 1), which provides a visual summary of the relative scale sensitivity of each driver of change.

Finally, we developed a typology of scale-sensitivity of drivers of change by classifying the respective indicators into five categories according to their behaviour (change in intensity and evenness) across the four administrative levels. The analysis was performed with the R software using the Ward's agglomerative clustering method.

Visualization of indicators across administrative levels

The visual examination of the maps produced for all administrative levels from NUTS 0 to NUTS 3 emphasize some differential break-up patterns of indicators' values through the European territory. These differentiations are more or less marked depending on the indicators, the regions or the level concerned. This is a first evidence for the existence of scale sensitivity in the way drivers disaggregate across the EU. Below we present and discuss the visualisations for a sample of 3 drivers across scales. The whole set of 108 visualizations (27 indicators ×4 maps at NUTS 0-3) is available at http://www.scales-project.net/).

Figure 2 focuses on the situation of agricultural conversion in Germany and Poland. At NUTS level 0, both

Figure 1. Conceptual framework to assess scale sensitivity of drivers (A, B, C, D and E). Source: Tzanopoulos et al. (2013).

countries tend to show a similar pattern in the share of surfaces affected by agricultural conversion. However, mapping similar data at NUTS level 1 reveals some strong regional contexts. Globally, Polish NUTS 1 regions have medium rates of conversion whereas a contrasting pattern is observed in Germany. In eastern Germany, NUTS 1 regions show a strong agricultural conversion while regions from the western part have low rates of conversion. In this case, an observation of the conversion process at the country level can lead to a misinterpretation of the situation since high values are spatially clustered over the boundary of administrative units and are produced by different processes (i.e. Eastern Germany was strongly affected by decollectivization and transition to market economy).

Conversely, mapping GDP in southern Sweden at NUTS level 2 and 3 does not highlight many differentiations (Figure 3). The underlying phenomenon stands in the homogeneous GDP distribution over the five NUTS 3 regions.

To the contrary, similar indicator values can be clustered within broader administrative units, as shown in Figure 4. The eastern NUTS 2 regions that comprise the Danube delta, experience much higher absolute wetland loss than the western regions. A similar pattern is observed at NUTS level 1.

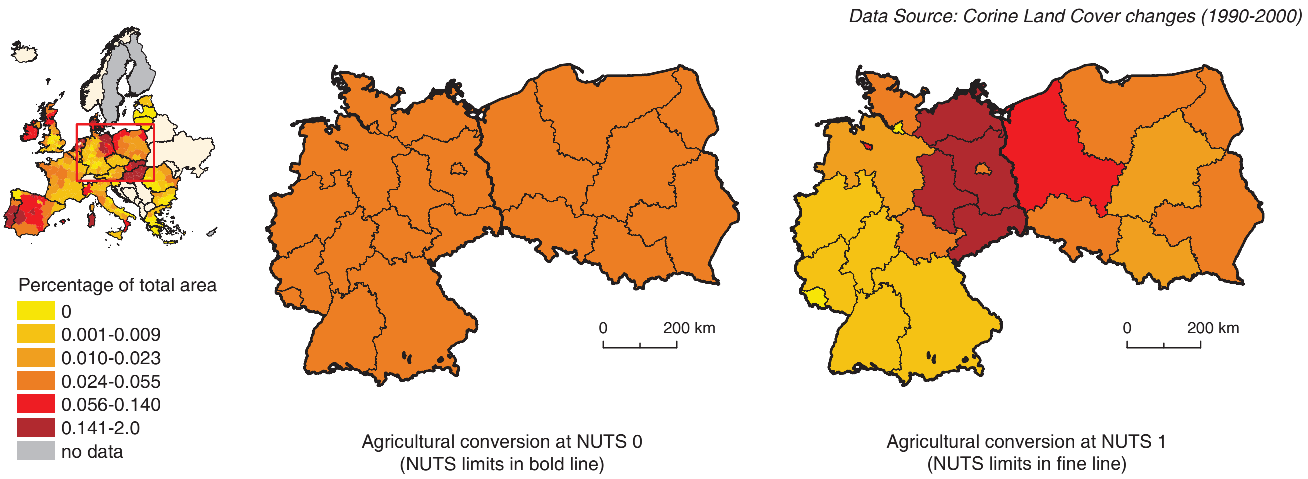

Figure 2. Agricultural conversion at NUTS 0 and NUTS 1 in Germany and Poland.

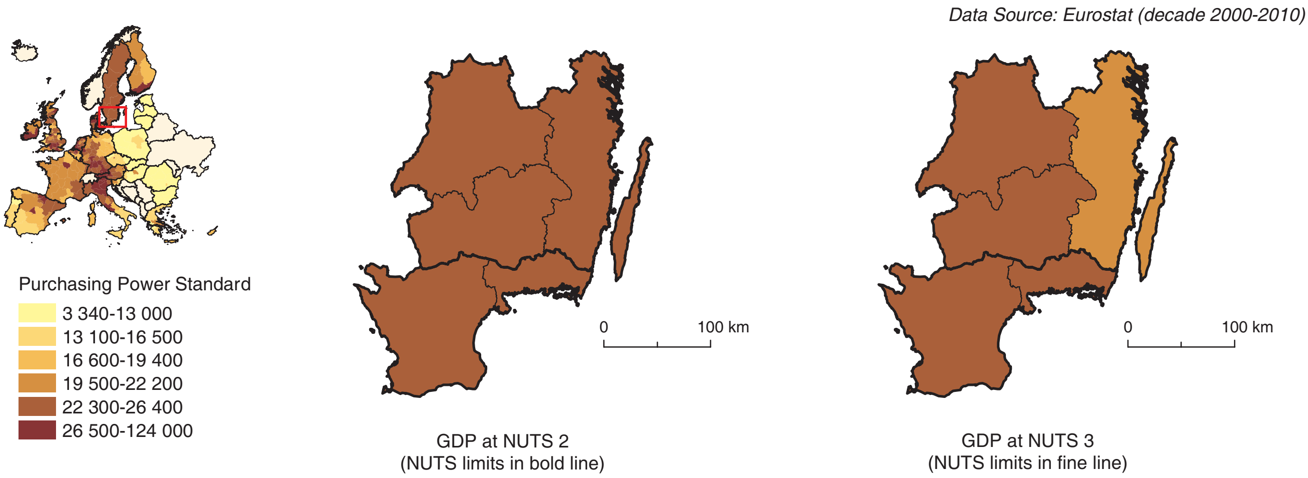

Figure 3. Gross Domestic Product at NUTS 2 and NUTS 3 in Småland and South Sweden (Sweden).

Thus, maps allow presentation of complex information into compact and accessible forms. Specific geographical patterns, such as discontinuities or concentrations, can be pointed out. Despite the many advantages in using maps in such an analysis, they also have their limitations. Regarding scale issues, the information they show at EU level is still very complex and predominantly qualitative. Therefore, there is a need for a quantitative assessment of the drivers' scale sensitivity.

Classification of drivers based on their scale-sensitivity

The calculation of evenness and change in intensity across administrative levels provides strong evidence of scale-sensitivity; the evenness and intensity of indicators can vary in amplitude, direction and depend on the levels concerned. However, the classification analysis shows that 5 different patterns (classes) (Table 2) can be identified regarding the behaviour of various drivers and their indicators across administrative levels (Figure 5). In class 1, the change in evenness and intensity is very limited. This class groups together most of the indicators related to demographic and economic indirect drivers; those indicators show a rather homogeneous pattern across EU countries and across different administrative levels. This pattern is expected to a certain extent since most of the EU Member States are among the most developed globally and thus, if compared to other countries across the globe, they do appear rather homogenous in terms of economic development. Class 2 exhibits a similar behaviour. However, slight differentiations across administrative levels are more apparent: evenness is lower than in class 1 and tends to increase when moving towards lower levels. Indicators in class 2 are mainly related to changes in the agricultural sector and rural areas (e.g. farm size, livestock density, pasture area). The following classes (from 3 to 5) can be characterized as much more scale-sensitive. First of all, class 3 displays both an increase in evenness and an increase in intensity when moving towards lower levels. In contrast, classes 4 and 5 show an increase in evenness, but exhibit a decrease in intensity at finer administrative levels. Classes 4 and 5 differ in their amplitude of variation, with class 5 having much stronger variation. Most of the indicators in class 3 refer to geographical dynamics of urban areas (e.g. urbanization, population density,

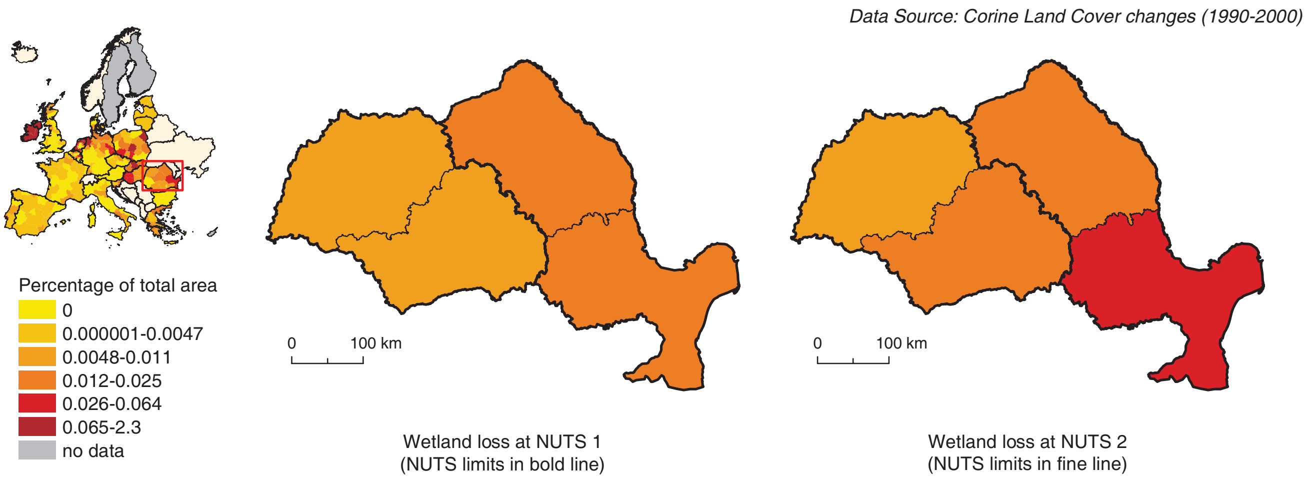

Figure 4. Wetland loss at NUTS 1 and NUTS 2 in Macroregiuneaunu and Macroregiuneadoi (Romania).

tourism infrastructure). Urban dynamics traditionally appear spatially clustered in specific locations across Member States. Finally, classes 4 and 5 consist of indicators related to land conversion; their high scale-sensitivity indicates that land conversion is taking place in a rather spatially clustered pattern with very different geographical signatures among EU regions.

Discussion and conclusions

Spatial analyses across scales are essential to understand what makes a driver scale-sensitive and to reduce mismatches between the scale at which drivers operate and at which they are addressed by policy instruments (Henle et al. 2010). Thus it is important to investigate:

- To what extent is a driver scale-sensitive?
- To what level of detail should indicators be mapped?
- To what extent is an effort for making data available at lower scales necessary?

Our analysis has shown that scale-sensitivity varies considerably among drivers. Drivers were classified into five broad categories of scale-sensitivity depending on the response of evenness and change in intensity as we move across administrative levels. Generally speaking, indirect drivers tend to show low scale-sensitivity; most of the indicators referring to economic sectors or demographic and social drivers show minimal changes as we move across administrative levels. However, not all indirect drivers behave as non-scale-sensitive, and tourism is a characteristic example of a scale-sensitive indirect driver. In contrast, most direct drivers show high scale-sensitivity with characteristic examples being deforestation, agricultural conversion and wetland loss.

The presence of scale-sensitivity has important implications for policy-making. Policies addressing direct drivers of change (such as land conversion) need to be scale-sensitive (i.e. to take scale into consideration during the designing process) in order to

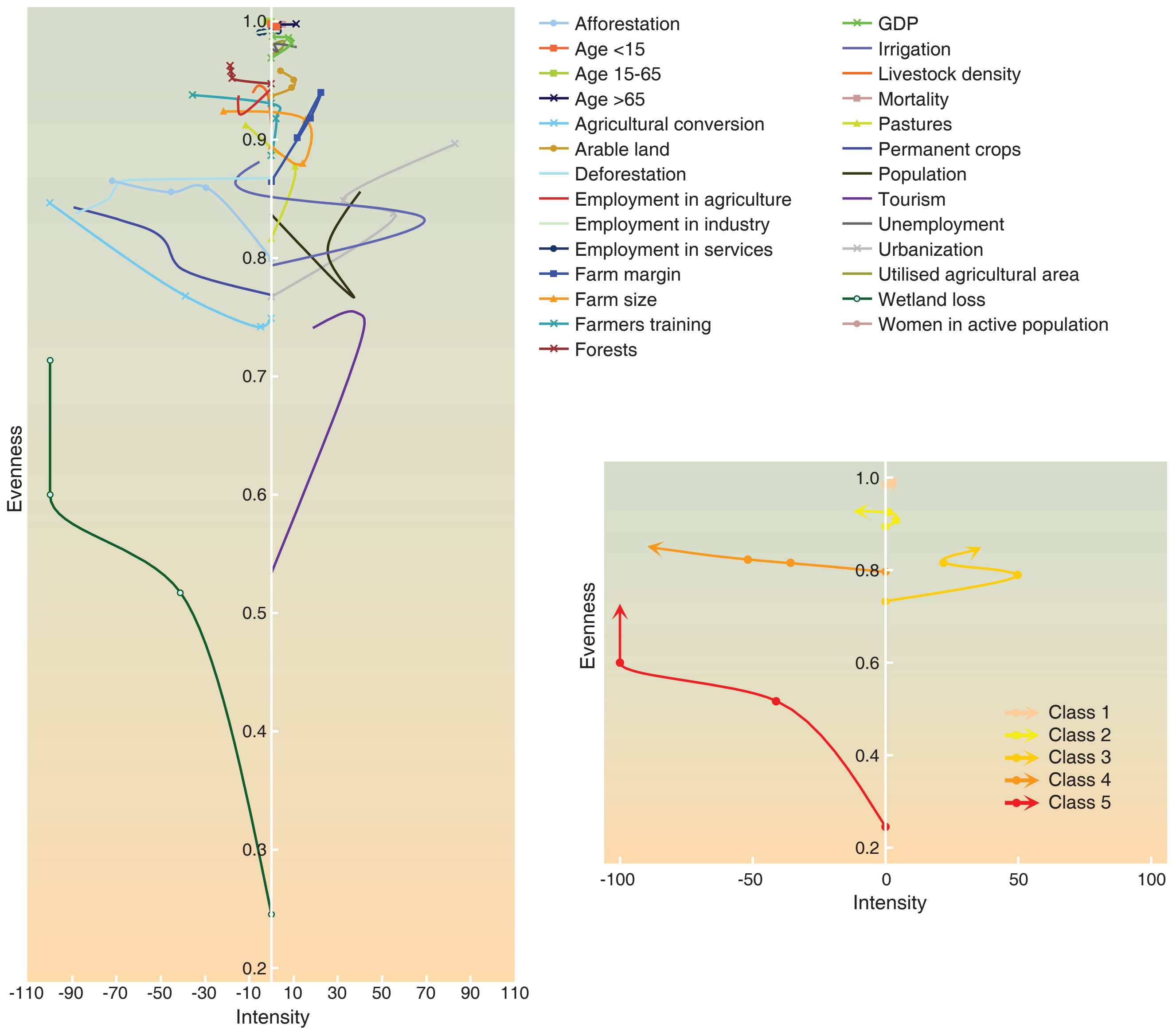

Figure 5. Change in intensity and evenness of indicators and classes of indicators across administrative levels.

Table 2. Classification of indicators according to their scale-sensitivity.

Class	Scale sensitivity	Evenness	Change in Intensity	Indicators
1	Very low	Almost no change (0)	Almost no change (0)	Age structure Arable land Employment in industry Employment in services GDP Mortality Women in active population Utilized agricultural area Unemployment
2	Low	Slight increase (↑)	Slight increase (↑)	Employment in agriculture Farm margin Farm size Farmers training Forests Livestock density Pastures Urban area
3	Moderate	Moderate increase (↑)	Moderate increase (↑)	Hotels Irrigation Population Urbanization
4	High	Moderate increase (↑)	Moderate increase (↑)	Afforestation Deforestation Agricultural conversion Permanent crops
5	Very High	Large increase (↑)	Large increase (↑)	Wetland loss

better respond to scale differentiation that is observed across administrative levels. It is important that direct drivers are examined at least at NUTS 3 level in order to capture more efficiently spatial variation and polarization effects. The high scale-sensitivity of direct drivers of change advocates for flexibility and a degree of autonomy in regional/local decision-making for environmental management and planning. Strengthening multi-level governance could tackle such scale-sensitivity of direct drivers of change.

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Scaling of habitat loss in Natura 2000 network

Konstantinos Touloumis, John D. Pantis

Introduction

The loss and the fragmentation of natural habitats are among the most significant threats for global biodiversity (Sala et al. 2000). Expanding human related land uses have modified the majority of the planet's terrestrial area (Haberl et al. 2007) contributing to the decline and/or the loss of a great amount of populations and species. Apart from the destruction of natural habitats, the consequent pollution arising from human land uses as well as the increasingly important role of climate change project an even more worrisome future for the environmental conditions of Earth.

Nevertheless, human impact on the landscape could have positive effects on biodiversity, since several habitat types resulting from these transformations (e.g. semi-natural grasslands, meadows, landscape mosaics from High Natural Value farming) are considered to be habitats of high interest for biodiversity conservation. Nowadays, Protected Areas (PAs) and their legislative frameworks represent the basic conservation tool towards managing landscape in a beneficial way for biodiversity. Globally, PAs cover more than 12% of Earth's land surface and their coverage is expected to expand in the next few years (Joppa and Pfaff 2011). In Europe a common conservation strategy has led to the development of the largest conservation network of PAs in the world. The Natura 2000 network counts today more than 25,000 sites. Its main goal is to "enable the natural habitat types and the species' habitats concerned to be maintained or, where appropriate, restored at a favourable conservation status in their natural range" (EU Directive 43/92). According to the Directive 43/92 the conservation status of a habitat could be characterized as "favourable" (among others) when "its natural range and areas it covers within that range are stable or increasing" while a species conservation status is favourable when "there is […] a sufficiently large habitat to maintain its populations on a long-term basis". Still, to what extent the Natura 2000 network has managed to maintain or restore natural or/and semi natural habitats within its territories, has not been tested yet.

By using data from 25,703 protected areas distributed over 24 EU member states, we examined different patterns of land cover changes between the area occupied by the Natura 2000 network and the unprotected European territory, for the time period 2000-2006. We focused our research on the expansion rates of the CORINE land cover classes 'Forests and semi-natural areas' and 'Agricultural areas', since they have been recognized as crucial estimators of the PAs efficiency. Our target was to detect "hot-spots" of natural habitat loss within the Natura 2000 network across the EU and across administrative levels and to examine whether the trends of natural habitat transformations could potentially be associated with corresponding changes in the agricultural land.

Methods

We used two versions of CORINE land cover maps (CLC2000 and CLC2006) to quantify land cover transformations from 2000 to 2006. The two maps were produced by satellite images and they had comparable accuracy (>85%) and time consistency (±1 year). Our analysis covers 24 EU Member States who participated both in CLC2000 and CLC2006 programs. We calculated land cover transformation for two of the five broad categories of CORINE classes: class 2, 'Agricultural areas' and class 3, 'Forests and semi-natural areas'. Land cover changes between 2000 and 2006 have been calculated on the basis of an index, NET%, which is an expression of the net land cover change for a particular CORINE class as a percentage of the area this land cover class occupied in 2000. The index has been calculated only for the parts of the landscape that are situated within the boundaries of Natura 2000 network. Furthermore, our analysis was conducted for three levels of the administrative scale: National level, NUTS 2 (basic regions for the application of regional policies), NUTS 3 (small regions for specific diagnoses). At each level, land cover changes were calculated separately for the parts of the network that are within the boundaries of every territorial unit (i.e. at National level, we calculated NET% index for each one of the 24 national networks of Natura 2000, etc.).

Results

Eight Member States (Sweden, Finland, Germany, Czech Republic, Spain, Italy, Bulgaria and Romania) exhibited minor losses of 'Forests and Semi-natural areas' within their national networks of Natura 2000 sites (Figure 1a). On the contrary, in sixteen Member States 'Forests and Semi-natural areas' has expanded inside Natura 2000 and in three of them, Ireland, Netherlands and Slovakia the NET% index was higher than 2%. At the second level of territorial units (NUTS 2 level), we found that major losses (NET% <2%) of 'Forests and Semi-natural areas' inside Natura 2000 network has been recorded only in five provinces around the EU: Sicily in Italy, South-East development region of Romania, South Sweden, Hamburg in Germany as well as in East Flanders in Belgium (Figure 1b). On the contrary, high values of NET% gains (>2%) have been recorded in several provinces, mainly in Eastern Europe (in Poland, Slovakia and Hungary) in Northern Europe (Netherlands, Belgium, Denmark, Ireland) and in one province in Italy (Marche). Finally, at the third NUTS level, we found that administrative units where 'Forests and Semi-natural areas' has been lost inside the Natura 2000 network are scattered throughout the EU territory (Figure 1c). Indeed, when we focus on this NUTS level, several administrative units with high losses emerge, mainly in France, northern Italy, and eastern Germany as well as in other parts of Europe. As far as the administrative units where NET% is higher than 2% are concerned, again clusters are apparent in Eastern Europe, Ireland, Belgium and Netherlands, as well as in regions in France, northern Italy, Sicily, and southern Sweden.

In 'Agricultural areas' CORINE class and at National level, we notice that high losses (NET%<-2%) are recorded in three Member States (Poland, Slovakia and Netherlands), mi-

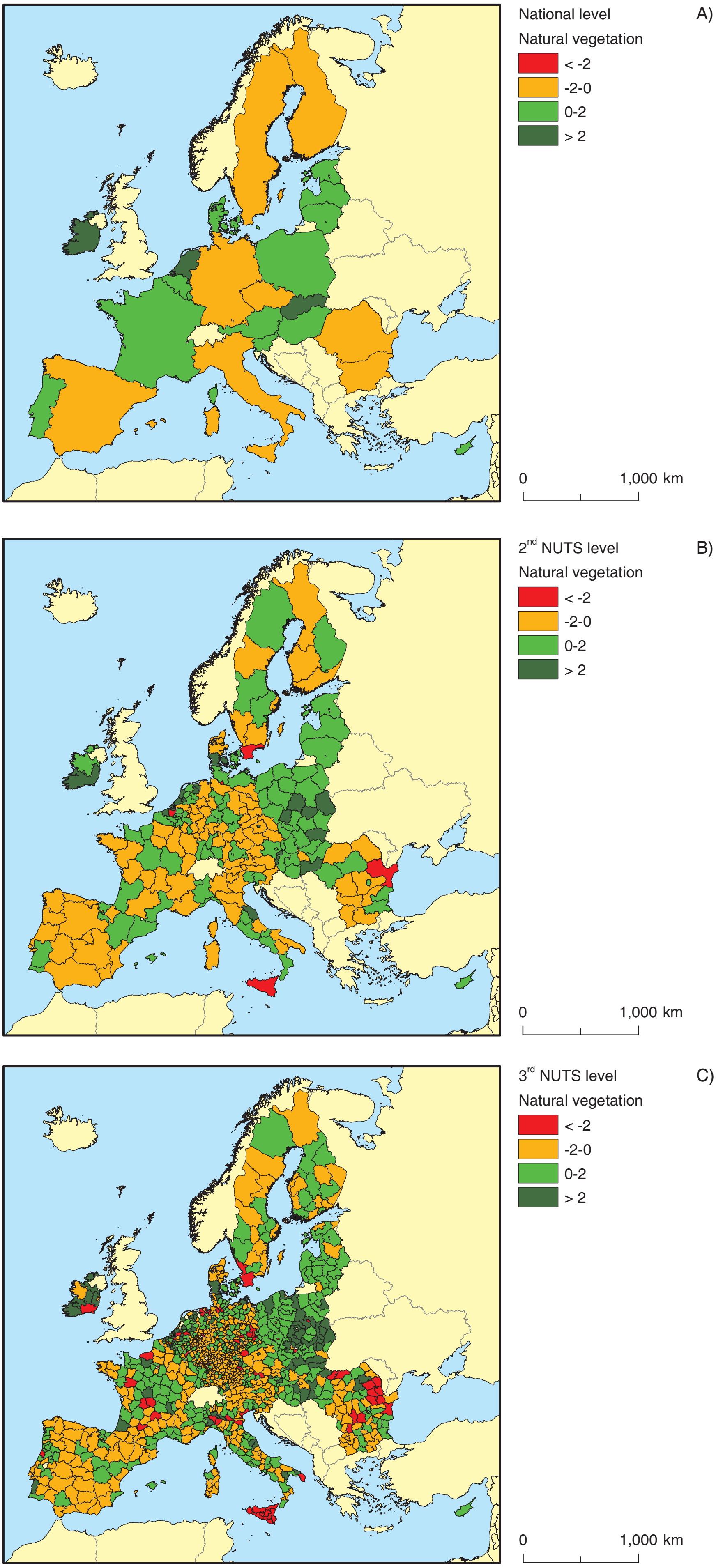
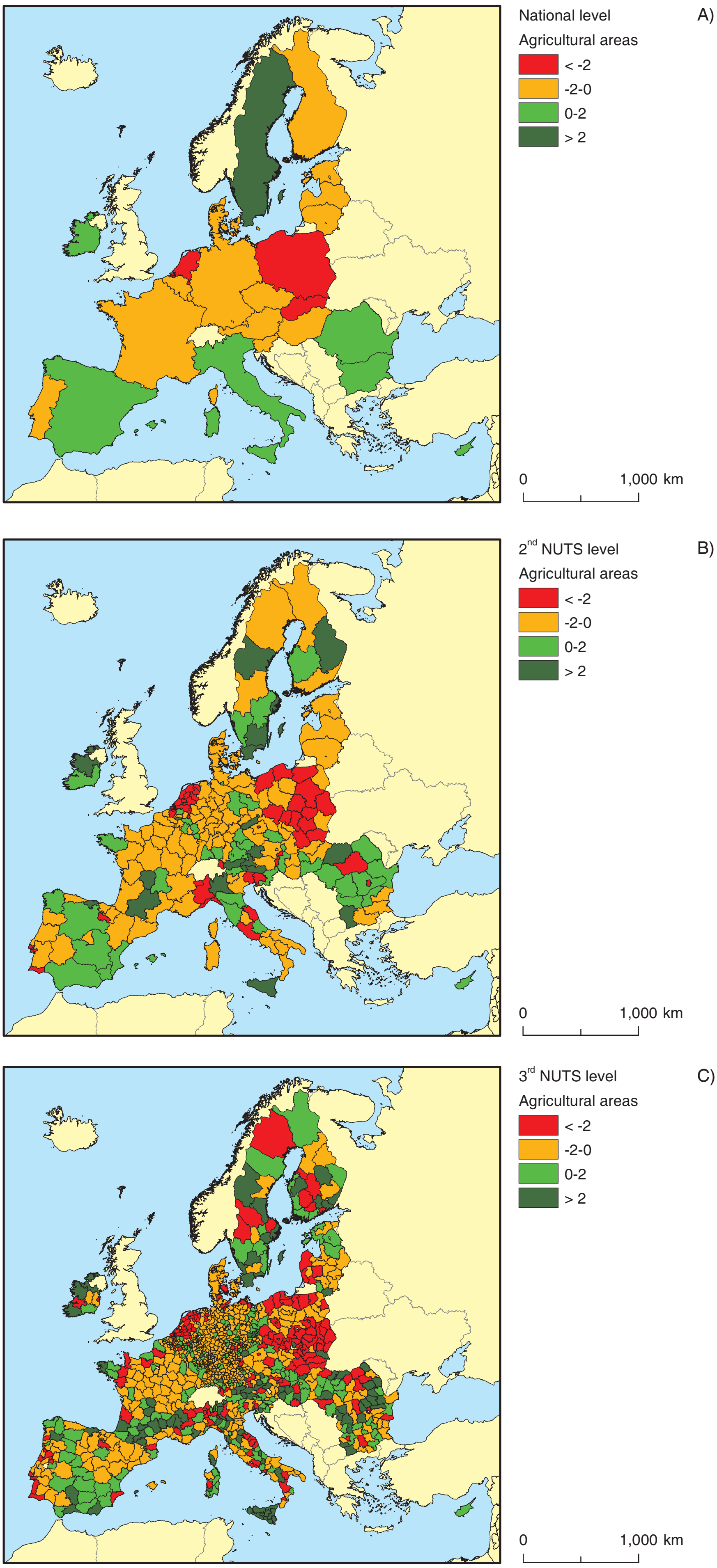


Figure 1. Net land cover change (NET%) for CORINE class 'Forests and Semi-natural areas' within the Natura 2000 sites at A) National B) NUTS 2, C) NUTS 3 level.



nor losses (-2%<NET% <0%) were recorded in fourteen Member States, minor expansion (0%<NET% <2%) in six states and, finally high expansion (NET%>2%) was recorded only in Sweden (Figure 2a). In general terms, we notice that expansion in 'Forests and Semi-natural areas' class is usually accompanied with a corresponding loss in the 'Agricultural areas' class and vice versa. This pattern is found in nineteen Member States, while in Finland, Germany and Czech Republic both 'Forests and Semi-natural areas' and 'Agricultural areas' exhibited minor losses within their national network of Natura 2000. Ireland and Cyprus were the only states where expansion was recorded in both CORINE classes.

Generally, this 'trade-off' pattern between 'Forests and Semi-natural areas' and the 'Agricultural areas' class is also apparent at the second as well as in third level of NUTS. Indeed, the majority of provinces with high losses of 'Agricultural areas' are found in Eastern Europe (Poland, Slovakia and Romania), in Belgium and Netherlands as well as in northern and central Italy (Figure 1b), i.e., in regions where high expansion of 'Forests and Semi-natural areas' were recorded. On the contrary, in Sicily, Romania, Southern Sweden and in other scattered provinces in the EU 'Agricultur-

Figure 2. Net land cover change (NET%) for CORINE class 'Agriculture areas' within the Natura 2000 sites at A) National B) NUTS 2, C) NUTS 3 level.

al areas' seem to expand at the same part of the Natura 2000 network where natural vegetation is lost.

Conclusions

The land cover types describe landscapes by taking into account the observed physical and biological cover of the earth as well as its man-made features. In our analyses, we calculated land cover changes within the Natura 2000 network for 'Forests and Semi-natural areas' and 'Agricultural areas'. The former is associated with the majority of the natural habitat types that the Natura 2000 network was established to conserve. The latter has been identified as the most common cause of habitat loss around the world (Tilman et al. 2001), although in the European continent agriculture could vary from mild (beneficial for biodiversity) to very intense (adverse for biodiversity) (Henle et al. 2008). The results of this study, conducted within the Natura 2000 sites and at three different levels of the administrative scale are consistent with previous studies showing that a major landscape transformation is occurring in Europe characterized by an extension of forested areas, a decrease in agricultural land and finally, an increase in artificial areas.

Our analyses at the National level revealed that although the majority of the EU Member States showed a positive change in preserving vegetation coverage inside the network, several examples of the opposite were also detected. Interestingly, a trade-off between the 'Forests and Semi-natural areas' and 'Agricultural areas' was detected in the majority of the states. This trade-off, which dictates that an expansion in 'Forests and Semi-natural areas' class in a terrestrial unit, is usually accompanied with a corresponding loss in the 'Agricultural areas' class and vice versa, is also detected in the rest of the administrative levels. At first glance, this outcome does not violate Natura 2000 networks' designation philosophy, which is not based on the exclusion of human activities but it rather gives the framework to regulate them within the sites.

However, we should also take into account, that apart from the legislative framework set by the Natura 2000 network, a variety of other policies implemented at various levels of administrative scales, such as the Common Agriculture Policy (CAP), transport, planning, development or energy policies strongly influence economic activity and land use changes at the networks' areas (Paloniemi et al. 2012). Furthermore, there are also differences in the geographical, socio-economic, historical and political factors as well as in the EU accession date or the designation date for each site in each Member State. As a result, although the network throughout the whole EU territory was designated under a common policy framework, several Member States, as well as provinces and smaller administrative units presented a variety of different signatures of land cover transformations. As a result, several administrative units with high losses of natural habitats emerge when we focus our analyses on finer administrative levels. This outcome dictates that efforts should be made to implement policies affecting the land cover transformation in a regional and/or local levels (Tzanopoulos et al. 2014 this book). As far as 'Forests and Semi-natural areas' are concerned, in the NUTS 2 level only five provinces in the territory exhibit high percentages of loss. Since the trade-off with the 'Agricultural areas' is valid for these provinces, the regulation of agricultural activities in combination with the environmental policies could be proved critical for the restoration of their natural habitats. In general terms, this kind of approach is also valid at the local level.

To sum up, analysing land cover transformation in the Natura 2000 network at three administrative levels revealed several hot spots of natural habitat loss within the network and around the EU. In these areas, the need for an integrated approach in EU policies through the combination of environmental and other policies regulating agricultural, forest and land cover management is even more prominent.

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Fragmentation across spatial scales

Anna V. Scott, Konstantinos Touloumis, Veiko Lehsten, Joseph Tzanopoulos, Simon G. Potts

Introduction

Habitat availability and connectivity are imperative structural elements in the maintenance of biodiversity, having a major impact on the overall resilience of populations and communities. Habitat loss and fragmentation usually occur together, combining to result in a decrease in the size and connectivity of habitats, and an overall drop in biodiversity (Collinge 1996). Increases in fragmentation are thought to have a significant effect on species losses (Saunders et al. 1991), whilst increasing the density and connectivity of habitats can increase the numbers of some declining species (Davies et al. 2005). Measures of habitat loss and fragmentation have been used as a proxy for measuring biodiversity losses (Butchart et al. 2010). Habitat loss and fragmentation data calculated across temporal scales, together with species abundance data, can enable an assessment of risks to biodiversity within the landscape. Comparisons across countries or regions can then be made.

Declining biodiversity is still very much on the European and international agenda with several new biodiversity targets set for 2020. The EU Biodiversity Strategy's second 2020 target (European Commission 2011) is that "ecosystems and their services are maintained and enhanced by establishing green infrastructure and restoring at least 15% of degraded ecosystems" suggesting that this will "ensure better functional connectivity between ecosystems within and between Natura 2000 areas and in the wider countryside". The Convention on Biological Diversity's (CBD) Aichi Targets focus on resilience, sustainability and ecosystem services; habitat loss and fragmentation are important aspects of this. Target 5 in particular states that: "by 2020, the rate of loss of all natural habitats, including forests, (should be) at least halved and where feasible brought close to zero, and degradation and fragmentation (should be) significantly reduced" (Convention on Biological Diversity 2010).

By examining fragmentation across different spatial and temporal scales the risks to biodiversity can be highlighted. By observing temporal trends policy makers and practitioners are better able to understand the way in which their sites, networks, regions and countries might be changing, and highlight areas, habitats and species that are at greatest risk. Matching spatial and temporal scales to species and habitats through scale aware policies and management strategies can improve the effectiveness of nature conservation in the future. Regions that are susceptible to further fragmentation and potential biodiversity losses can be targeted for restoration or conservation management.

Example: Multi-scale analysis of natural grassland in the European Union

Methods

Historic CORINE Land Cover Maps from the European Environment Agency provide a solid information base on which to examine changes in the structure of the landscape and highlight the most vulnerable regions. Our work focused on the changes in abundance and connectivity for functional Landcover types between 1990 and 2006. 'Landcover types' were used as a proxy for 'habitats' since complete data on habitats across the EU is not yet available. Different scales were investigated, with a particular focus on and around Natura 2000 sites.

The abundance of landcover types was calculated directly from the CORINE dataset. Fragmentation of the CORINE landcovers was examined using an established method of analysis: a Morphological Spatial Pattern Analysis (MSPA) performed by GUIDOS (http://forest.jrc.ec.europa.eu/download/software/guidos/) (Soille and Vogt 2009). MSPA-GUIDOS segments the land cover data into mutually exclusive feature classes (see Table 1) based on their geometry and connectivity. 'Foreground' is considered the area covered by the respective landcover, whilst 'background' is all other land covers. 'Edges' were defined at 100m, with 'core' being un-isolated areas inside of the 100m edge. By using GUIDOS-MSPA an assessment can be made about how fragmented the landscape is based on how many 'islets', 'branches' and 'core' areas there are. All results were then amalgamated at two administrative scales called the Nomenclature of Territorial Units for Statistics (NUTS); NUTS 3 is equivalent to a Local Authority in the UK, and NUTS 0 is equivalent to a Country scale.

Scaling of fragmentation across spatial and temporal scales is explored in this example using natural grassland, which is a critically important landcover throughout Europe. Natural grassland data are readily found in the historic Corine Landcover map for 1990 and 2006 and is defined as 'low productivity grassland, often situated in areas of rough, uneven ground, and frequently including rocky areas, briars and Heathland'. An example is given at a single Natura 2000 site (*Llanos De Zorita Y*

Table 1. Description of Morphological Spatial Pattern Analysis (MSPA) classes.

MSPA Class	Description
Core	Interior foreground area excluding foreground perimeter
Islet	Disjoint foreground object and too small to contain core
Loop	Connected at more than one end to the same core area
Bridge	Connected at more than one end to different core areas
Perforation	Internal foreground, object perimeter
Edge	External foreground, object perimeter
Branch	Connected at one end to Edge, Perforation, Bridge or Loop

Embalse De Sierra Brava in Central Spain), which shows the basic segmentation of the land cover into the different feature classes (Core, Islet, Perforation, Edge, Loop, Bridge, and Branch).

The two maps (Figure 1 and Figure 2) show a snapshot of Natural Grasslands in 1990 and in 2006. It is clear that some areas of 'core' land cover have declined between 1990 and 2006, and what was once a large area of connected natural grassland is now two unconnected areas, with only a very small region of natural grassland connecting them (purple circles (1) in Figure 1 and Figure 2). Some connections in the north of the site have been strengthened between 1990 and 2006 (orange circles (2) in Figure 1 and Figure 2). However many connections with natural grassland in other Natura 2000 sites and with the wider landscape are few and far between (pink circles (3) in Figure 1 and Figure 2 highlight a narrow connection with another Natura 2000 site).

The proportions of natural grassland categorised as 'core' or 'bridge' were further analysed inside and outside of Natura 2000 sites. This was in order to understand how Natura 2000 sites are protecting 'core' or connecting areas of natural grassland compared to the wider landscape. Results of paired t tests at the NUTS 0 (country) level are displayed below in Table 2. 'Core' areas of natural grassland were found to be significantly higher within Natura 2000 sites; however, there was not a significant difference in the proportion of 'bridge' structures that were protected. At the Natura 2000 level there are significantly more 'core' natural grasslands. While there may have been a focus on 'core' areas when designating sites with natural grasslands, the important message is that Natura 2000 sites are protecting a significantly higher proportion of natural grassland compared to areas outside Natura 2000.

Results at the site level can be extremely helpful for managers to understand how their site has changed, and how well different land covers connect with each other and

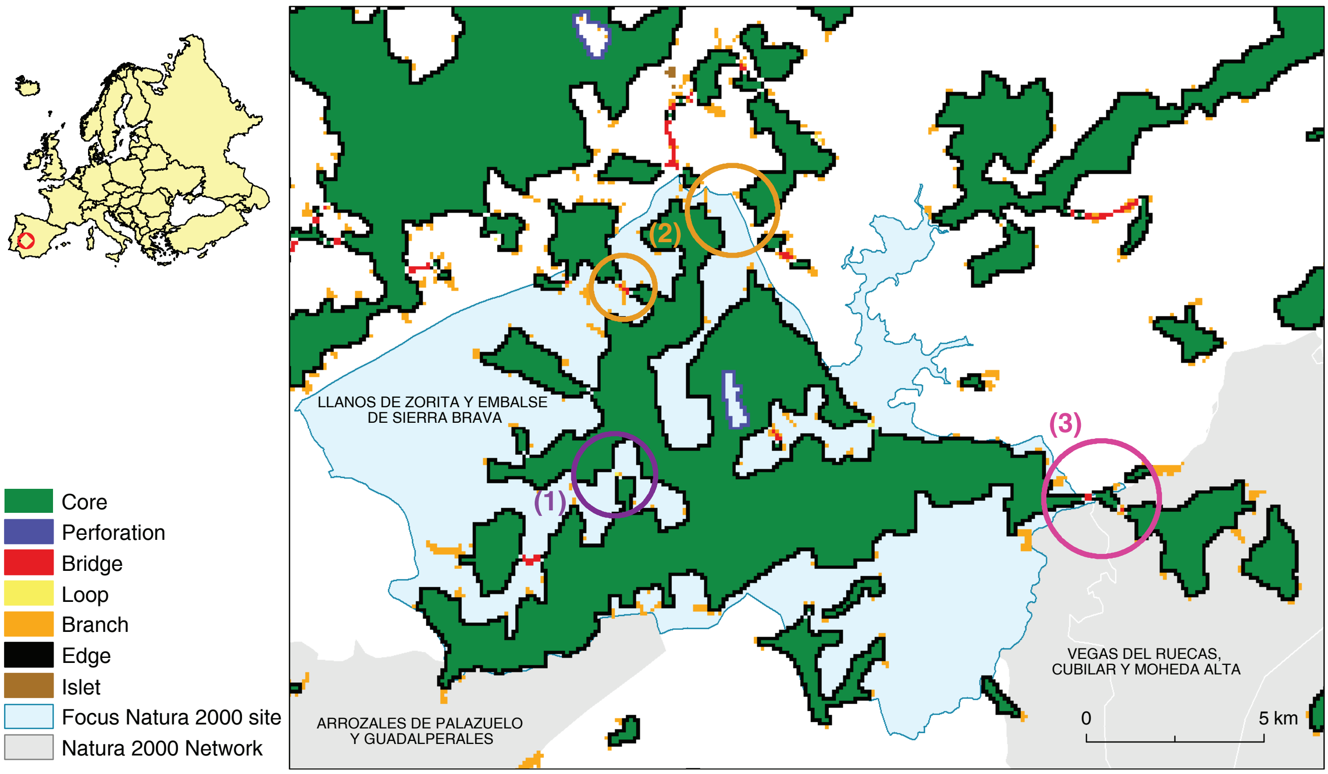

Figure 1. GUIDOS-MSPA analysis using CORINE Land Cover Map 1990 (Copyright European Environment Agency). Purple circle (1) demonstrates fragmentation; Orange circles (2) demonstrate connectivity; pink circle (3) demonstrates fragmented connections between Natura 2000 sites.

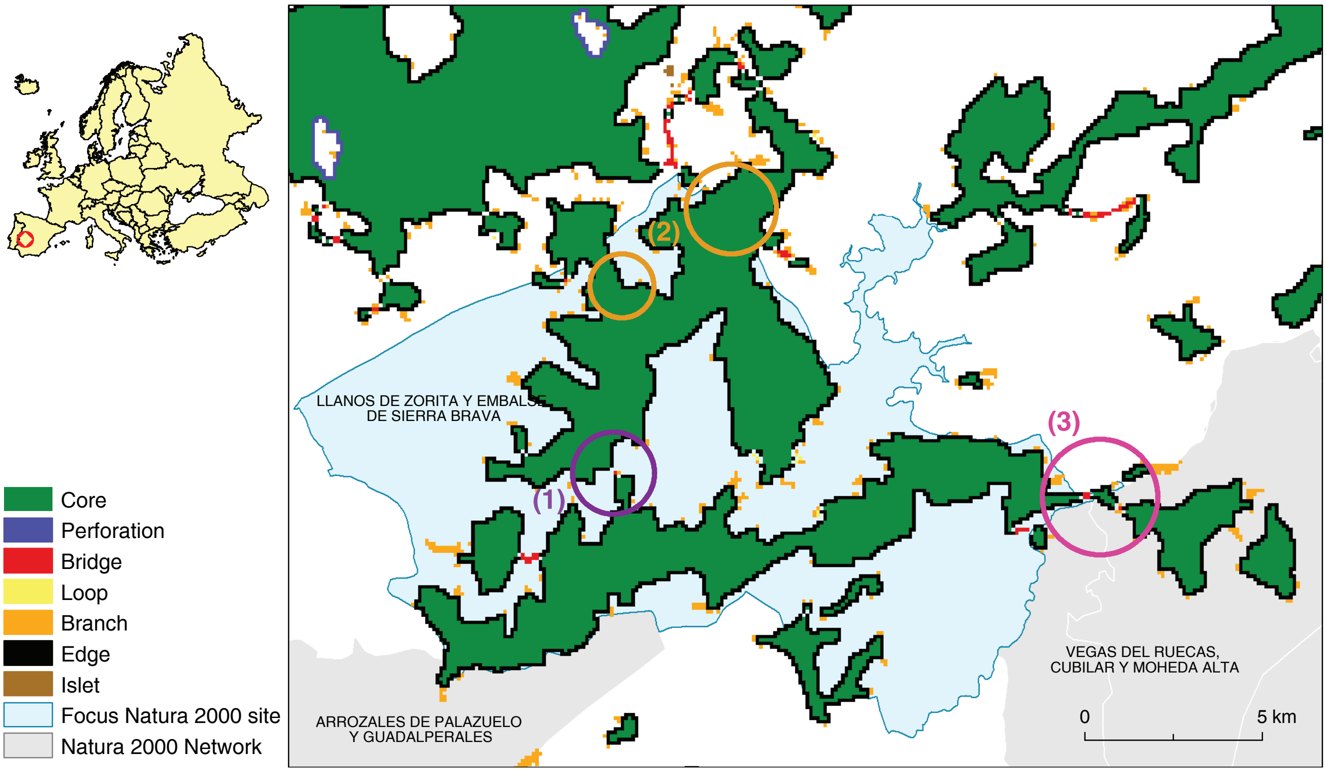

Figure 2. GUIDOS-MSPA analysis using CORINE Land Cover Map 2006 (Copyright European Environment Agency). Purple circle (1) demonstrates fragmentation; Orange circles (2) demonstrate connectivity; pink circle (3) demonstrates fragmented connections between Natura 2000 sites.

the wider landscape. Trends at higher levels and wider scales can also provide valuable information, particularly administrative levels, where policies are designed and implemented. The maps in Figures 3 and 4 demonstrate the amalgamation of these results at two administrative levels within the EU: NUTS 3 and NUTS 0. Changes across a temporal scale between 1990 and 2006 are illustrated.

Across Europe natural grasslands decreased by approximately 2.4% (1900 km^2) between 1990 and 2006. While overall changes in land cover at both NUTS 3 (Figure 3) and NUTS 0 (Figure 4) have remained low (< ±4%), there have been some larger changes in 'core' structures in several countries. For example, at a NUTS 3 level (Figure 3) many localised areas in the Netherlands have experienced an increase in 'core' and 'bridge' structures. This suggests that patches of natural grassland are becoming larger and more connected within these regions. However, there are also NUTS 3 regions in the West of France, for example, where the proportion of 'core' has dropped and there has been an increase in the proportion of 'bridge' structures (Figure 3). This pattern of change suggests that the land covers in these subregions are becoming perforated and remaining connected despite a drop in 'core' areas.

When up-scaling again to a NUTS 0 level (country by country) the changes seen at the site level and the NUTS 3 level become much less clear, with few changes above ±4% in any of the land classification categories (Figure 4). For example, at a NUTS 0 level the Netherlands experiences a slight decrease in 'core' natural grassland, despite large parts of the country showing an increase in 'core' at the NUTS 3 level. Similarly, there appears to be no changes in the proportion of 'core' land covers in France at a NUTS 0 level, despite large changes at a NUTS 3 level. The increases and decreases experienced across the local scales (NUTS 3) have evidently cancelled each other out at a national scale (NUTS 0). These differences across spatial scales are known as non-linearities.

Implications and policy recommendations

Fragmentation and habitat loss are major threats to biodiversity and should be brought to a halt. The Con-

Table 2. Summary of paired T-tests for MSPA-Guidos 'core' and 'bridge' Landcover structures inside and outside of Natura 2000 sites.

		Mean Proportion	Significance level α
'Core' natural grassland	NUTS 0	0.0107	0.0024
	Natura 2000	0.0252	
'Bridge' natural grassland	NUTS 0	0.0007	0.0705
	Natura 2000	0.0017	

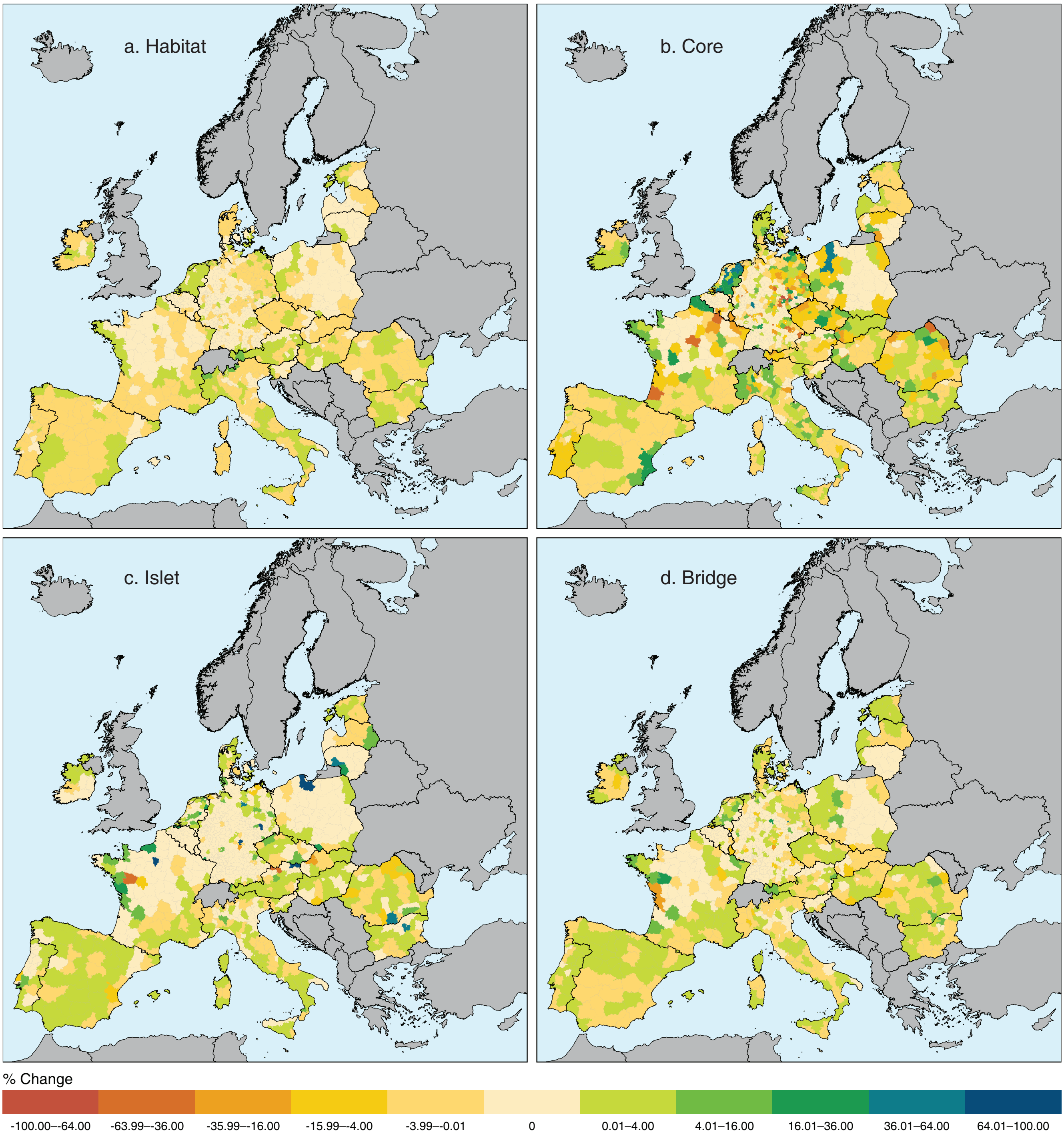

Figure 3. a. Changes in % natural grassland per NUTS 3 between 1990 and 2006 CORINE Land Cover Map (LCM) (Copyright European Environment Agency). b., c. and d. Changes in % of natural grassland classified as 'core', islet and 'bridge' between 1990 and 2006 CORINE LCM (Copyright European Environment Agency).

vention on Biological Diversity and the EU's Biodiversity targets 2020 are beginning to address fragmentation and habitat loss; however, there is much more work to be done. Targeting areas that are vulnerable to fragmentation, and monitoring ongoing fragmentation are important tools in reducing negative effects on biodiversity. The way in which fragmentation is examined, and the scale at which it is examined, can have a significant effect on the results. As in the example given here, upscaling results from a local scale to regional or national scales produces different perspectives. In the Netherlands at a national (NUTS 0) scale there appears to be an overall increase in natural grassland, along with an increase in 'islet' and 'bridge' structures, but a reduction in the proportion of 'core' areas. This suggests that across the whole of the Netherlands there has been a reduction in 'core' areas of natural grassland, compensated with an increase in connectivity, demonstrated by an increase in 'bridge' structures. However, at the regional (NUTS 3) scale there are many places where there has been a big increase in the proportion of 'core' areas; this is at odds with the national picture, and is known as a non-linearity between

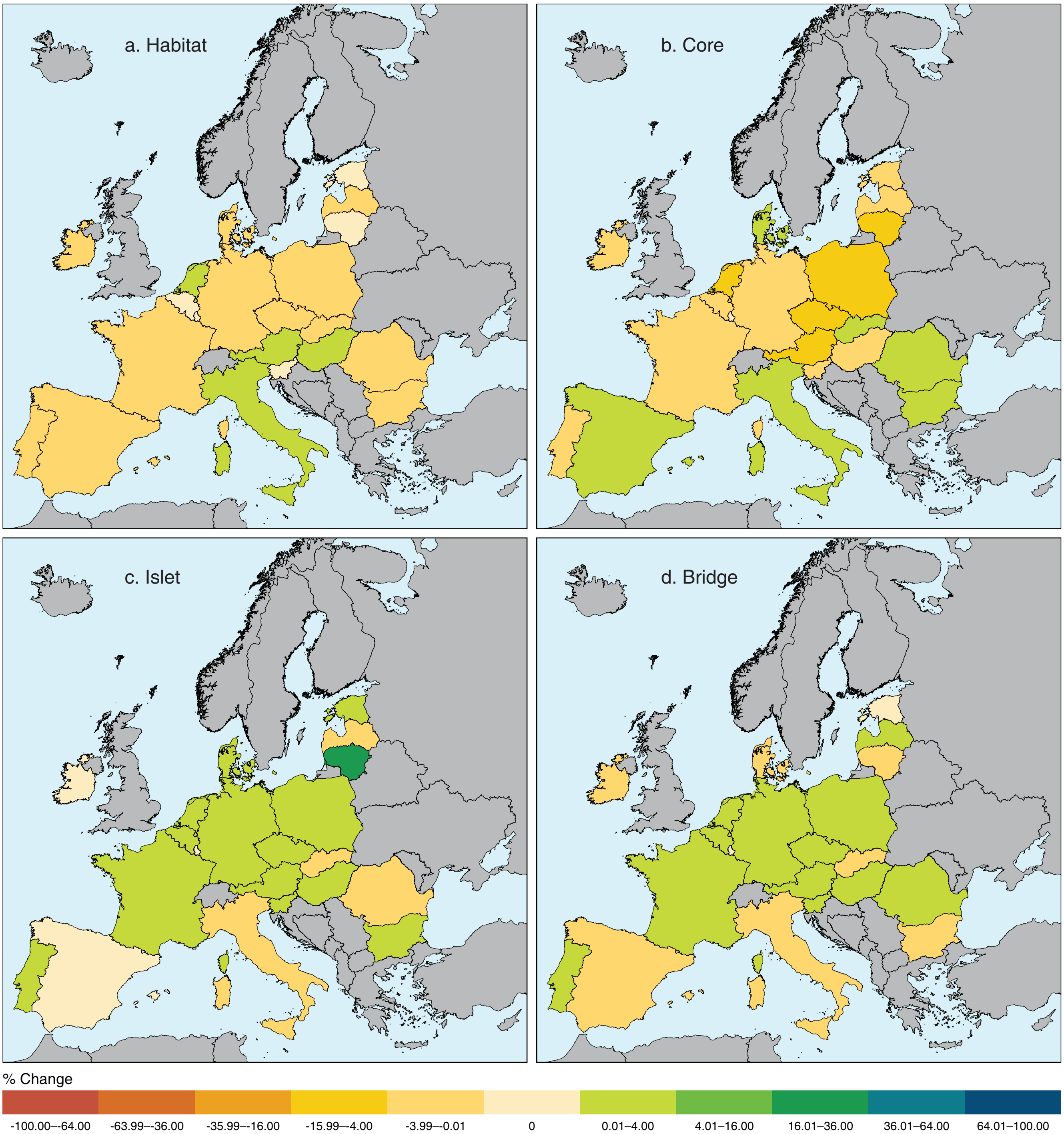

Figure 4. a. Changes in % natural grassland per NUTS 0 between 1990 and 2006 CORINE LCM (Copyright European Environment Agency). b., c. and d. Changes in % of natural grassland classified as 'core', islet and 'bridge' between 1990 and 2006 CORINE LCM (Copyright European Environment Agency).

scales. Therefore to gain a full understanding of the changes in fragmentation, it should be examined at different spatial and temporal scales.

Regional, national and international scales should be fully taken into account during conservation planning because context and connectivity with the wider landscape has a significant impact on individual sites, networks and species. Similarly, landscape and conservation policies, which target fragmentation at different administrative scales (e.g. NUTS scales) should allow for flexibility at local and site scales, where patterns may vary from national and regional trends. New landscape and conservation initiatives (e.g. agri-environment schemes) should cover a long time scale in order to promote consistency. Monitoring should also be examined over multiple time scales in order to examine both short-term and long-term changes and fully assess both the positive and negative effects on species and landscapes.

Increasing and improving guidance on green infrastructure strategies and spatial planning across the EU may encourage a greater quantity and quality of such projects. Green infrastructure is particularly important in areas where continuing development threatens habitat connectivity and integrity (see Kettunen et al. 2014 this book, Paloniemi et al. 2014 this book). Well implemented spatial planning and green infrastructure strategies may help to limit further fragmentation of landcovers and natural habitats.

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European projections of habitats and carbon stocks: Negative effects of climate and positive effects of CO_2 changes dominate, but land use is also of importance

Veiko Lehsten, Anna V. Scott

Introduction

The vegetation of Europe has changed drastically in the last few decades together with land use practices which have altered significantly in many parts of Europe, as society places increasing demands on natural systems. Many agricultural changes, for example, have been driven by technological developments which have increased harvests in some areas and thereby decreased agricultural demands in other parts. As we look towards the future, social, economic, and climatic drivers will continue to affect the form and structure of European landscapes. Many regions are predicted to experience warmer weather with more extreme weather events, associated with increasing CO_2 levels (IPCC 2007).

Predicting future changes in European land cover and carbon stocks is essential for understanding how these changes may affect both biodiversity, and social and economic development. Current distributions of habitats are far from their natural state within the European Union; to generate reliable projections of them it is necessary to take land use, current tree species distribution, land use change and climate change into account. The Intergovernmental Panel on Climate Change (IPCC) provides a framework under which researchers can coordinate their work using similar assumptions. These scenarios are subsequently used by climate and land use modelling groups to project changes in climate and land uses. To generate a tool to assess the likely development of European vegetation, and the carbon stocks therein, we used a modelling approach to predict future habitat distribution and carbon stocks under different socio-economic scenarios across Europe. While some parts of the analysis were carried out at a 1 km scale, the modelling of carbon stocks was carried out at a resolution of 0.5 degree. We analysed the effect of three potential drivers of change: climate change, land use change and CO_2 increase. We consider the European continent as our case study area and perform the analysis at this scale; we present detailed results of a small region to illustrate the generated outputs.

Methods

The dynamic vegetation model LPJ-GUESS (Smith et al. 2008) in combination with the land use projection model Dyna-CLUE (Verburg et al. 2010) was used to predict changes in the distribution of eight habitat types (croplands, urban areas, Mediterranean forests, grasslands, needle-leafed forest, broad-leafed forests, mixed forests and shrublands) between 2000 and 2050. LPJ-GUESS is a flexible framework for modelling the dynamics of terrestrial ecosystems from landscape to global scales. It consists of a number of sub-modules containing formulations of subsets of ecosystem processes at a defined spatial and temporal scale. The model is primarily driven by the climate variables temperature, short wave radiation, precipitation and soil type. The different processes modify state variables such as net primary productivity, evapo-transpiration, run-off and carbon sequestration. All simulated processes are described in detail in Sitch et al. (2003) and Smith et al. (2008).

The model output contains annual values of ecosystem state variables such as vegetation composition, leaf area index and biomass (in kg carbon per square metre) and carbon pools for each simulated species or at a stand level. The carbon pools contain the living vegetation, the litter (dead, but non-decomposed vegetation) and the carbon in two different carbon pools of the soil (the fast decomposing and the slow decomposing carbon pool of the soil).

The climate projection data used to drive LPJ-GUESS were generated by the Hadley Centre (HadCM3, http://www.metoffice.gov.uk/climate-change/resources/hadley, Gordon et al. 2000) and by the National Center of Atmospheric

Research (PCM1: http://www2.cgd.ucar.edu/) and based upon the two SRES AR4 scenarios A2 (representing a regionally orientated economic development) and B1 (representing global environmental sustainability). These two scenarios were chosen since they represent two extremes of expected climate change (IPCC 2007).

In the example given here, the LPJ-GUESS model was adapted to account for socio-economic processes, such as land use activities using the approach by (Lindeskog et al. 2013). The Dyna-CLUE model (Verburg et al. 2010), which simulates land use changes based on CORINE Land Cover types, was used as an input. It is based on the dynamic simulation of the competition between land uses. The main driving factors of the Dyna-CLUE model are demography, overall economic development, technological change and policies. The allocation rules are configured with respect to each country to account for country specific context and land use preferences.

The LPJ-GUESS model is initialised with the 20 major tree species according to the tree species map of the European Forestry institute (Brus et al. 2011). We performed one set of simulations in which we assume that the tree species choice remains the same as it is for now (this set will be called EFI thereafter) and another one where we assumed that the forest will be reforested by the species which would naturally occur (later on called Natural). This reflects current developments in some EU member states where foresters are obliged to use native species.

For each 0.5 degree longitude / latitude grid cell inside LPJ-GUESS, the proportion of areas covered by each of the different land use categories is calculated from the Dyna-CLUE scenario data (which has as a starting point the CORINE land cover data set, Büttner et al. 2004) and handed over to LPJ-GUESS. All changes from one land use type to another are explicitly simulated. For example, if forest areas are transformed to urban areas, first a complete harvest of the forest is simulated before the different land use is taken into account. Hence processes such as land abandonment, urbanisation and land use intensification are taken into account to the extent that they are present in the land use scenarios generated by the Dyna-CLUE model.

The habitat classification is performed according to the dominance of certain plant types (for needle-leafed, grassland broad-leafed, Mediterranean and mixed forests) or land use (e.g. for pastures cropland and urban land uses). Since for a number of applications the habitat classification at the scale of the land use data is of interest, we scaled-down the land cover projections from 0.5 degrees to the 1 km scale as a post-process. By assigning one of the four LPJ-GUESS land use types to the 1 km² cells (or one of the habitats), maps can also be generated containing any of the simulated parameters: biomass, leaf area index or net primary productivity per species at this fine scale.

While the distribution of the major habitats is one important result, the changes in carbon stocks is another. We analyzed the carbon pools in standing biomass as well as in the soil and the net primary productivity (NPP). We used decadal means for all variables. All simulations were repeated including only two of the three potential drivers (climate change, land use change and CO_2 increase) to assess the effect of each driver on the change. The relative contribution of a single variable is subsequently calculated as the difference between the simulated effect of the simulation with all drivers and the simulated effect of the simulation containing the two drivers that are not included. For example, when analyzing the effect of CO_2 increase, we would compare the simulations including all drivers to the simulations including only land use and climate change. To do so, a pan-European sum was calculated as a mean value for the 1990's and as a mean for the 2040's. The total change for each category is calculated as follows (Eq. 1):

$$\Delta_{tot} = \left(1 - \frac{mean_{2041-2050}}{mean_{1991-2000}}\right) * 100\% \qquad \text{Eq. 1}$$

Subsequently, the effect of the single drivers has been calculated according to equation 2.

$$\Delta_{driver} = \Delta_{tot} - \left(1 - \frac{mean_{driver2041-2050}}{mean_{1991-2000}}\right) * 100\% \qquad \text{Eq. 2}$$

Here the variable Δ_{tot} is derived from Eq. 1 and $mean_{driver2041-2050}$ indicates the mean of the variable of interest resulting from the simulation which does not contain the driver in focus.

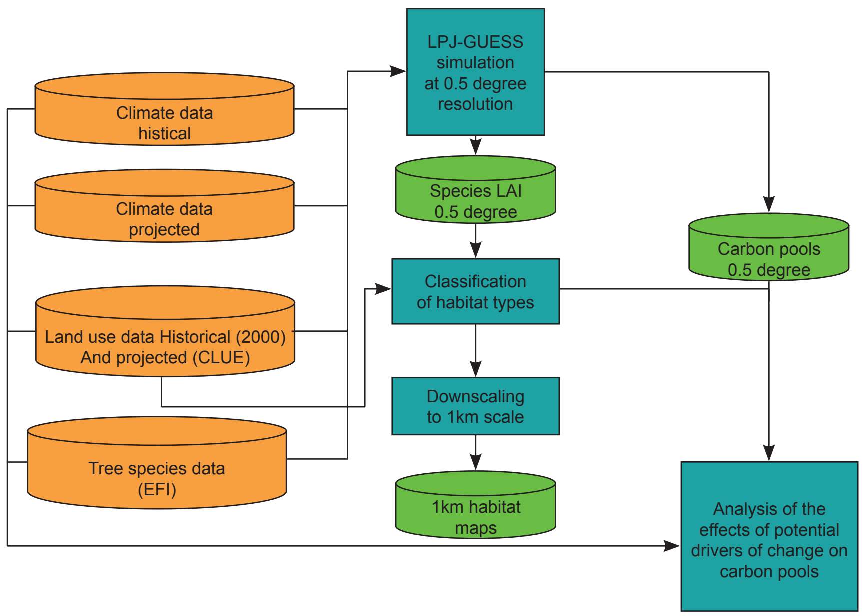

Figure 1. Flowchart of the steps within this study. Orange parts are external data sets, green parts are datasets (round shapes) and models or procedures (rectangles) generated or used in this study.

The overview of the steps involved in this study is given in Figure 1. Orange elements indicate externally generated data that was read in. Green elements indicate parts that were used or generated in this study. The distribution of the habitats at 1 km scale is one output of this study, the other is the analysis of the relative effects of potential drivers of change on carbon pools which has been performed using the carbon pool data at 0.5 degree resolution.

Results

Broad habitat types

Figure 2 displays the results of the simulated habitat types for the year 2001 in the upper panel, and for the year 2050 in the lower two panels in a part of Greece. The resolution is at 1 km. Land abandonment is taking place and grasslands are replaced by Mediterranean forests; another feature is the urbanization and expansion of Athens. When analyzing the fine scaled data, one has to bear in mind that, though the positions are to some extend based on the CORINE data, the simulated changes in Dyna-CLUE (which are translated by LPJ-GUESS into the displayed broad habitat categories) are not precise spatial predictions of where a certain type of change occurs, but should rather be

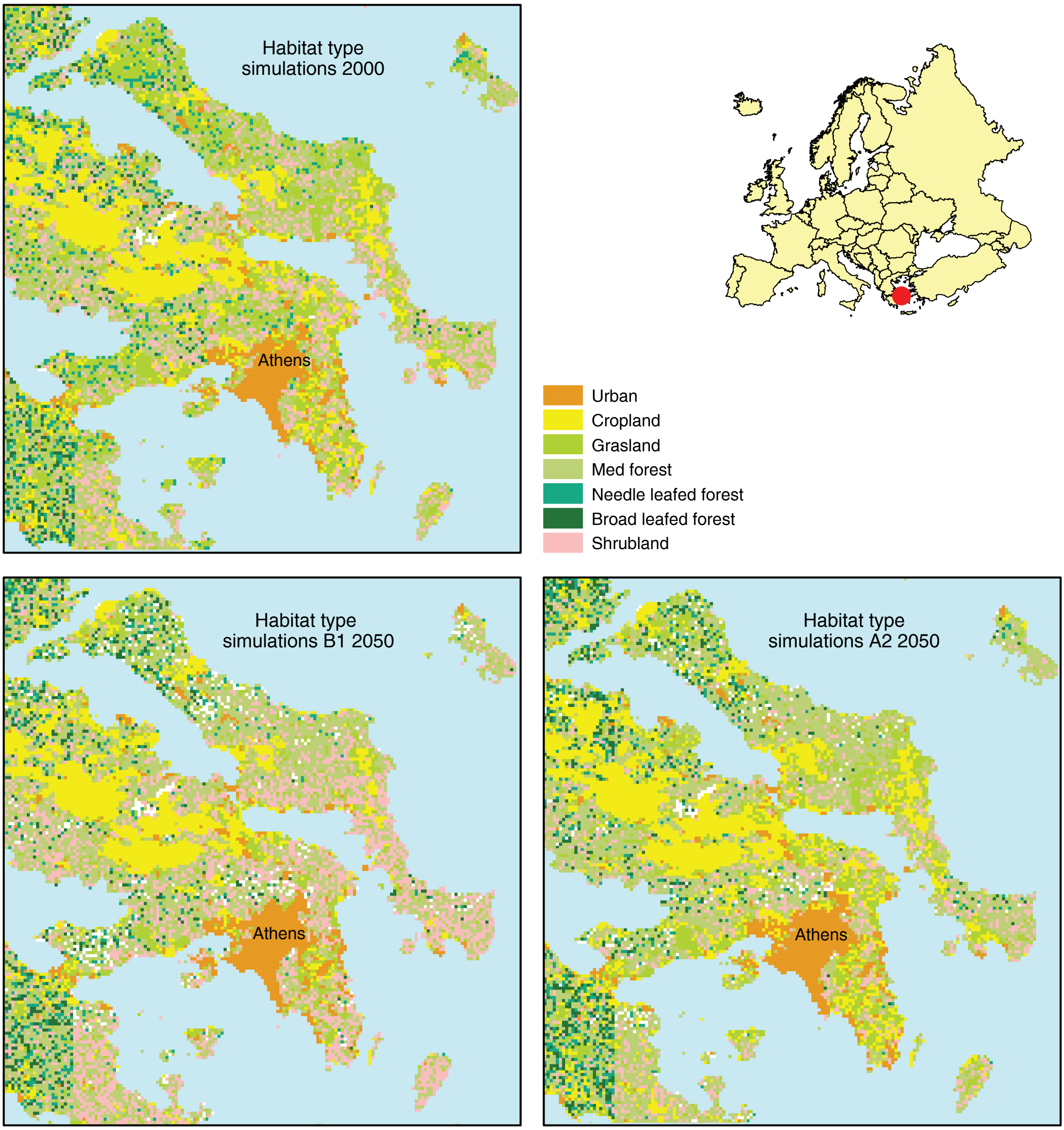

Figure 2. Simulated habitat types for the year 2000 (upper panel) and 2050 (lower two panels) for the area around Athens under the two SRES scenarios A2 and B1. Simulation based on Dyna-CLUE land use data and performed with LPJ-GUESS. Note that the figure displays general features of the current distribution of the habitat types now, and in the projection. The spatial location might not coincide with current distributions.

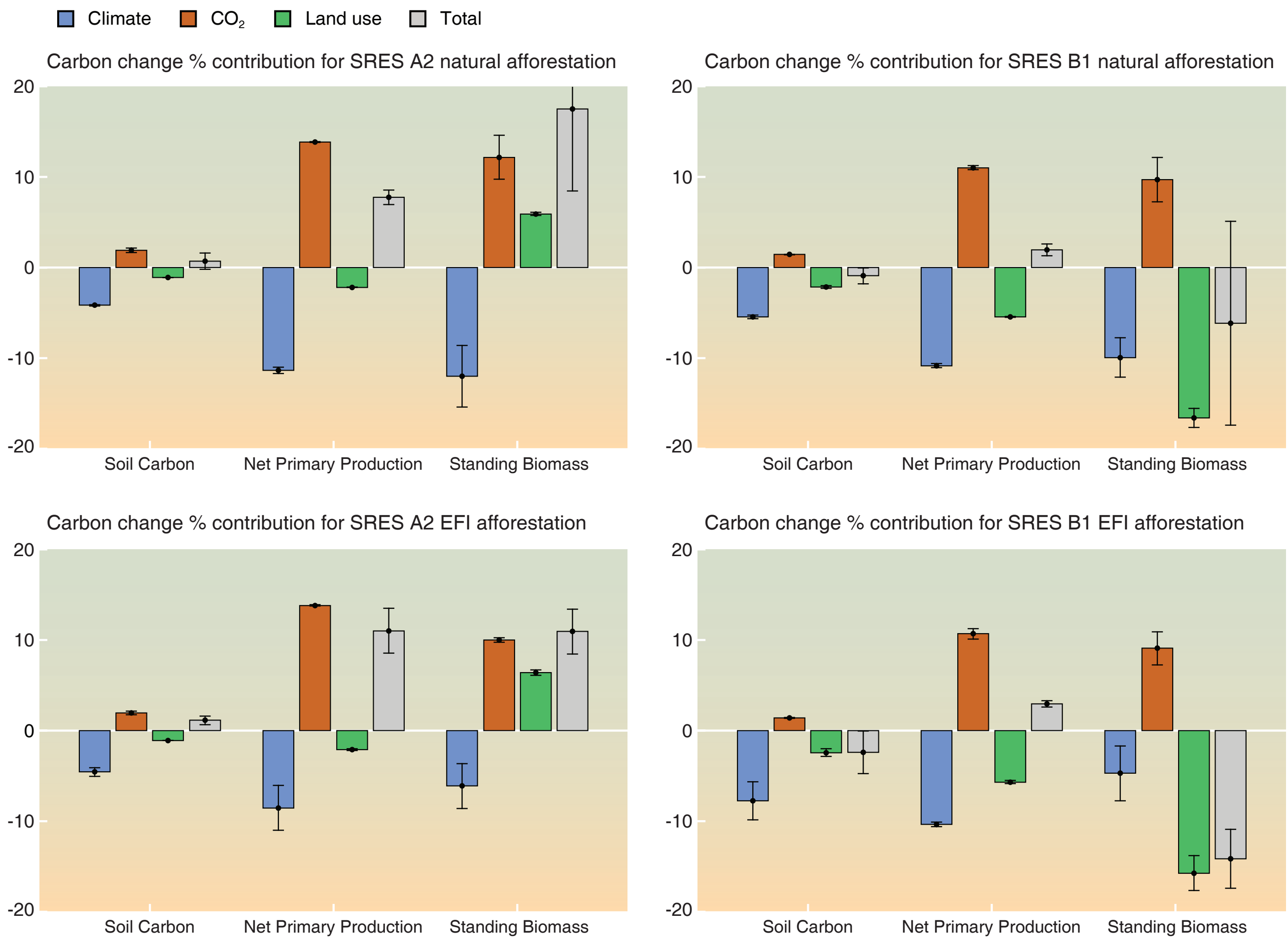

Figure 3. Contribution of the three drivers of change CO_2 increase, climate and land use change to the total change in carbon stocks at a European scale. Displayed are the contributions according to the SRES scenarios A2 and B1 under the re-forestation options EFI and natural reforestation.

seen as general trends. Hence, for the example of Athens, the model projects that the city will expand to some extent and a large proportion of the grasslands will transform into Mediterranean forest. However, the model projection does not explicitly define whether one particular grid cell will change its attribute.

In Figure 3, we present the effect of the different drivers on European soil carbon mass, NPP and standing biomass. CO_2 has a positive, and climate change has a negative, effect on both carbon stocks as well as carbon exchange regardless of the scenario or afforestation option used (EFI or Natural). For all considered ecosystem properties, except standing biomass under the A2 scenario, CO_2 increase is the major driver, while the effect of land use dominates the standing biomass under the A2 scenario. While Figure 2 was generated using the Hadley climate change projections, we also used the PCM1 projection for the estimation of the effects of drivers on the carbon related ecosystem properties to indicate the uncertainty related to the choice of the climate projection (General Circulation Model; error bars).

Discussion

We present a novel approach to combine a land use change model and a dynamic vegetation model to simulate the changes in habitat distribution, carbon pools and fluxes.

This approach is quite straightforward as it is performed offline, and the land use change model (Dyna-CLUE) is executed as a first step, and the dynamic vegetation model (LPJ-GUESS) reads in the results. Hence this does not allow the inclusion of any feedbacks of the two models. This could potentially lead to effects where Dyna-CLUE takes its decision to keep a certain cell as forest based on the assumption that the cell would produce a certain amount of wood while the tree species planted by LPJ-GUESS might be (based on the EFI forest map) growing slower and producing less timber. To avoid these type of errors, the models would have to be fully coupled. However, this would also increase the complexity of the model and may not increase the realism of the results since an increase in complexity also increases the random error of the model.

The simulation shows that in the example area around Athens, the urban areas are expected to increase as well as the Mediterranean forests and shrub lands at the expense of grasslands and croplands. These results highlight a strong threat for biodiversity, since grasslands comprise high biodiversity habitats, but also have implications for fire risk as both shrublands and forests pose a higher fire risk than crops and grasslands. The extension of urban areas is additionally increasing the urban–rural interface, and hence putting more built up areas at risk of fire.

Compared to the information in the Dyna-CLUE map (which for the year 2000 is a rescaled and aggregated CORINE map), our projection retains the areas covered by cropland, meadows and urban areas, but uses a finer classification scheme for areas classified as forested areas or natural areas. In these areas, our map distinguishes between grassland and shrubland or the different forest types. Forests can be classified as grasslands in an early stage of their succession, since from an ecological point of view they represent grasslands in terms of species composition and biomass.

However, not only the habitat distribution but also the carbon pools are projected to change in the near future. Our results show that the increase in CO_2 levels will cause an increase in NPP at a European scale, while climate change has a negative effect on carbon pools as well as fluxes. Both effects cannot be directly mitigated at a regional scale.

On the other hand, our results also show that land use change and the choice of forest species have a strong effect on standing biomass. Both can be directly influenced at any administrative level. Our simulations incorporate land use projections based on current policies and developments. As they can be changed for the future, so can the carbon storage of the landscape.

Zaehle et al. (2007) also used a modelling approach to project European carbon pools, including the effects of climate and land use change, up to the end of the century. Their results regarding the carbon stocks are comparable to our results to the extent that NPP is increasing in all scenarios, though at different levels, at least until 2070. They conclude that the majority of variability lies in the choice of the General Circulation Model (GCM). However this only applies to the time after 2050. The variability in the first half of the century is relatively low compared to the second half. The fact that Zaehle et al. (2007) attribute the highest variability to the choice of the GCM might also be influenced by the fact that only a single land use change model was used (similarly to our study). While currently a large amount of data on climate projection change is available for applications in climate change effect studies, like this one, only a limited amount of land use change projections has been generated. Additionally, there is no central coordination of work, as there is for the climate data given by the Coupled Model Intercomparison Project (CMIP) infrastructure.

To assess the uncertainty attributed to the land use change projections (which has been shown to be of major importance for habitat change), as well to produce more reliable projections on all aspects habitat distribution, biodiversity and carbon stocks, more research leading to a variety of land use change projections similar to the currently available climate change projections is needed.

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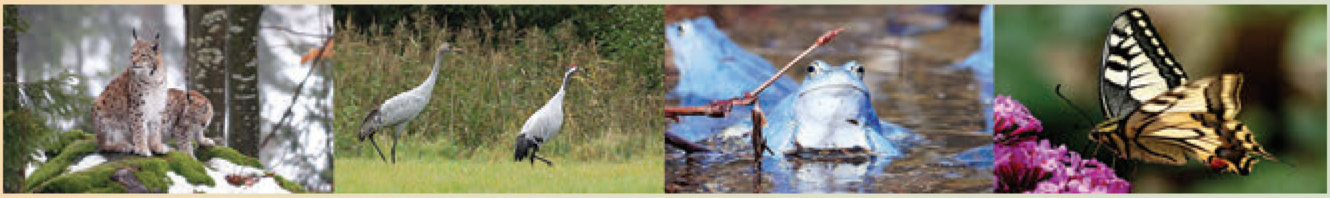
CHAPTER III



Scaling of biodiversity patterns and processes

The scaling of genetic diversity in a changing and fragmented world

Miguel Arenas, Stefano Mona, Audrey Trochet, Anna Sramkova Hanulova, Mathias Currat, Nicolas Ray, Lounes Chikhi, Rita Rasteiro, Dirk S. Schmeller, Laurent Excoffier

Species living in a changing world

Most species do not live in a constant environment over space or time. Their environment is often heterogeneous with a huge variability in resource availability and exposure to pathogens or predators, which may affect the local densities of the species. Moreover, the habitat might be fragmented, preventing free and isotropic migrations between local sub-populations (demes) of a species, making some demes more isolated than others. For example, during the last ice age populations of many species migrated towards refuge areas from which re-colonization originated when conditions improved. However, populations that could not move fast enough or could not adapt to the new environmental conditions faced extinctions. Populations living in these types of dynamic environments are often referred to as metapopulations and modeled as an array of subdivisions (or demes) that exchange migrants with their neighbors. Several studies have focused on the description of their demography, probability of extinction and expected patterns of diversity at different scales. Importantly, all these evolutionary processes may affect genetic diversity, which can affect the chance of populations to persist. In this chapter we provide an overview on the consequences of fragmentation, long-distance dispersal, range contractions and range shifts on genetic diversity. In addition, we describe new methods to detect and quantify underlying evolutionary processes from sampled genetic data.

Spatial and temporal genetic simulation using SPLATCHE2

Computer simulations mimic the processes that occur in the real world and allow us to study which patterns may affect systems. We have developed the program SPLATCHE2 (http://www.splatche.com) (Ray et al. 2010), which performs spatially explicit simulations of genetic data under different environmental scenarios and accounting for recombination, complex migration and long-distance dispersal. As input, the program requires a map (specified by a grid of demes) where the carrying capacity (K) and the migration rate must be user-specified for each deme. Optionally, both K and migration rate can change with time (moreover, a model allowing for different migrations rates in different directions is also implemented). Other important inputs are related with demography (e.g., initial population size and geographic origin, growth rate, total number of generations and a number of demographic models). Then, SPLATCHE2 performs a demographic simulation over the map followed by a coalescent simulation based on user-defined samples (Figure 1). The coalescent simulation just traces the evolutionary history of the sampled genes going backwards in time until their most recent common ancestor. It is followed by a simulation of genetic data (DNA, STRs and SNPs) along the coalescent (gene) genealogy. Although the model makes several assumptions (such as a molecular clock or non-overlapping generations) it is probably one of the most realistic software packages available and has been used in a variety of important publications.

Genetic diversity can be scale-dependent as a consequence of environmental or evolutionary heterogeneities, the former ones being potentially driven by climatic changes, whereas the latter can be driven by natural selection. Thus, geographic barriers, geographic provenance, or migration abilities of the species may increase genetic heterogeneity at various scales. Below, we study a variety of complex evolutionary scenarios with scaling genetic diversity by using our simulation evolutionary framework.

Influence of habitat fragmentation on genetic diversity

Previous studies have suggested that environmental heterogeneity can affect genetic diversity, but these effects were not evaluated at different spatial scales. For instance it is unknown if a given climatic change will equally affect (e.g. decrease) genetic diversity within and between populations, which is fundamental information for nature conservation and management studies, such as to predict the influence of climate change on global and local biodiversity. By using the results from extensive simulations, we address here the influence of fragmented habitats at different scales on the species genetic diversity. Using SPLATCHE2, we simulated range expansions where demes were partitioned into groups (patches) by adding barriers to dispersal. We also included scenarios with long-distance dispersal events, where individuals can migrate to non-neighboring demes. Then, samples were collected within demes, patches, regions and at the global landscape level.

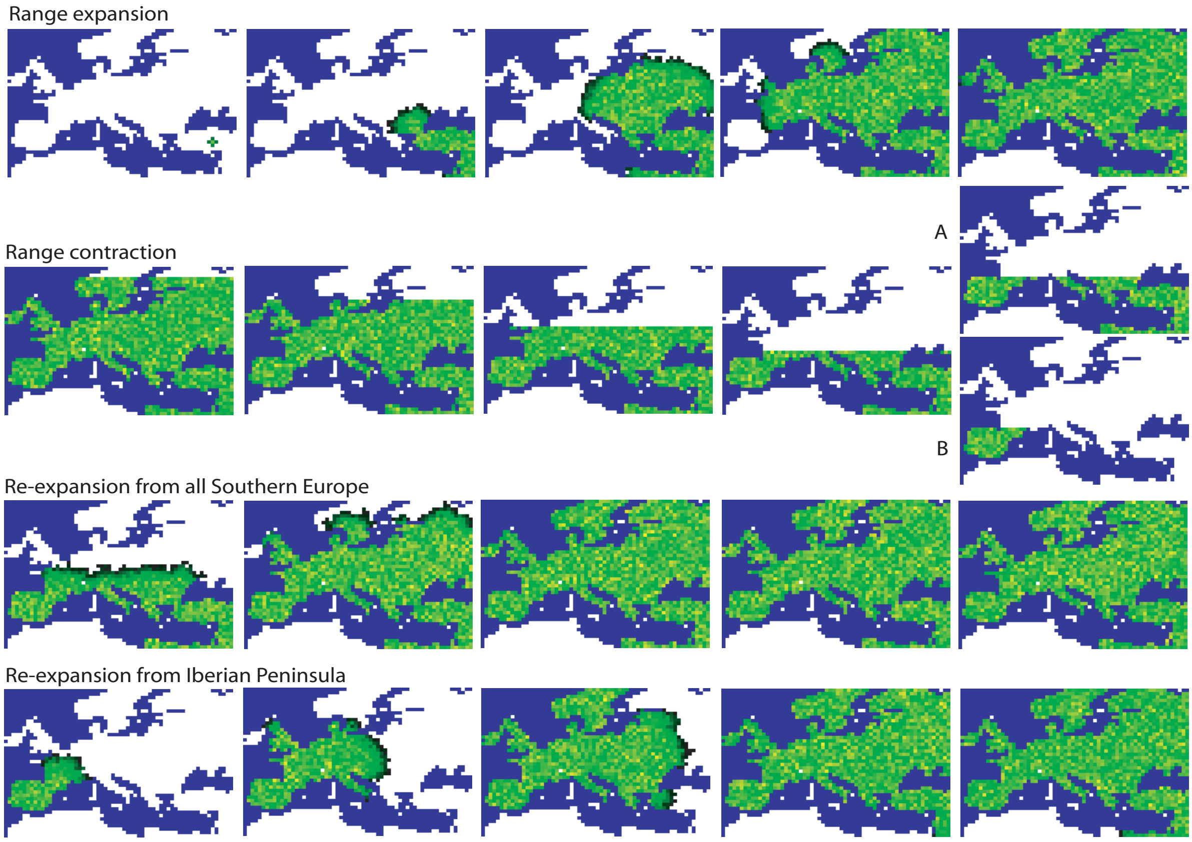


Figure 1. Timeline simulation of complex scenarios of range expansion, range contraction and posterior re-expansion. Each plot corresponds to a snapshot of the program SPLATCHE2. White areas indicate unoccupied demes while green areas represent occupied demes. Snapshots presented at each line differ in 50 generations, see detailed settings in Arenas et al. (2013). At the top, we describe a range expansion over Europe from the Near East. Then, we show a range contraction from the north to the south, which mimic the Last Glacial Maximum period and leads to two situations (as shown on the left of the second row: A: refuge areas cover all southern Europe, and B: there is a single refugium in the Iberian Peninsula. The third and fourth rows show a re-expansion from these two types of refuge areas.

As expected, we found that strong levels of fragmentation result in a severe loss of genetic diversity in the population at a global scale, but we also found that the detection of this decreased diversity requires sampling at different scales (Mona et al. 2014). Moreover, we varied fragmentation intensity at specific time points and we found that local genetic diversity and population differentiation were markedly affected by ancient fragmentation, and much less by recent events (Mona et al. 2014). Our results explain why recent habitat fragmentation does not always lead to detectable signatures in the genetic structure of populations. Conversely, if habitat fragmentation is removed, it also takes a long time to recover lost diversity by natural processes, suggesting that long-term conservation measures (e.g., by restoring gene flow) should be implemented to locally restore previously lost genetic diversity (Mona et al. 2014). We also found that species with long-distance dispersal abilities can, however, migrate across the barriers. As a consequence, their diversity is less influenced by the fragmented landscape.

Influence of range contractions and range shifts on genetic diversity

Range contractions and range shifts may occur as a consequence of temporal climatic fluctuations, depending on the geographical structure of the landscape, the duration of the climatic changes, or the species' dispersal abilities. Under such environmental changes, a common response of species is migration towards more suitable regions. Many studies have analyzed the migration behaviour and spatial distribution of range-contraction and -shifting species; nevertheless, less attention has been paid to the influence of such processes on genetic diversity. We simulated DNA sequence data in populations suffering diverse range shifts and contractions over a landscape constituted by a grid of demes (Arenas et al. 2012). Simulated scenarios of range shifts and range contractions varied according to dispersal abilities and migration patterns. For example fast range contractions (e.g., as a consequence of rapid climate change) may lead to the extinction of populations that do not move. We analyzed genetic diversity of the simulated data. Contrary to our expectations, we found that fast contractions preserve higher levels of diversity and induced lower levels of genetic differentiation among refuge areas than slow contractions towards refuge areas. Thus slow contractions have the highest negative impact on final levels of diversity. We obtain rather different results when the range of species is shifting rather than expanding: fast range shifts lead to lower levels of diversity than slow range shifts. Interestingly, we found that species actively migrating to-

Box 1. Effect of range contractions on current European molecular diversity

The genetic signal of range contractions can be also observed in genetic gradients estimated by principal component analysis (PCA), a method for analyzing patterns of similarity between multiple samples. Initial studies that represented genetic relationships among human populations with PCA revealed the presence of a southeast–northwest (SE-NW) gradient of genetic variation in current European populations, which was interpreted as being the result of a diffusion process of early Neolithic farmers during their expansion from the Middle East. However, this interpretation has been widely questioned, as PCA gradients may occur even when there is no expansion, and because the first PC axis is often orthogonal to the expansion axis (i.e. the main axis of change in levels of genetic diversity is perpendicular to the expansion direction). However, the effect of more complex evolutionary scenarios on PCA, such as those including both range expansions and contractions, had not been investigated.

In a recent study, we (Arenas et al. 2013) performed simulations of range contractions that might have occurred during the last glacial maximum period to better understand the formation of genetic gradients across Europe. In particular, we have simulated range contractions of human Paleolithic populations and admixture between Paleolithic and Neolithic populations over Europe (see Figure 1). The simulations were performed for diverse levels of admixture and under two range contraction scenarios where the refuge areas were either over all southern Europe or only in the Iberian Peninsula (see Figure 1). We observed that the first PC (PC1) gradients were orthogonal to the expansion, but only when the expansion was recent (Neolithic). More ancient (Paleolithic) expansions altered the orientation of the PC1 gradient due to 1) a spatial homogenization of genetic diversity over time, and 2) the exact location of the Last Glacial Maximum (LGM) refugia. Overall we found that PC1 gradients consistently follow a SE-NW orientation if there is a large Paleolithic contribution to the current European gene pool, and if the main refuge area during the last ice age was in the Iberian Peninsula. Our study suggests that the observation of a SE-NW PC1 gradient is compatible with the view that range contractions have affected observed patterns of genetic diversity, and suggest that the genetic contribution of Neolithic populations to the current European gene pool may have been limited (Figure 2). Although this study was focused on humans, this framework could be applied to other species that might have experimented with range contractions as a consequence of environmental changes.

wards refuge areas can actually bring additional diversity to these areas, but only if the range contraction is rapid. When contractions or shifts are slow, we found that active migrations towards refuge areas could lead to a more pronounced loss of diversity than if migration was similar in all spatial directions (Arenas et al. 2012). These results suggest that species with different generation times and different migration abilities should be differently affected by environment change.

Inference of fragmentation levels from genetic data gathered at different scales over the species range

Populations living in a heterogeneous environment usually show a large variance in local population densities and migration rates, and generally present less local genetic diversity and higher levels of population differentiation than populations of similar size living in a constant and uniform environment. This is because genetic diversity is more rapidly lost in small demes than it is gained in large demes, leading to higher rates of local genetic drift.

Patterns of genetic diversity have been used to assess many properties of a population, but no attempt has been made to estimate the degree of environmental heterogeneity directly from patterns of diversity at different scales. It would therefore be useful to be able to infer the degree of environmental heterogeneity directly from genetic data, especially for sparse and cryptic species, or for species for which the exact definition of the population is difficult to assess.

We have simulated environmental heterogeneity using SPLATCHE2 where local deme carrying capacities (K) can vary in space according to a Gamma distribution with mean $\overline{K}$ and shape parameter α. The Gamma distribution is often used to describe various levels of heterogeneity of a given biological parameter (e.g. mutation rate, migration rate, population size, etc). The important thing to note here is that small values of α (typically $\alpha < 1$) are indicative of strong environmental heterogeneity, where a few demes have very high population densities and most others have very low densities (even being zero, which correspond to uninhabitable regions). Therefore, because habitat fragmentation usually creates uninhabitable regions, it is also associated to high levels of environmental heterogeneity. On the other hand, large values of α (typically $\alpha > 5$) imply little environmental heterogeneity, such that most demes have a very similar carrying capacity. Previous studies have shown that both local genetic diversity and levels of population differentiation would strongly depend on α, suggesting that patterns of genetic diversity at different scales could be used to infer α, and therefore, indirectly, the level of environmental heterogeneity.

We used an analytical method based on the Approximate Bayesian Computation approach (a statistical inference method allowing one to estimate parameters in complex models by computer simulation) to infer the shape parameter of a Gamma distribution directly from patterns of genetic diversity of several samples taken from a population having gone through a recent range expansion. Our results show that the degree of environmental heterogeneity (α) can be very well estimated if all other parameters of the model are known (Figure 3). When all other parameters need to be co-estimated, the estimation of α becomes difficult, and we can mainly distinguish small from large α values (Figure 4). In other words, we only have power to distinguish very heterogeneous environments from more homogeneous ones, but little prospect to get accurate estimations of α.

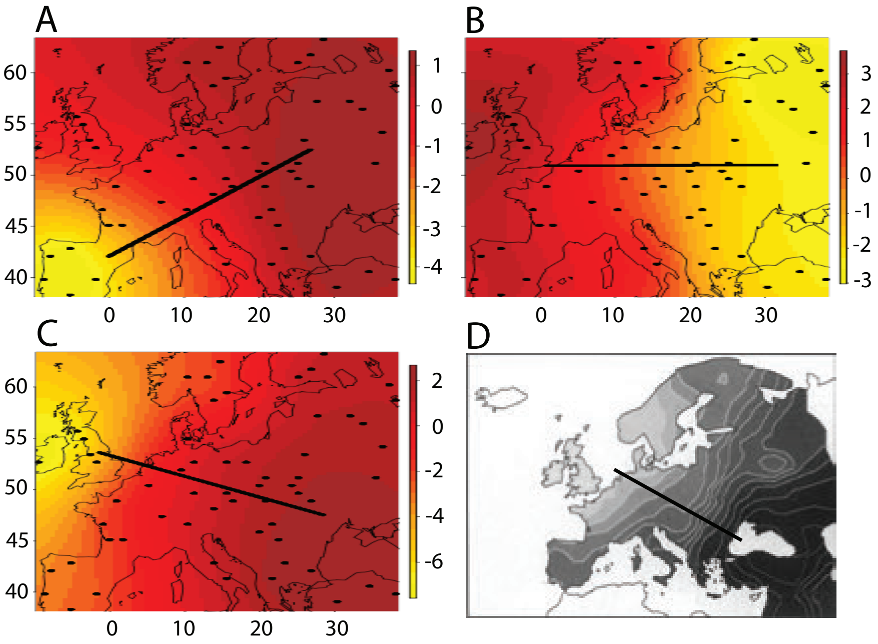


Figure 2. Influence of range contraction on Principal Component (PC) maps. We show the results of Principal Component Analysis (PCA) on Single Nucleotide Polymorphism (SNP) data in the case of a Neolithic range expansions from Middle East resulting in a final population that shows 80% with the pre-existing Paleolithic population: (A) Illustrative example of PCA derived from a range expansion. The PC1 gradient has a SW-NE orientation. (B) Illustrative example of PCA derived from range expansion followed by a range contraction towards all of southern Europe, and subsequent re-expansion. The PC1 gradient has an E-W orientation. (C) Illustrative example of PCA derived from range expansion followed by a range contraction towards the Iberian Peninsula only, and subsequent re-expansion from this refugium. The PC1 gradient has an NW-SE orientation. (D) Original PC1 map inferred from Piazza et al. (1995) [© 1995 National Academy of Sciences, USA] with a superimposed line connecting positive and negative PC1 centroids. The PC1 gradient shown in (C), which is the most similar to real data (D), was also found in scenarios with a larger Paleolithic contribution and either pure range expansions or range expansions with range contraction towards the Iberian Peninsula (see Arenas et al. 2013 for further details).

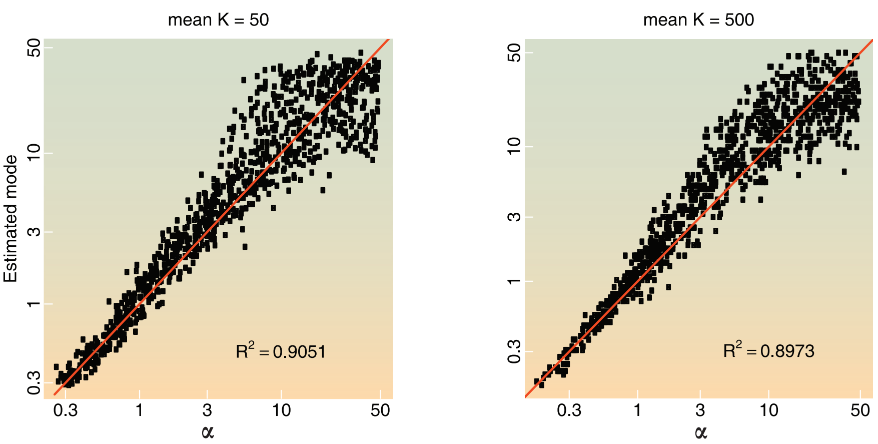


Figure 3. ABC estimation of our index of environmental heterogeneity (α) from genetic diversity simulated in species with small and large carrying capacity (K) when all other parameters of the model are known. The true value of α is shown on the x-axis and its estimation (as the mode of its posterior distribution resulting from an ABC analysis) is shown on the y-axis.

Concluding remarks

In this chapter we described the strong influences that habitat fragmentation and dispersal heterogeneity can have on genetic diversity, at different geographical and temporal scales. To this purpose, we mainly used the SPLATCHE framework to perform spatially explicit simulations of genetic diversity under complex demographic models, also allowing for temporal heterogeneity. We found that fragmented habitats often have a significant loss of genetic diversity relative to homogeneous habitats. This effect was reduced in species with long distance dispersal abilities. Similarly, range contractions led to a loss of genetic diversity, in particular when the contraction was slow. Note that the rate of environmental change needs to be considered relative to the generation time of the species involved, and the generation time of species needs to be taken into account when considering genetic diversity after climatic changes. Species with shorter generation times should suffer from more diversity loss after a range contraction than long-lived species (Arenas et al. 2012). We note however, that such species may also adapt more quickly to new environments. Fast range shifts, on the contrary, reduced genetic diversity more than slow range shifts where more individuals can track favorable environments. Indeed species with low migration rates and going through fast range shifts can easily become extinct (Arenas et al. 2012). In addition, we found signatures of range contractions on diversity by using PCA. In this case, a re-expansion after a range contraction introduces spatial genetic diversity gradients that depend on the location of refuge areas (Arenas et al. 2013). We also described a procedure to detect the level of habitat fragmentation from observed patterns of genetic diversity. Finally, we

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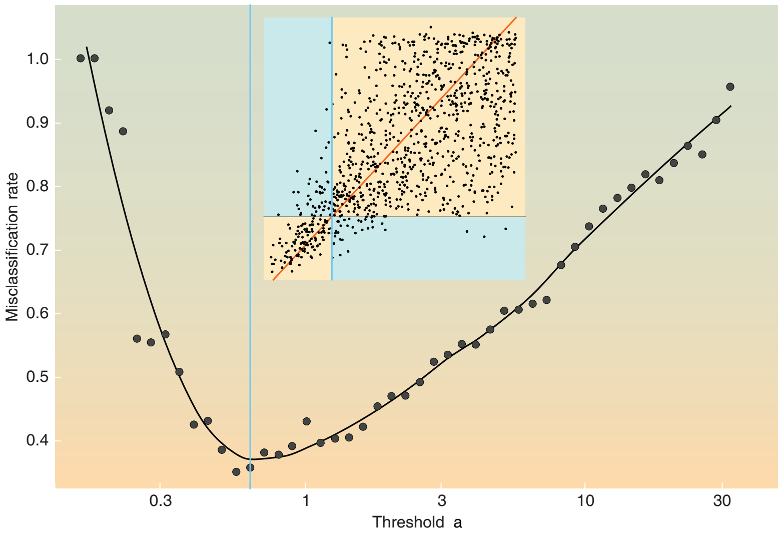

Figure 4. Optimal distinction between small and large α values when all parameters of the range expansion model need to be co-estimated with the environmental heterogeneity. The plot shows the estimated proportion of times where α was incorrectly estimated as below or above a threshold (a given true value). This incorrect assignment is minimized for α=0.63 (blue line), showing a maximal power to distinguish between values of α above and below this value. Here, the misclassification rate is inferred from an analysis of the plot of true (x-axis) vs. estimated (y-axis) α values shown in the central insert. Misclassification rate is obtained as the sum of the proportion of points in the blue regions relative to those in the orange regions on the left and right hand side of the blue line.

Box 2. Sex-biased dispersal

Population genetic structure is influenced by migration patterns. This includes sex-biased dispersal, likely impacting life-history evolution, population genetic structure and metapopulation functioning. In population genetics, sex-biased dispersal may not only reflect a difference in the number of dispersing individuals of one sex in relation to the opposite sex, but also the unequal reproductive success of dispersers. Fine-scale genetic structure and adaptation to local environments might therefore be promoted by sex-biased dispersal. Sex-biased dispersal can be identified and quantified by e.g. comparing the genetic differentiation of females to that of males. The sex with the highest dispersal frequency would have a lower genetic differentiation among different subpopulations (i.e. as measured by the genetic parameter F_{ST}). Similarly, sex-biased dispersal could be measured by comparing the level of genetic structure inferred from nuclear markers (inherited by both parents) to that indicated by mitochondrial DNA (as children inherit their mitochondria from their mothers) or Y chromosome (which male children inherit from their fathers). If the level of genetic differentiation inferred from mtDNA is higher than that inferred from nuclear markers, male-biased dispersal may be assumed. Simulations, undertaken with a different program inspired by SPLATCHE2 (Rasteiro et al. 2012), clearly show that different patterns of genetic differentiation can be detected under three scenarios, 1) bilocality (no sex-biased dispersal), 2) matrilocality (male-biased dispersal), and 3) patrilocality (female-biased dispersal, Figure 5). Y-chromosome genetic diversity is very low, especially in the patrilocality scenario for which only one Y-haplotype often remains after 1000 simulated generations. Note that the same effect was not seen in simulated mtDNA, probably due to differences in mutation rates and types of markers (Rasteiro et al. 2012). Indeed, the authors showed that the simple difference in mutation rates between the two types of sex-related genetic systems is sufficient to create an asymmetry that could be mistaken for differences in migration rates, even under bilocality scenarios.

Accounting for sex-biased migration in population and conservation genetics studies is of great importance as significant differences in sex-biased dispersal have been demonstrated among different taxonomic groups. Dispersal of mammals, reptiles and fishes were more frequently male-biased whereas dispersal in birds was more frequently female-biased (Figure 6). Therefore, knowledge on sex-biased dispersal may prove essential to develop and assess habitat management and landscape planning strategies for different species.

In many species, population decline has been linked directly to loss and fragmentation of habitats and indirectly to reduced inter-patch dispersal. Concerns about habitat fragmentation and landscape structure are usually based on the ability of wildlife to disperse between the blocks of habitat types that they require. Our simulations showed that patterns of sex-biased dispersal can have important consequences on some genetic markers and conversely they should inform us on the importance of sex-biased dispersal in natural systems that are difficult to study. Some studies have suggested that the different sexes may have a differing impact on demographic connectivity at different scales, the less dispersing sex more on local scales, while the more and farther dispersing sex on larger scales. Another consequence of sex-biased dispersal is that the rate of natural recolonization of locally extinct populations may be slower as it requires that both sexes disperse. Sex-biased dispersal may also act as a buffer against reduction of genetic variability due to high genetic drift in populations with small effective size (Schmeller and Merila 2007). Ultimately, explorations of the implication of unequal effective population size, migration rate and non-random individual dispersal will be necessary for synthesizing ecological and genetic theory on dispersal and population structure.

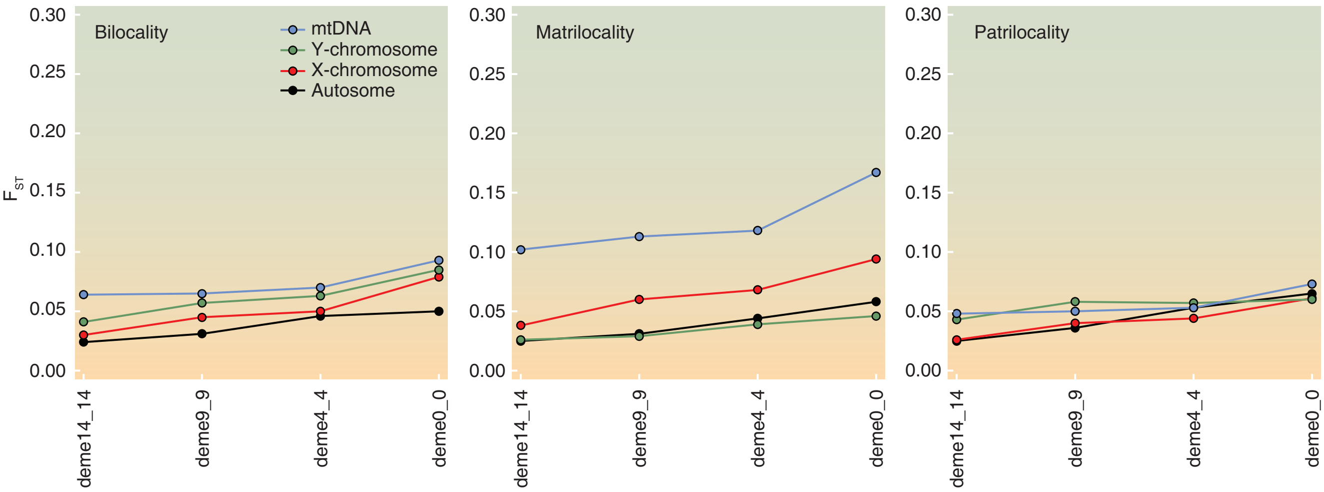

Figure 5. Genetic differentiation patterns under sex-biased migration patterns. Simulations were performed using a forward simulation program similar to SPLATCHE2. A square environment of 400 demes (20×20) was simulated under three scenarios, 1) bilocality (no sex-biased dispersal), 2) matrilocality (male-biased dispersal), and 3) patrilocality (female-biased dispersal). For each scenario we simulated independent autosomal loci, Y and X chromosome and mtDNA sequences. For each scenario and genetic marker type we computed a measure of genetic differentiation between demes at increasing distances. For simplicity only demes from the diagonal were used and compared to the same deme located in one of the corners (deme 19,19). As the panels show, sex-biased migration has a strong impact on the overall level of genetic differentiation, and on the differences between markers. The results also show that mtDNA and Y chromosome markers do not necessarily play symmetrical roles in the patrilocality and matrilocality scenarios because they differ also in mutations rates, as noticed by Rasteiro et al. (2012).

performed simulations incorporating sex-biased migration and found that such a bias could highly impact genetic data, which can therefore be used to infer sex-biased dispersal in species that are difficult to study in the field.

The fact that habitat fragmentation, dispersal patterns, and range movements strongly alter genetic diversity of species implies that they need to be considered for biodiversity conservation strategies.

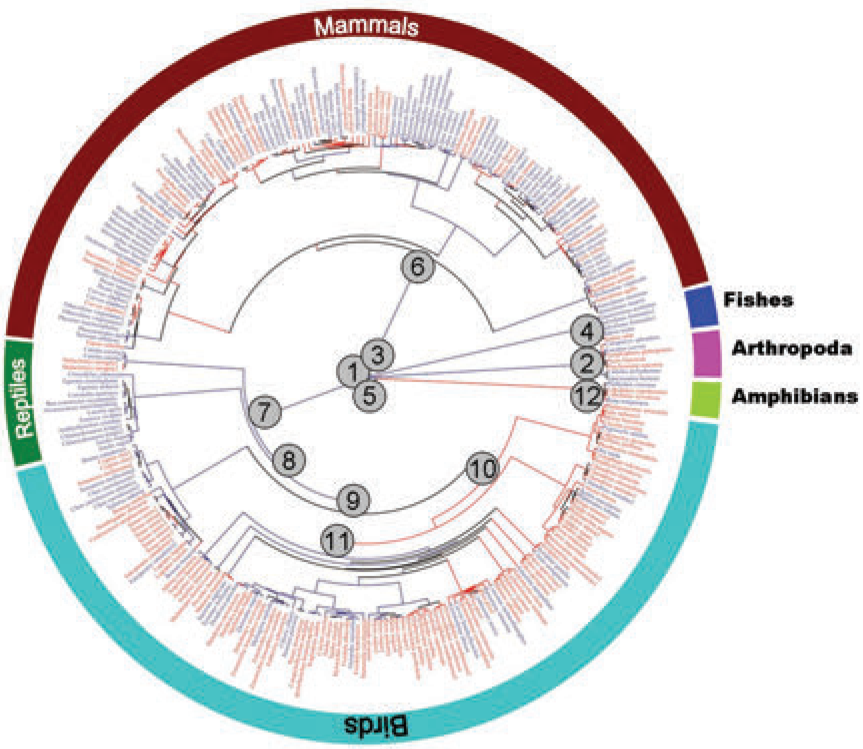

Figure 6. Phylogenetic tree of the ancestral character states reconstruction of sex-biased dispersal based on a parsimonious method on the 216 species (275 populations from publications) used. Branches and tips are coloured in blue for a male biased dispersal state and in red for a female biased dispersal state. In grey, branches for which the reconstruction method did not allow one to choose between a male or a female bias. Numbers on nodes correspond to: 1. Bilateria, 2. Arthropoda, 3. Osteichthyes, 4. Fishes, 5. Tetrapoda, 6. Mammals, 7. Amniota, 8. Sauria, 9. Neognathae, 10. Neonaves, 11. Birds, 12. Batrachia.

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Population viability: On the move from small to large scales and from single to multiple species

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Introduction

The European targets to halt the loss of biodiversity by 2010 have failed, and new targets for 2020 have been set. Yet to shift from observing this decline to altering the fate of species, it is imperative to understand the key drivers affecting species' population dynamics. One approach is to apply population viability analyses (PVA, Box 1) for a rigorous and systematic assessment of the potential fate of populations under alternative scenarios. The use of PVA has become so well established for directing conservation planning and management that international organizations, such as the International Union for Conservation of Nature (IUCN), recommend its application wherever desired and possible.

As conservation efforts move to larger scales and try to encompass a multitude of species, one may wish to obtain generalisations from the vast knowledge that has accumulated in the field. Examples of emerging questions are: How can PVA help guide conservation decisions? How can we use PVA for reserve design, e.g., by considering the area requirements of species? How can we best collate the outcomes of PVA studies toward advancing the conservation of other species? This chapter delineates some achievements and challenges in answering these questions.

How does population viability knowledge affect decisions? Sinai baton blue butterfly as a case study

An example on how PVA can help in establishing the conservation status of species and direct management plans is the case study of the Sinai baton blue butterfly (*Pseudophilotes sinaicus*). The world's smallest butterfly is only found in the St. Katherine Protectorate of Southern Sinai, Egypt (Figure 1). It is one of two species endemic to the site and is considered a flagship species and conservation priority for the area (James et al. 2003).

The international conservation status of a species, following IUCN categories, can be based on the global distribution of the species. In this case the overall Area of Occupancy is smaller than 2 km² and all butterflies occur in one location that could be affected by a single threatening event such as drought. Availability of monitoring data from the past decade indicates that the butterfly populations undergo dramatic fluctuations, associated with large fluctuations in the state of its sole host plant, Sinai thyme (*Thymus*

Box 1. Population viability analyses

Population viability analyses (PVA) encompass a broad range of models used by conservation scientists for various purposes, including advancing conservation theory, policy, and management. They are particularly important for assessing the risks of population extinction and identifying effective management options (Beissinger and McCullough 2002).

Typically, a PVA involves simulating the dynamics of a population (or metapopulation) over time. Repeating the process (to account for stochasticity) enables extracting population viability measures, e.g. the probability of extinction after a certain time (e.g., 100 years) or the average time to extinction (Pe'er et al. 2013).

The term PVA covers a broad range of modelling techniques varying in approach and type of data used. Models range from simple matrix models, to structured population models, to complex individual-based models simulating the behaviour and fate of each individual in a population. A range of available software (e.g., VORTEX, RAMAS, ALEX, Meta-X) now offers straightforward means of answering a set of typical questions in conservation biology.

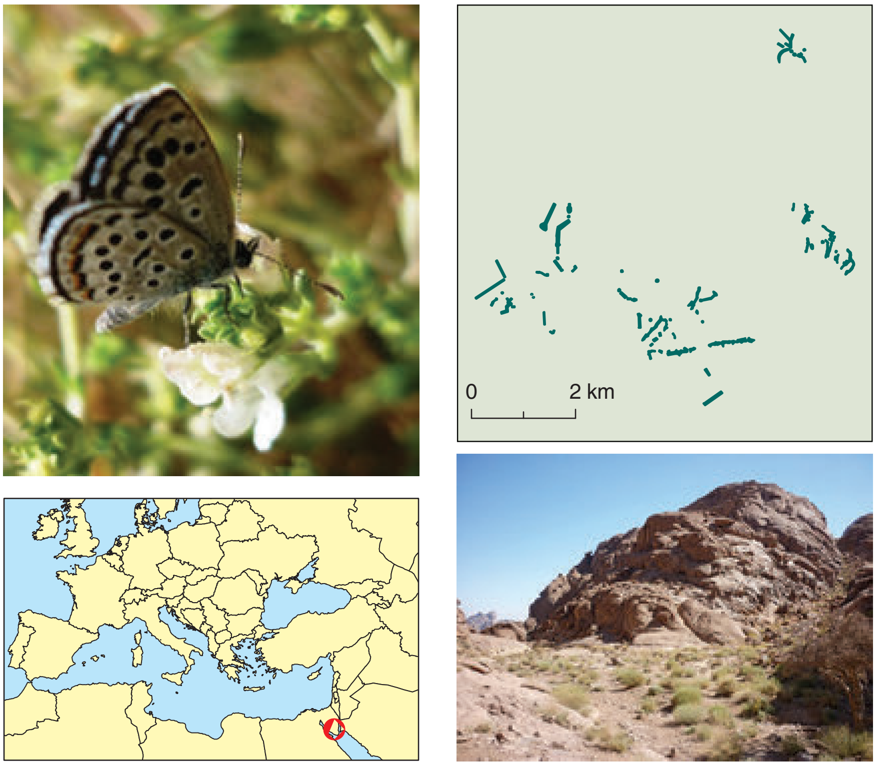

Figure 1. Sinai baton blue (*Pseudophilotes sinaicus*), its marked localities of occurrence, and the habitat where it occurs (photo: Katy Thompson).

decussatus). The combination of these three pieces of information qualifies listing the species as "Critically Endangered" according to IUCN criterion "B" (Geographic range).

Yet one can go a step further and use the data to parameterize a PVA. Feeding the monitoring data to the programme VORTEX (Lacy 1993), alongside environmental data (in this case, state of the host plant), resulted in the butterfly's extinction in > 20% of simulations within just 20 years. This, however, would enlist the species "merely" as Endangered under category "E" (Quantitative analysis available). This demonstrates that the availability of quantitative methods can actually lead to somewhat lower conservation concern, because the precautionary approach dictates taking a more conservative decision in the absence of good knowledge. Note, however, that the exact quantitative outcome of a model may strongly depend on software selection, model complexity, and input parameters (Box 2). One should therefore use PVA not to obtain exact numbers but to identify parameters to which model outputs are most sensitive, and hence, what factors should receive highest attention in species protection. In this case, mortality rates and stochasticity in carrying capacity were the most important parameters affecting viability, indicating that management plans should focus on improving the patches of host plant to reduce mortality and stabilize carrying capacity.

This example demonstrates the potential usefulness of a simple PVA, but of course, more complex approaches can be used as well. An important question to consider, therefore, is how much complexity is needed in order to address a conservation question (see Box 2). Once carefully designed and applied,

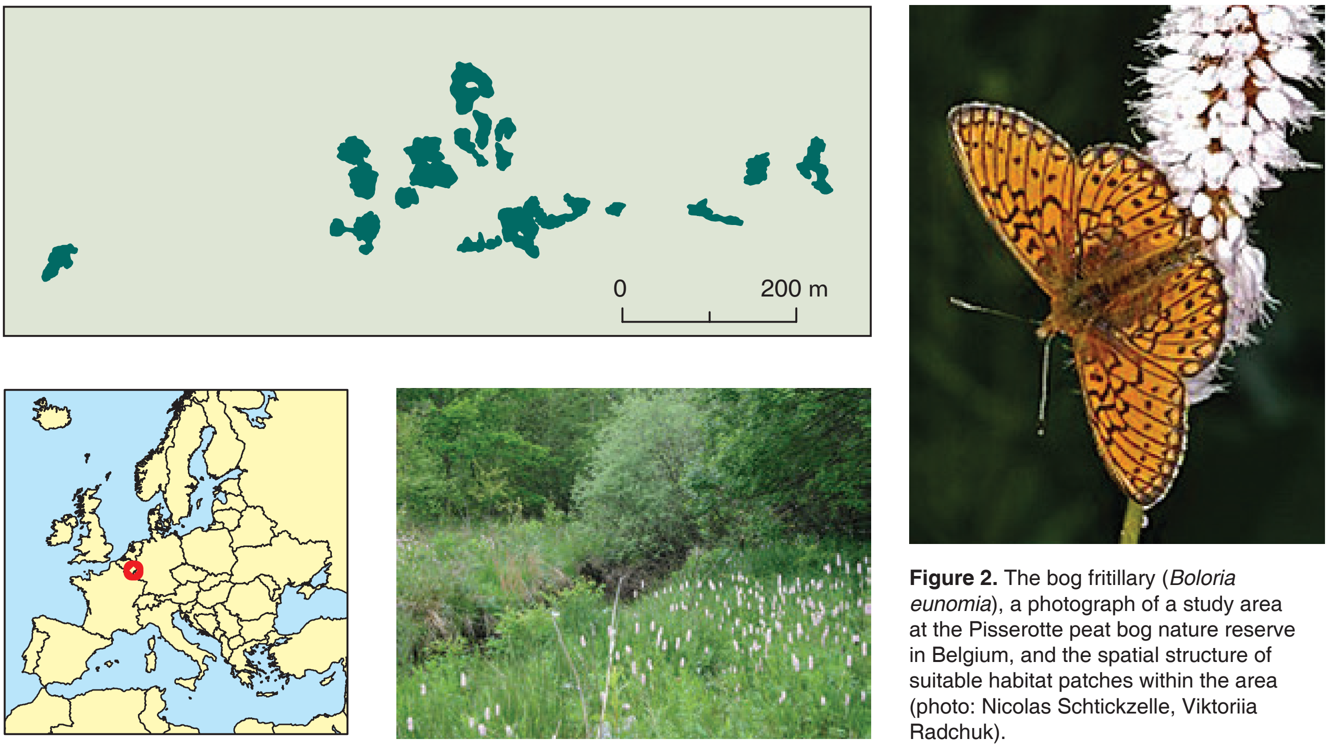

Figure 2. The bog fritillary (*Boloria eunomia*), a photograph of a study area at the Pisserotte peat bog nature reserve in Belgium, and the spatial structure of suitable habitat patches within the area (photo: Nicolas Schtickzelle, Viktoriia Radchuk).

Box 2. How simple or complex should a model be?

PVA model structure is usually defined by three important determinants: its goal, the life history of the species, and data availability. For maximum efficacy, it is important to know what level of model complexity is sufficient for delivering useful management guidelines. One of the ways to address this is by contrasting the performance of several models, differing in complexity, for the same species and landscape. For example, Radchuk et al. (2014) recently assessed whether yearly time steps (as commonly used in PVA on butterflies) are appropriate for describing the population dynamics of the Bog fritillary butterfly (*Boloria eunomia*) in the Pisserotte peat bog nature reserve in Southern Belgium (Figure 2). They compared a yearly stage-based model (ySBM) to a daily individual-based model (dIBM), with both models incorporating the same environmental descriptors. The two models were then compared in their ability to reproduce population data observed under current environmental conditions and their viability predictions under three different temperature change scenarios.

Both models matched with the observed field data in terms of the relationship between population density and population growth rate (Figure 3a). They also yielded the same ranking of temperature change scenarios in terms of their impact on the population viability, both indicating the highest forecasted temperature change as the most detrimental for population viability (Figure 3b). However, the models differed substantially in the absolute outcomes and rate of population decline, with the daily IBM generating much more pessimistic predictions. This difference stems from the IBM's ability to incorporate inter-individual heterogeneity, leading to phenological shifts induced by temperature change. This was not possible with the coarser ySBM, emphasizing that small-scale factors, such as daily variations and variability between individuals, can reflect onto much larger-scale ecological patterns, such as the sensitivity of populations and species to changes in climate.

The choice of ecological level (individual level or populations) and temporal grain (day or year in this case would not have affected the ranking order of alternative scenarios, but would substantially affect one's conclusion in terms of the urgency of action. Model choice should therefore be made with caution, taking into account the ecological specifics of the studied species, the scale at which climate change affects populations and species, and the exact purpose of the exercise – be it to qualitatively rank alternative scenarios or attempt to quantitatively assess future viability.

it is then important to ensure valid interpretation, e.g., by ranking alternative solutions (Franz et al. 2013).

How can we use PVA for reserve design? What are the area requirements of (multiple) species?

One of the potential outcomes of a PVA is an estimation of the Minimum Area Requirements (MAR) – namely, the area that is needed to support a viable population. The MAR concept explicitly addresses area, and is therefore highly relevant for conservation planning and policy. In a recent compilation of MAR estimates from the literature, Pe'er et al. (2013) found available estimates for 216 terrestrial animal species from 80 studies. These originated from two types of sources: either PVA studies, or empirical studies inspecting occupancy patterns in islands or isolated habitat patches and assessing the area under which occupancy probability falls under a certain threshold (Pe'er et al. 2013). Using estimates from PVA, they found that a large proportion of the variation in MAR between species can be explained by body mass (Figure 4a). Adding one or two life-history traits (among the following: feeding guild, generation length or offspring size) or environmental variables (average precipitation and temperature) further improved the predictive power of the statistical model. By contrast, estimates coming from empirical studies of occupancy patterns deviated from those that were based on PVA. They also showed no relation to body mass (Figure 4b), and generated MAR estimates that were best explained by more complex statistical models combining taxon, feeding guild, and additional life history traits. This probably reflects the sensitivity of occupancy patterns to transient dynamics (extinctions, colonisations) and especially to connectivity. These results, combined with a freely available database (http://scales.ckff.si/scaletool/index.php?menu=6&submenu=1), support the future use of PVA to derive estimates of species' area requirements based on a set of simple traits.

How can we best collate the single-species outcomes of PVA studies to advance the conservation of other species?

Given the large number of existing PVA studies, it is tempting to try to ask what other generalizations can be derived beyond minimum area requirements. For example, what factors are species sensitive to? Or, what would affect our estimates of species' needs? While some reviews offer partial answers, the specificity and complexity of PVA, alongside a lack of standard communication of

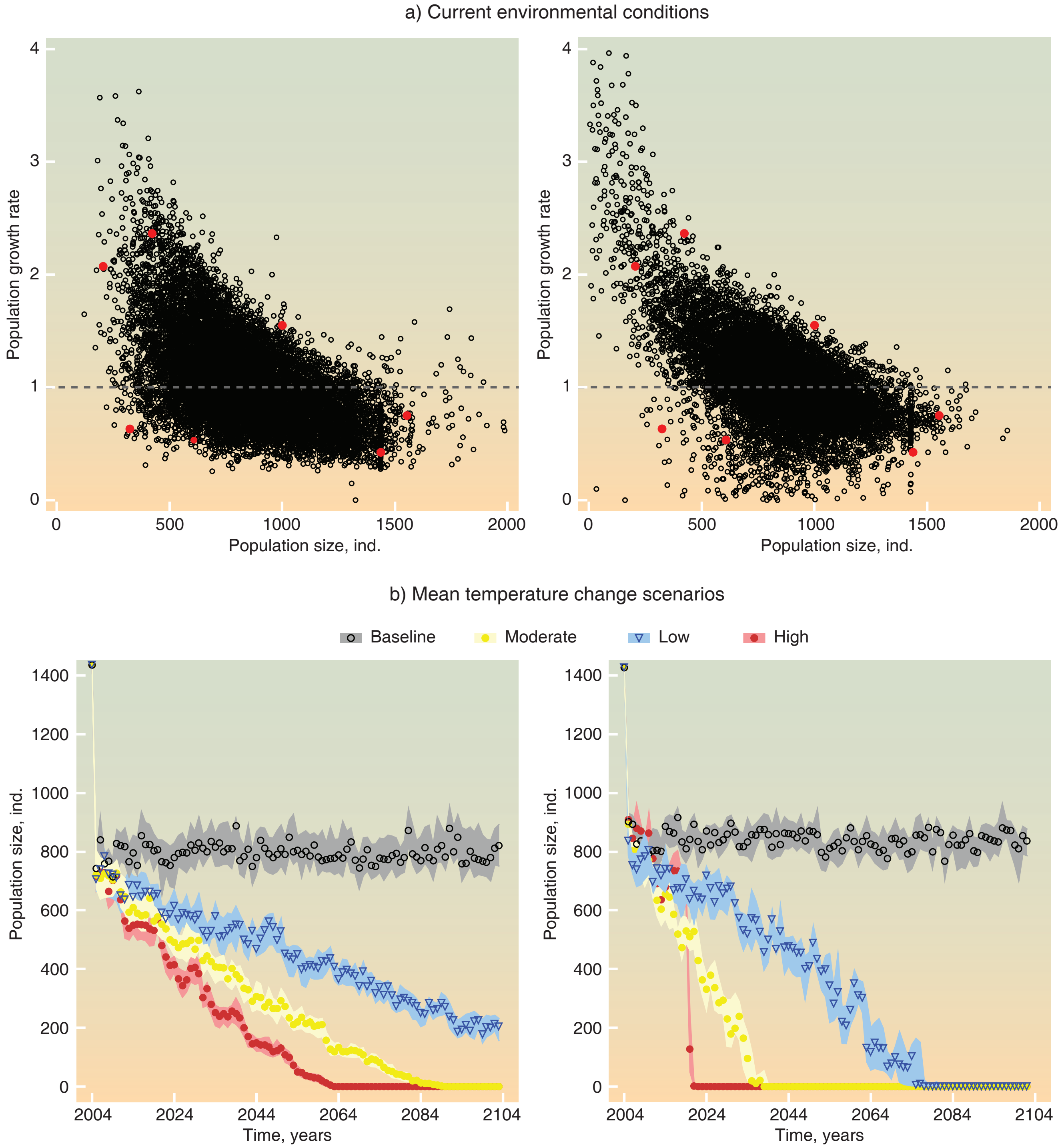

Figure 3. a) Comparison of a yearly stage-based model (left) with a daily, individual-based model (right) in terms of simulated population data (black) versus observed data (red). Both models match well the observed data under current environmental conditions as judged from the plot of population growth rate versus population size. The dashed grey line marks a growth rate of 1 (potentially stable population).
b) Comparison of the population dynamics resulting from the two models under three scenarios of change in mean temperatures – small, medium and large. For each year the median population size (and 95% CI) is shown based on 100 simulations.

results, impedes rigorous quantitative analyses across studies (Naujokaitis-Lewis et al. 2009, Pe'er et al. 2013). Guidelines do exist for applying PVA, and standard protocols are available for documenting and communicating ecological models, but an organized, standard set of guidelines was missing until recently for PVA. Pe'er et al. (2013) therefore suggested a common standard for the Design, Application and Communication of PVA: the DAC-PVA protocol (http://scales.ckff.si/scaletool/dac-pva.php). The protocol can enhance communication and repeatability of PVA, strengthen credibility and relevance for policy and management, and improve the capacity to generalise PVA findings across studies. Thereby, single-species studies can hopefully serve as pieces in building the greater puzzle of understanding the needs and sensitivities of species, communities and ecosystems.

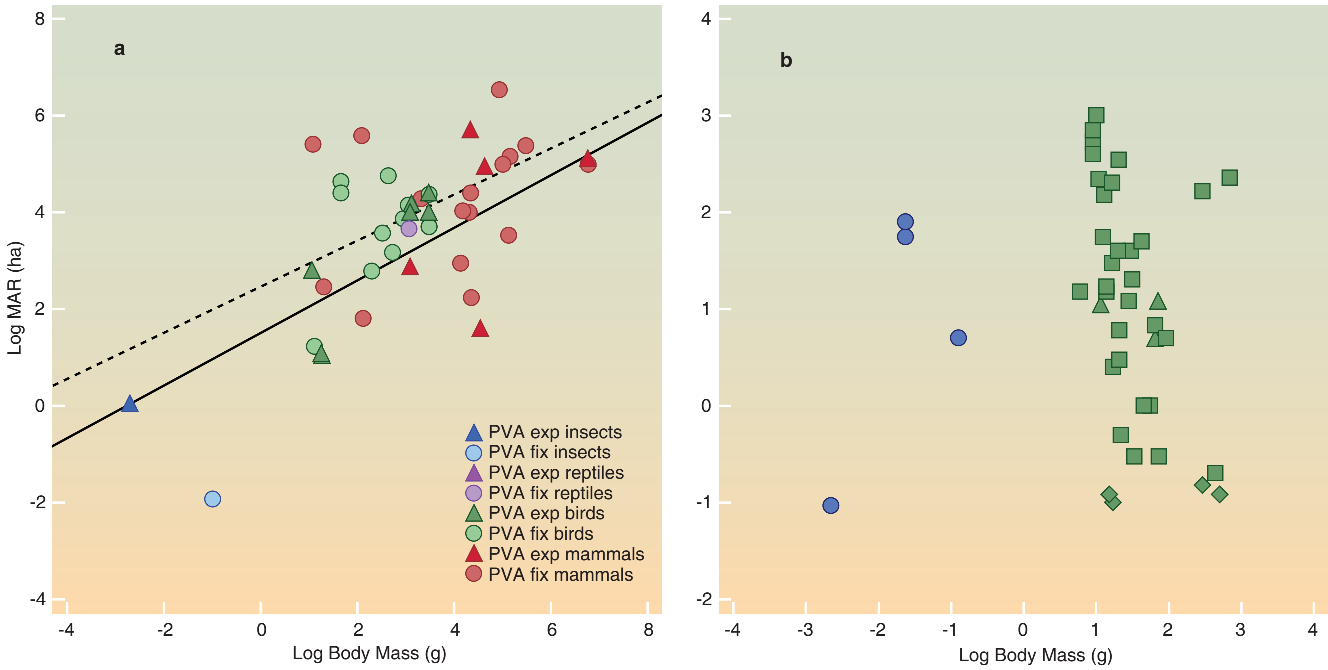

Figure 4. Relationship between the minimum area requirement and body mass based on a) PVA papers and b) empirical studies of occupancy across islands or isolated habitat patches differing in area. PVA studies in (a) are divided into those that explored a range of areas (triangles and darker colours) versus those where authors provided a fixed MAR value, or reported the minimum viable population size (MVP) alongside area-relevant information, such as density, without exploring area (circles and paler colours); Red = mammals; green = birds; blue = insects; purple = reptiles; solid line = PVA exploring area, dashed line = PVA studies reporting a fixed MAR; Shapes in (b) refer to the threshold occupancy value set by authors: diamonds=30%, squares=50%, triangles=80%, circles=90%; blue = insects, green = birds. Regression lines are provided only for significant relationships.

Conclusions and recommendations

The vast amount of knowledge that has been gathered by theoretical and empirical studies and ecological models offers excellent opportunities to support biodiversity conservation. Yet the focus on particular species and spatial scales requires further joint efforts for scaling up: from single species to communities and from focal landscapes to the entire species' distribution. This is particularly important given that the combined pressures by climate- and land-use changes operate across large scales on multiple species.

To support such up-scaling efforts, we recommend that researchers:

- adhere to standards in communication
- select the simplest model that encompasses the most important biological processes for the question at hand, considering data availability and quality, as well as the uncertainties and limitations associated with the approach (as demonstrated in Box 2)
- apply multiple models where possible and ensure ranking of alternative scenarios as an output; and
- attempt to translate the outcomes of their analyses into minimum area requirement estimates, because unlike many other viability measures, area units are intuitive and useful for decision-makers and planners to work with.

For policy-makers, we recommend:

- supporting the design and application of Population Viability Analyses as means to obtain robust answers to (clearly defined) questions; and
- using the outcomes of such estimations not as quantitative 'predictions' but as means to rank alternative scenarios.

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Scaling communities and biodiversity

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Introduction

One of the main targets of nature conservation is to protect biological diversity, or biodiversity. But what is biological diversity? In a simplest case, it is just a number of species living at given place, e.g. a habitat patch. Naturally, if this number is comparatively high, the place in question may deserve protection. However, there are two complications. First, the number of species depends on the area of the given habitat patch, so that it is not very suprising if we count many species on large habitat patches and lower number on smaller patches. We would need to assess whether the number of species is higher than would be typical for an area of that size. It does not make sense to quantify biological diversity without accounting for the area of the study plot.

Second, there are some landscapes that are not characterized by particularly high local species richness (so-called *alpha diversity*), but where the overall biological diversity across the landscape is high because very diverse sets of habitat patches are found within it, each of which hosts a uniquely different set of species. In such a case, the landscape has high *beta-diversity* (species turnover among individual sites or habitats) and consequently also high *gamma-diversity*, or regional diversity. Under such circumstances, protection of any particular habitat patch is of little value, and instead the whole landscape mosaic should be protected. The conclusion is clear: biological diversity is scale-dependent, and it is necessary to consider this scale-dependency whenever dealing with the number of species (Storch et al. 2007).

The classic way of examining this scale-dependency in species richness is based on a species-area relationship (or SAR): plotting the number of species found as a function of the area of a sample (e.g. a habitat patch, landscape or region). The fact that the number of species generally increases with area is actually tightly related to beta-diversity, i.e., to uniqueness of different places in terms in their species composition. Imagine a homogeneous landscape characterized by low beta-diversity, i.e., by very similar species composition on individual sites. In such a landscape, any increase of sampled area is followed by only a negligible increase of total number of species. In contrast, in landscapes characterized by high beta-diversity, where every spot is different in its species composition, species richness increases quickly with increasing area. Indeed, many indices of beta-diversity (Box 1) are mathematically related to the slope of the species-area relationship (Šizling et al. 2011). Understanding the SAR is thus crucial for understanding all patterns of biological diversity.

The species-area relationship (SAR)

The fact that the number of species increases with area is obvious, but the exact form of this increase deserves attention. It is only rarely linear, and if plotted in non-transformed, arithmetic axes, the rate of increase of species richness with area gradually slows down (Figure 1A). However, if we plot both axes on logarithmic scales, the relationship often becomes almost linear (Rosenzweig 1995) (Figure 1B), at least over a range of scales. A linear relationship in the log-log scale can be expressed as a power-law, so that the slope of the line becomes the exponent of the relationship, Z.

This form of the relationship has a useful property, i.e., that it is scale-invariant. This means that an increase of area by a given multiple leads to the increase of species number by a constant (different) multiple, regardless of the absolute values. This would also mean that if we know the slope Z and species richness for a particular area (scale) we should be able to predict species richness at both smaller and larger spatial scales. This has been used for the prediction of diversity loss due to area loss (e.g., May et al. 1995). For instance, it has been claimed as a rule of thumb (following Darlington 1957) that a 90% area loss should lead to the eventual extinction of about half of all species in that area, regardless of the size of the initial area. Such a claim is based on an assumption that the SAR is a power-law with the slope Z equal to 0.3 (a value sometimes observed on islands, see Rosenzweig 1995).

However, the situation is not that simple. The relationship between species richness and area is often not exactly linear in logarithmic space, and consequently several other forms of the SAR have been proposed (Tjørve 2003). Indeed, recent research suggets that if we plot the SAR across a sufficiently wide range of spatial scales (e.g., from square metres to continental scales), it tends to appear triphasic in logarithmic space: with species richness initially increasing steeply at fine scales, but with a decreasing slope (Harte et al. 2009), then becoming approximately linear (that is to say: power-law) over intermediate scales, and finally accelerating upwards to produce steep slopes at very coarse (continental) scales (Figure 2) (Storch et al. 2012). Different mathematical expressions of the SAR proposed in the literature may be actually related to the fact that different researchers have studied different parts of the whole triphasic curve. The non-linearity of the SAR implies that simple scale-invariant estimates of species richness changes with changing area are unlikely to give accurate predictions.

The curvilinearity of the SAR at fine scales is caused by the fact that species number is constrained by the limited number of individuals in a small sample area; to put it simply, you cannot sample more species than you have individuals, and you are unlikely to sample many individuals at scales that are close to the sizes of individual home ranges (Figure 3) (Šizling et al. 2011). At extremely fine scales, the slope of the SAR should approach 1 (as the first individual sampled will necessarily also sample 1 species). This has one nontrivial consequence: the local slope of the species-area relationship (its derivative) is related to the ratio between total number of individuals and number of species (Harte et al. 2009). If the number of species is relatively high in relation to total number of individuals, and mean population sizes of the species are thus small, the local slope of the SAR will be high, and vice versa. Moreover, since the increase of species richness with area is related to beta-diversity, beta-diversity is expected to be high whenever there is only a limited number of individuals per species, i.e., with low mean population sizes (species are relatively rare). In contrast, when the average population size is high (species are generally abundant), beta-diversity is expected to be relatively low at these scales.

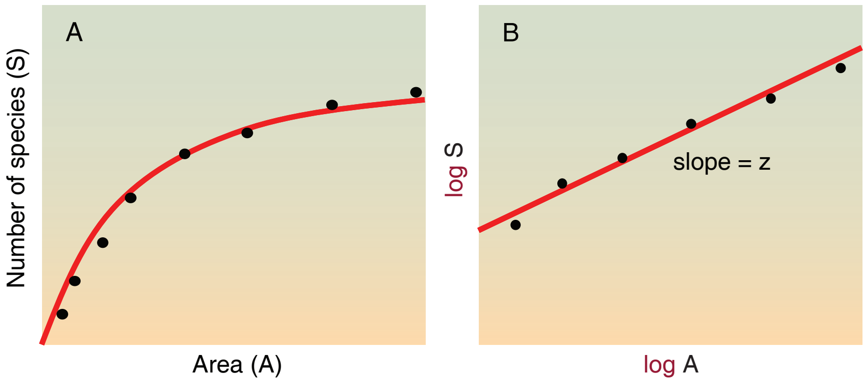


Figure 1. The typical form of the species-area relationship in arithmetic (A) and logarithmic (B) axes. The increase of the number of species is progressively decelerating in arithmetic space, but close to linear in logarithmic space, although particular measurements deviate from perfect linearity, and thus the straight line must always be taken as an approximation. Since the line can be expressed by the equation y = ax + b where *a* is the slope of the line and *b* is the intercept, in this case it can be written as log(S) = Zlog(A) + log(c), where *S* is the number of species, *A* is area, *Z* is the slope of the line, and *c* is a constant related to mean number of species per unit area. This equation in non-logarithmic form is expressed as $S = cA^Z$, i.e. the slope of the line in the logarithmically plotted SAR becomes the exponent of the power-law.

Box 1. Beta-diversity measures and related problems

Differences in species composition of distinct spatial units may be – and have been – measured in many ways, which, however, are often not equivalent and measure different aspects of the pattern. The simplest index of beta-diversity is called Whittaker index (R.H. Whittaker introduced it in 1960 as the very first index of beta diversity), which is just the total (gamma) diversity of all samples together, divided by mean local (alpha) diversity. Clearly, if the individual local plots strongly differ in their species composition, the total species richness of all plots together (gamma diversity) must be much higher than mean diversity of one plot, resulting in high value of this index. Since the slope of the species-area relationship is also given by the ratio of species richness of a larger area (gamma) to richness of the smaller area (alpha), Whittaker beta-diversity index is mathematically linked to the slope *Z* of the SAR. However, this mathematical connection holds only for adjacent plots (in which the gamma diversity represents the species richness of the large area comprising all the adjacent smaller plots together). Beta-diversity may be calculated also for distant, non-adjacent plots, but the relationship to the species-area relationship is in this case more complex and not straightforward. Moreover, it is also affected by the *distance-decay of similarity*, i.e., on how the beta-diversity depends on the distance between plots.

There are indices of beta-diversity that are mathematically related to the Whittaker index; for instance the Jaccard index of similarity which simply calculates the ratio between the number of species shared by the plots and the total number of species. However, many other published indices are not directly linked to these indices or to each other, and thus they actually measure different things. For this reason, the literature concerning beta-diversity is quite messy. For some authors, beta-diversity is a synonym for spatial species turnover, whereas others try to distinguish these as two separate matters. There is no consensus about terminology and the exact purpose of different indices. The practical way to deal with this conceptual complexity is to use a selected index consistently throughout a given study, to report which index was used and why, and to be cautious when comparing results based on different beta-diversity indices.

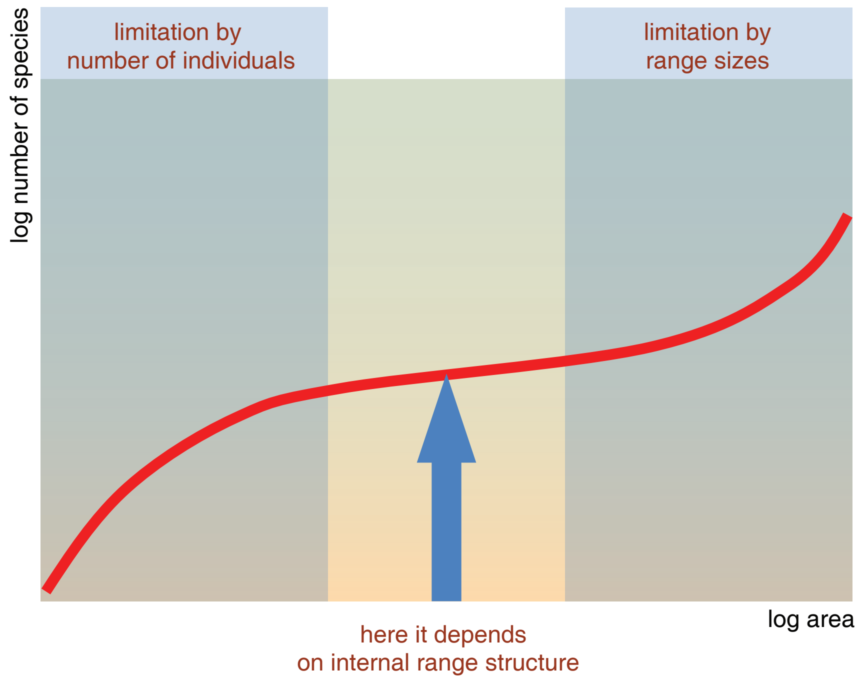


Figure 2. The species-area relationship across a wide range of spatial scales, with the finest scales corresponding to the size or home range of an individual, and the coarsest scales representing the size of whole continents. The curvilinearity at the finest scales is caused by the limited number of individuals sampled (see text and Figure 3), while the opposite curvilinearity at coarse spatial scales is due to limited range sizes (see text and Case Study 1).

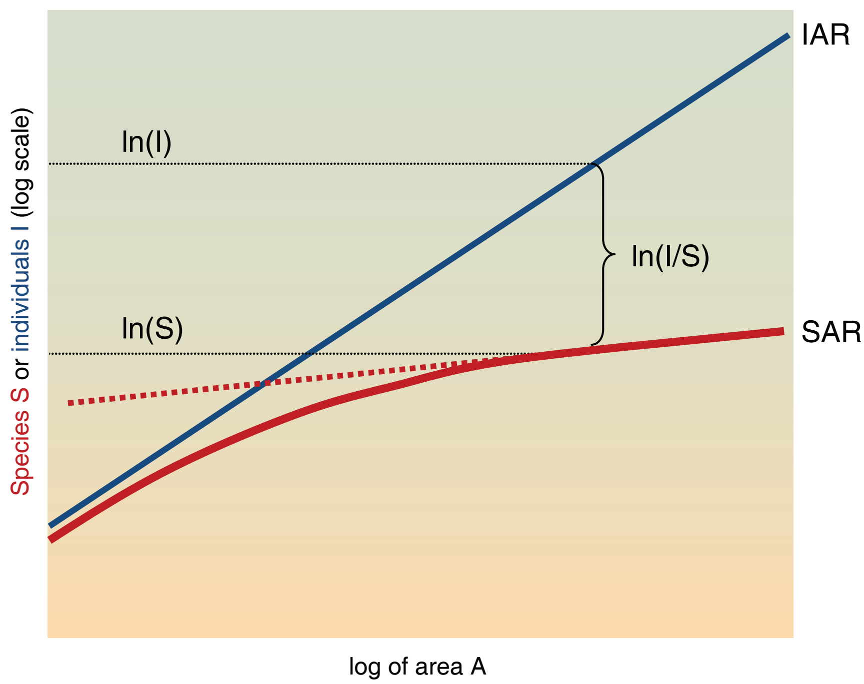


Figure 3. Species-area (SAR) and individuals-area (IAR) relationships at fine spatial scales. The average number of individuals in an area (summed across all taxa) has to increase linearly with area, i.e., in logarithmic space it is a line with a slope equal to 1. The number of species must be always lower than the number of individuals, and the increase in the number of species with area is less rapid than that of individuals, i.e. its slope must be less than one (unless every species is represented by just one individual; in such case both the curves coincide). This implies, however, that the species-area relationship cannot be linear across all spatial scales, since it cannot cross the individuals-area relationship (dashed line). This explains why the SAR is steeper when the number of species is closer to the number of individuals, i.e. when mean number of individuals per species is low. Mean population size is reflected by the distance between the two curves (since $\ln I - \ln S = \ln(I/S)$), and thus related to the slope of the SAR, and consequently also to various measures of beta-diversity (Šizling et al. 2011).

The increase in the local slope of the SAR at very coarse spatial scales is related to the fact that individual species‘ geographic ranges are limited, and they may be small in comparison to sample areas when we measure diversity at continental scales. If we increase the sampling window to be larger than the size of ecoregions or biomes, most species‘ ranges are included within the sampling areas, and further increases in area bring in completely new sets of species restricted to other areas, elevating the slope of the SAR. The precise spatial scale at which this upward bend in the SAR begins depend on mean geographic range of the taxon in question; e.g., if amphibians have on average smaller ranges than mammals, then this upward-increase will begin at finer scales for them and is more rapid. Indeed, if we rescale the values of area using mean range size for a given taxon and region, all the SARs collapse into one universal relationship at these large scales (Storch et al. 2012, Case Study 1).

The curvilinearity of the SAR at both very coarse and very fine spatial scales is therefore understandable from purely geometrical reasoning. The rapid increase of the number of species with area at fine scales is due to the limited number of individuals, while the increase at coarse scales is due to the geographic limits of species‘ ranges. However, most planning decisions relevant to conservation deal with intermediate scales: much larger than an individual‘s home range but much smaller than most species‘ geographic ranges. For such purposes, we can focus on the more linear (approximately power-law) middle section of the SAR, which is closely related to beta-diversity at these scales, as mentioned above. This then brings us to a vital issue: What are the factors responsible for determining beta diversity patterns?

Drivers of beta-diversity patterns and the SAR slope

Beta-diversity, and thus also the rate of increase in the number of species with area, is determined by the fact that species do not occur everywhere, and individuals of a given species are typically aggregated into certain parts of the landscape. This may happen because of two sorts of reasons (Keil et al. 2012). Firstly, different habitats or climatic conditions are found in different places, and consequently any species that requires specific conditions will be restricted in where it can live. Secondly, species have only finite powers of dispersal, so that even if good habitat is available for them in a distant area, they may not be able to colonise it due to dispersal barriers and/or limited time to spread from the centres of origin. High beta-diversity (and a steep slope of the SAR) is thus expected whenever there is pronounced habitat heterogeneity or important dispersal barriers.

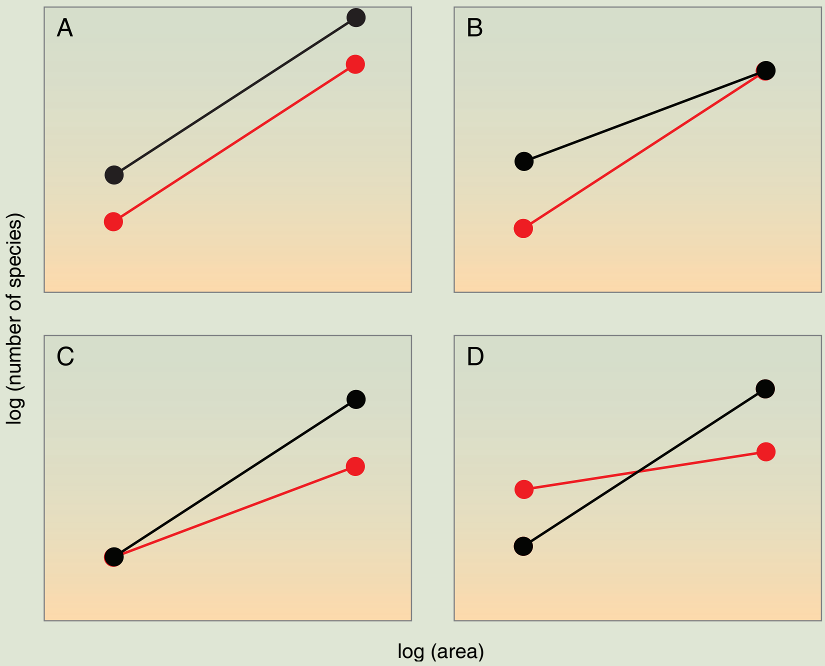
The effect of these contrasting factors may also be scale-dependent. Analysis of patterns of beta-diversity in European plants and animals indicated that the effect of dispersal limitation prevails at coarse spatial scales when we compare species composition of large areas across large distances. Climatic differences are important for determining beta diversity at somewhat finer scales, but these still involve larger areas than land-use differences, which drive fine-scale patterns of beta-diversity. Additionally, beta-diversity is lower between large than between small areas, which is in accord with the observation above that the SAR is steeper at small spatial scales (Keil et al. 2012, Case Study 2).

Beta-diversity of European plants and animals is particularly high in southern Europe (Figure CS2.3). This may be related to quite complex topography (mountains, peninsulas) creating dispersal barriers or notable variation in environmental conditions. However, this pattern may also represent a historical legacy of ice ages, when most of European fauna and flora persisted in small refuges in southern Europe, and many species are still confined to these areas. In any case, the proximate driver of high beta-diversity lies in the fact that many species in southern Europe have small geographic ranges. As we have discussed above, patterns of species distribution are tightly related to patterns in species diversity revealed in the SAR and beta-diversity. *The more restricted distribution of species, the higher is beta-diversity, and the steeper is the increase of the number of species with area.*

This has non-trivial consequences for diversity changes due to human

Box 2. Scale-dependence of biodiversity changes

The way temporal changes of the number of species vary with spatial scale can be illustrated with the changes of the species-area relationship (SAR; here referring just to two spatial scales, local and regional) before (black line) and after (red line) a disturbance. When the disturbance leads to extinction of some species without any bias towards more common or rare species, we should expect a parallel decrease of species richness in both scales, and thus no change in beta-diversity (A). However, if the disturbance just decreases the sizes of species ranges without leading to regional extinction of any species, only local species richness decreases, with consequently steeper SAR and thus higher beta-diversity (B). Alternatively, regional extinction of species may be compensated by the spreading of remaining species, so that local species richness remains the same (C). In such a case, the SAR slope and beta-diversity decreases. This pattern can be strengthtened by further spread of common species, increasing local diversity above the levels before the disturbance and further depressing beta-diversity (D). Such a situation is predicted to follow from biological invasion. Situations (C) and (D) refer to the process of biotic homogenization.



impact (Keil et al. 2011, Case Study 3). If human disturbance leads to the shrinkage of species' ranges, the average alpha-diversity (the number of species found on individual localities) should decrease, but beta-diversity is likely to increase. In contrast, the extinction of rare species leads to decreases in both alpha- and beta-diversity (and also regional diversity, i.e. gamma diversity). The spreading of invasive species, on the other hand, leads initially to increases in local (alpha) diversity (at least until the invasion begins to have adverse effects on native fauna and flora), and also beta-diversity, but when the invasive species becomes widespread, beta diversity characterized by the uniqueness of individual sites decreases (Box 2). This phenomenon is called biotic homogenization and is considered to be major process in the contemporary biosphere.

Case Study 1: Biodiversity scaling across continents

Traditionally, species-area relationship (SAR) has been approximated by a simple power function (i.e., a straight line if both the area axis and species richness axis are expressed in a logarithmic scale). It has been used for extrapolations of diversity across spatial scales or to estimate numbers of species that will go extinct after a given area is destroyed. It used to be assumed that if we know, e.g., the size of an area of a tropical rainforest which has been destroyed, we can use the species-area relationship to estimate the proportion of species which went extinct there. In reality, the relationship is more complex than a simple power law and it also differs across taxa and regions, which cast doubts on its usefulness.

A wide range of functions have been fit to species-area relationships, which might lead to an impression that different taxa and regions follow different scaling rules. However, the situation is not as complicated. Storch et al. (2012) have examined the distribution of all species of amphibians, birds, and mammals across all continental landmasses. Surprisingly, the species-area relationship at these large scales follows simple and yet non-trivial rules. Instead of being linear in the logarithmic scale (that is power-law), it is upward-accelerating for all taxa and continents (Figure CS1.1). Moreover, its curvature depends on mean species geographic range, so that taxa with smaller ranges – for example amphibians – reveal more prominent curvature and consequently higher slope of the relationship at large areas. When we express the area in units corresponding to mean species range of a given taxon within a given continent, all of the curves collapse onto one universal relationship (Figure CS1.2). The number of species for given area can thus be estimated using the knowledge of mean species richness for some given area, with only one additional piece of information – mean range size of given taxon within the region.

Additionally, an interesting pattern emerges when we look at the relationship between area and the number of species which are restricted exclusively to this area, i.e. which are endemic to it (the so called endemics-area relationship). These species are particularly relevant for extinction estimates as they will become globally extinct if the area they occupy is destroyed. At continental scales, it is the endemic-area relationship that follows a simple power law, so that the number of endemic (and thus potentially extinct) species is roughly proportional to the potentially destroyed area, indicating high risk of extinction from area loss (Storch et al. 2012).

Case Study 2: Beta-diversity patterns in European plants and animals

Keil et al. (2012) examined the scaling properties of beta diversity on the basis of high-quality distributional data for birds, butterflies, vascular plants, amphibians, and reptiles that were all arranged into a 50 × 50 km UTM grid across Europe (Figure CS2.1). For the investigation of smaller-scale patterns, national distributional atlases of butterflies (Finland), birds (Czech Republic and Catalonia), and vascular plants (United Kingdom) were used. Within each of these datasets, a series of 2-3 nested grids with the same spatial extent but with varying grain (resolution) was generated. Each cell within these grids was characterized by land cover and climatic conditions. Keil et al. (2012) then analyzed relationships between beta diversity, geographic distances, and environmental dissimilarities, and identified areas of rapid species turnover by mapping and analyzing patterns of beta diversity only in a set of adjacent grid cells (first distance class).

For birds, butterflies, vascular plants, amphibians, and reptiles beta diversity is higher and more variable at small spatial resolution. Hence, conservation efforts should be focused on preserving beta diversity at these smaller grains. In other words, the priority should be to preserve local uniqueness.

On the scale of Europe, dispersal limitation plays a major role in generating species turnover (Figure CS2.2). Hence, any efforts to conserve beta diversity (or local uniqueness) must carefully take into account not only the presence of natural migration corridors, but also natural migration barriers that can preserve beta diversity. It also means that European-wide modelling of shifts in species distributions must explicitly consider dispersal limitations.

Climatic and land-cover (habitat) differences have additional influence on beta diversity, and the relative importance of these variables differs at different spatial resolutions (Figure CS2.2). This shows that species turnover is a phenomenon driven by a complex interplay between dispersal limitations and climatic and habitat requirements of species. Correspondingly, conservation efforts on the continental scale (Europe) must consider all of the three phenomena by addressing climate change, connectivity (dispersal limitations), and land-use (management) on equal levels of importance.

On the scale of individual countries, the most important factor influencing beta diversity is climate, which means that climatic envelope modelling of species distributions may be relevant and

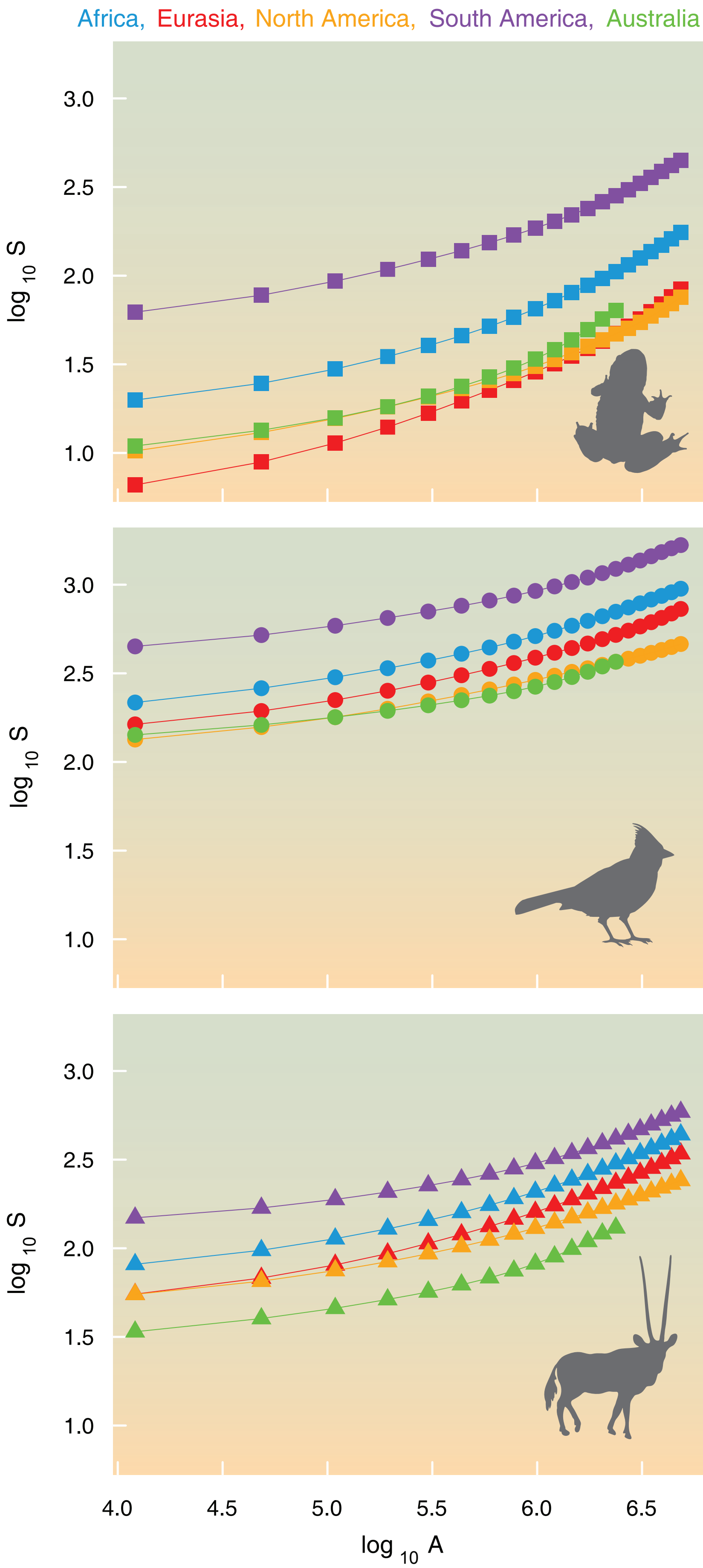


Figure CS1.1. SARs across five continents and three vertebrate classes. The SARs for amphibians, birds and mammals reveal an upward-accelerating shape for logarithmic axes.

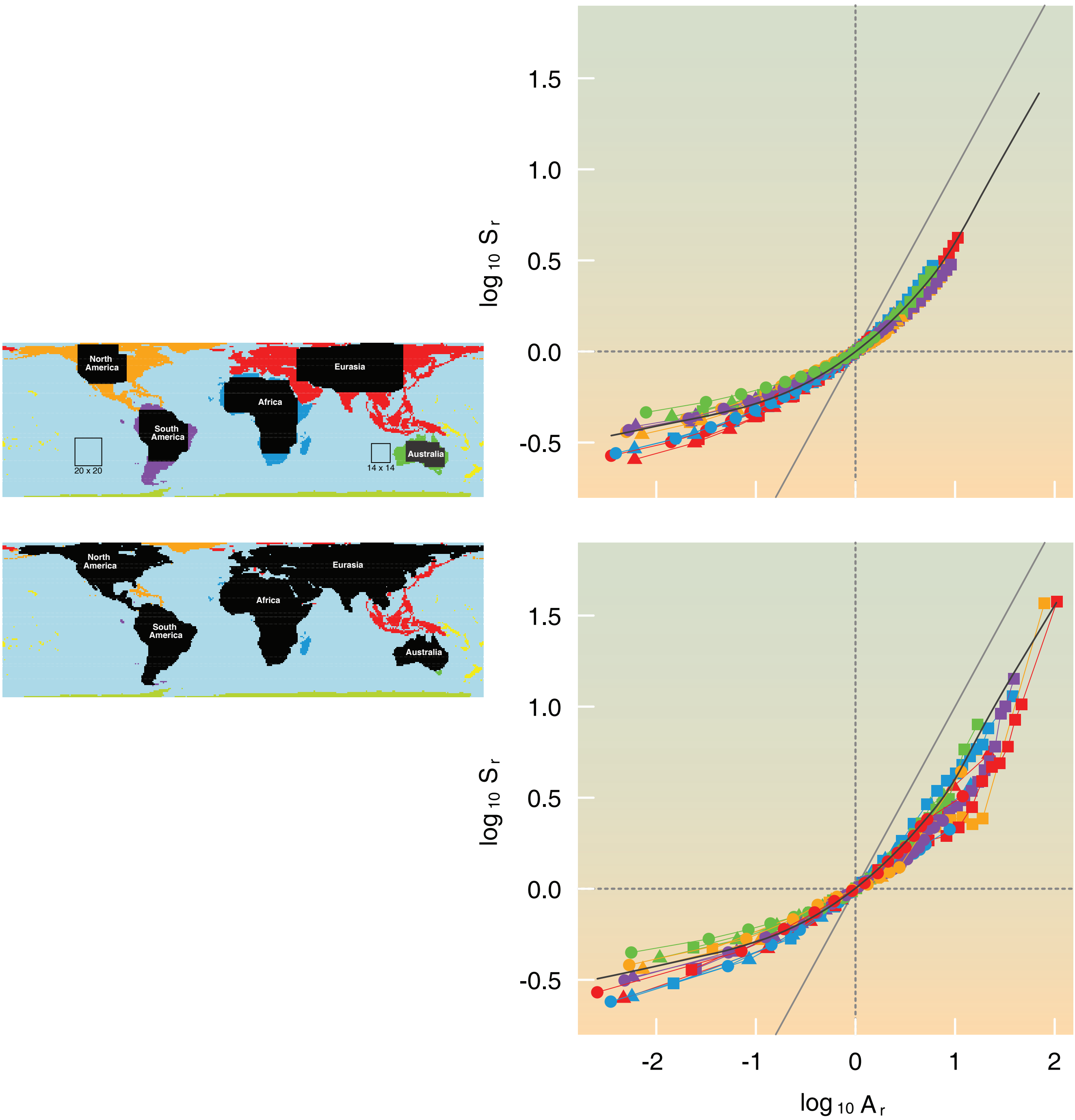


Figure CS1.2. SARs after rescaling for the sampling design based on square sample plots (above) and based on an alternative, continental shape design, in which sample areas are not quadrats but keep the shape of the given continent (below). After expressing the area in units corresponding to mean range size and standardizing the vertical axis so that it represents species richness relative to mean richness for a given unit area, all the SARs collapse into one universal relationship, although some deviations exist, particularly in small areas. Solid black lines refer to rescaled SARs predicted by simulations based on a random placement of simplified ranges. Solid grey lines all have slope of 1. The horizontal axis has been rescaled so that $A_r = A/\bar{R}_{t,c}$ where A_r is the rescaled area, A is the area of the study plot and $\bar{R}_{t,c}$ is the mean range size for taxon t and continent c. Vertical axis represents species richness proportional to the richness of an area equal to $\bar{R}_{t,c}$, i.e. $S_r = S_A/S_{\overline{R(t,c)}}$, where S_r is the rescaled number of species, S_A is mean number of species for a given area, and $S_{\overline{R(t,c)}}$ and $E_{\overline{R(t,c)}}$ are mean richness values for the area that equals the mean geographic range size of a given taxon and continent.

even useful within these smaller scales. It also shows that the expected climatic changes will most severely influence patterns of species turnover at the scale within individual European countries.

Interestingly, both *species rich* areas of southern Europe (Mediterranean peninsulas) and *species poor* areas of northernmost Europe (Fennoscandia) have high beta diversity (Figure CS2.3, CS2.4). Therefore, the value of the species-rich European areas lies not only in the species richness, but also in the rapid spatial species turnover. Moreover, it is also worth conserving the species poor areas in the north because they are unique – not only when compared with the rest of Europe, but also when compared with adjacent areas within Fennoscandia itself.

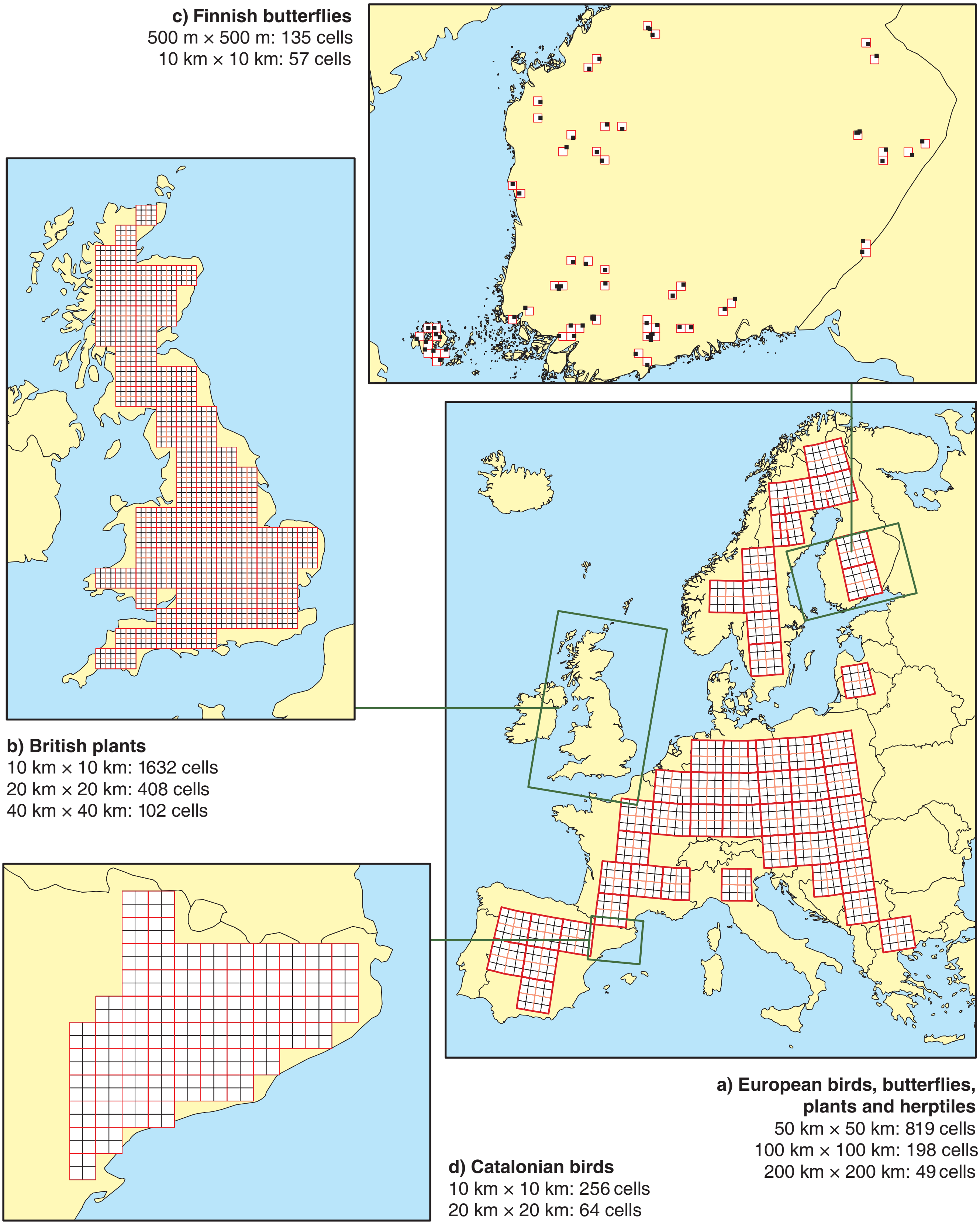


Figure SC2.1. Nested UTM grids used for the analyses of pan-European beta diversity patterns; a) across continental Europe, b) across the UK, c) Finland, and d) Catalonia. The different grains always cover the same area. Areas within the largest grid cells that overlapped sea, lack land-cover data, or those insufficiently surveyed were removed.

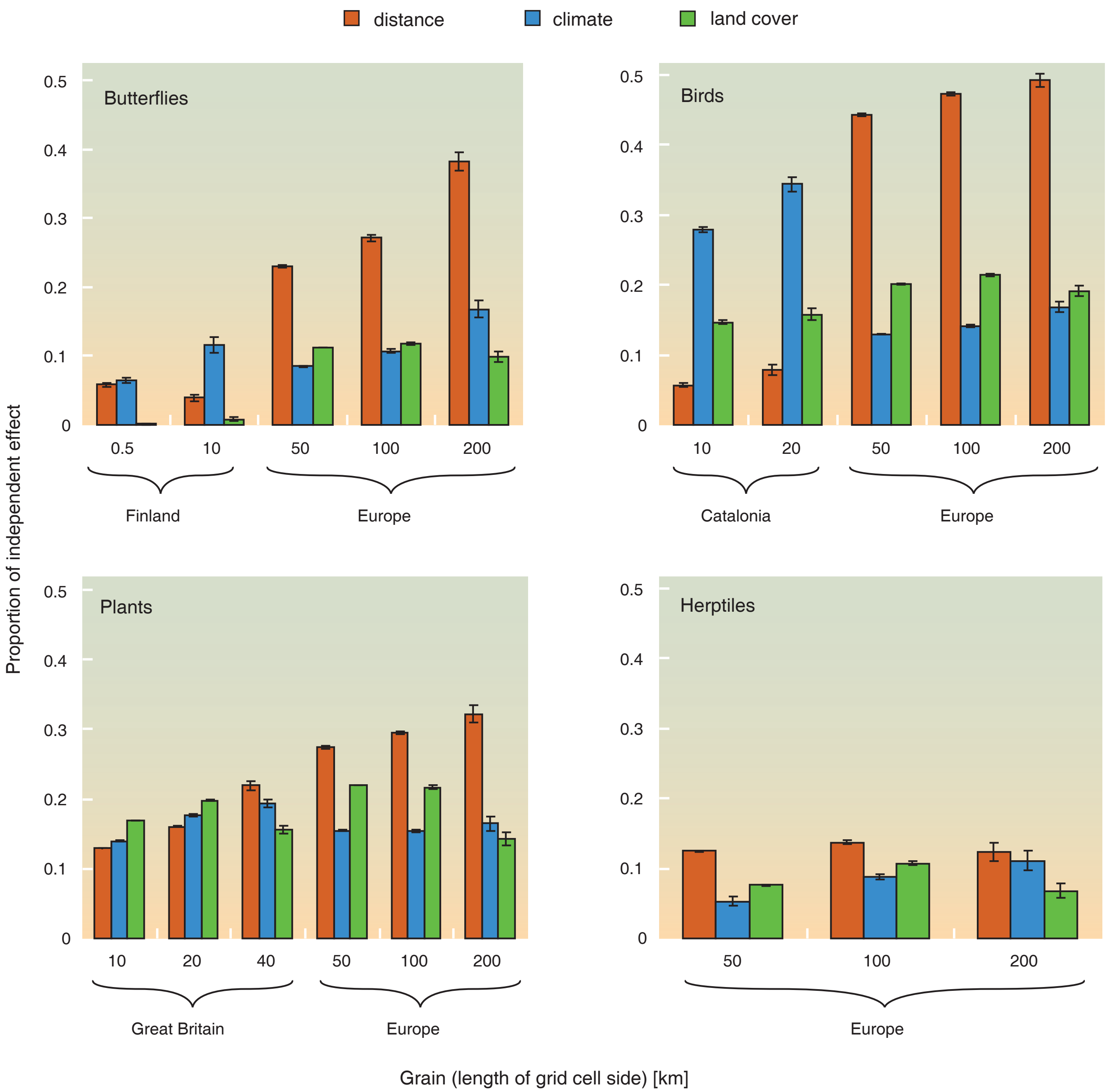


Figure CS2.2. Independent effects of climatic, land-cover, and geographic distances on beta diversity (measured as βsim index which estimates only pure species turnover controlled for the effect of different diversities in sampled areas) at various grain resolutions. At large scales geographic distance is the most important factor, while at the small scales climatic dissimilarity plays a major role in shaping patterns of beta diversity.

Case Study 3: Scale-dependence of biodiversity changes

Since the Rio summit in 1992, the issue of biodiversity loss has been high on the global list of priorities. Yet it is surprisingly difficult to measure whether biodiversity has increased or decreased, in part because of issues of scale. If biodiversity is intrinsically tied to scale, it is logical to assume that biodiversity change is also scale-dependent (Box 2). In fact, it is perfectly possible to have net *increases* in the species richness for each site in a landscape, but still have a *decrease* in species in the landscape as a whole.

While many researchers and conservationists have examined biodiversity change in the past, the issue of scale-dependent change has only recently been considered. Keil et al. (2012) examined hoverfly records in two European countries, the Netherlands and United Kingdom, examining biodiversity database records from before and after 1980. Only subtle scale-dependence in species richness change were found in the Netherlands, but strikingly different patterns of diversity change were found at different scales in the UK, with substantial declines in species richness at local scales shifting to either no change or even diversity increases at national scales (Figure

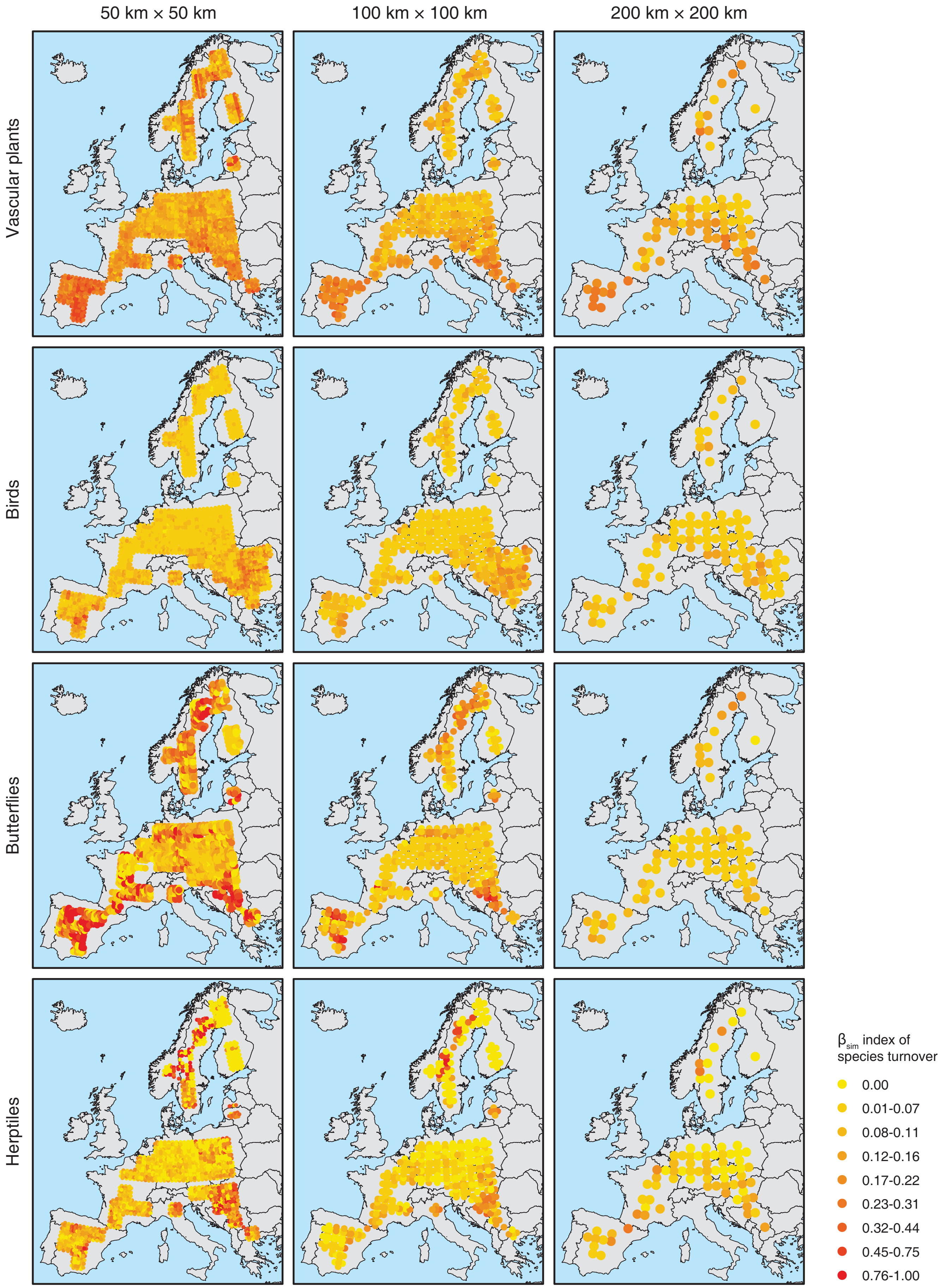


Figure CS2.3. Geographic patterns of beta diversity (measured as βsim) at first distance class (all pair-wise comparisons of all adjacent grid cells) for the four taxonomic groups at three grain resolutions. βsim value of 0 means identical species composition and value of 1 means completely different composition of species.

Figure CS2.4. Beta diversity does not depend on species number. One may find high values in species rich areas, e.g., in Mediterranean regions (left) (photo: Mathias Scholz/UFZ), as well as in species poor areas of Scandinavia (right) (photo: Reinhard Klenke).

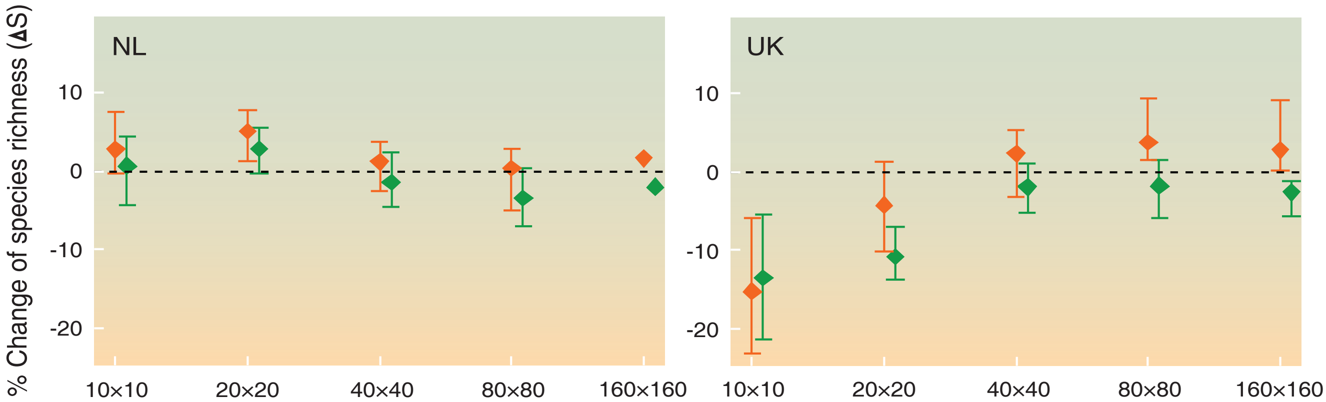


Figure CS3.1. Changes in species richness in hoverflies in the Netherlands (NL) and United Kingdom (UK) at different spatial scales. Orange symbols represent analyses comparing pre- and post 1980 records, while green symbols represent analyses comparing two equal time periods (1954–1979 and 1980–2005). Note that species richness showed signs of increasing in the NL at relatively fine spatial scales, but not at coarser scales, while in the UK substantial fine-scale declines coincide with either no change or even richness increases at coarser scales.

CS3.1). Carvalheiro et al. (2013) have additionally assessed shifts in beta diversity across scales, examining a range of plant and pollinator taxa (including bees and butterflies, as well as hoverflies), dividing biodiversity records into three 20-year periods, stretching from 1950 to 2009. The study showed evidence that biodiversity declines in most of these groups (at multiple scales) had slowed substantially in recent decades, but it also provided evidence of increasing biotic homogenisation in almost all of the taxa (Figure CS3.2). This suggests that the scaling of biodiversity in these groups is continuing to shift.

Taken together, these studies suggest that we need to take spatial (and temporal) scale into account when trying to assess how biodiversity has changed. It may well be that our biodiversity goals may themselves be scale-dependent. Thus for example, the role of pollinators or biocontrol agents in providing ecosystem services to agriculture may depend on diversity measured at a fine scale (that of a field or landscape). On the other hand, our conservation goals (such as those adopted at the Rio summit and in subsequent accords) may be more concerned with maintaining biodiversity at a national, continental or even global scale.

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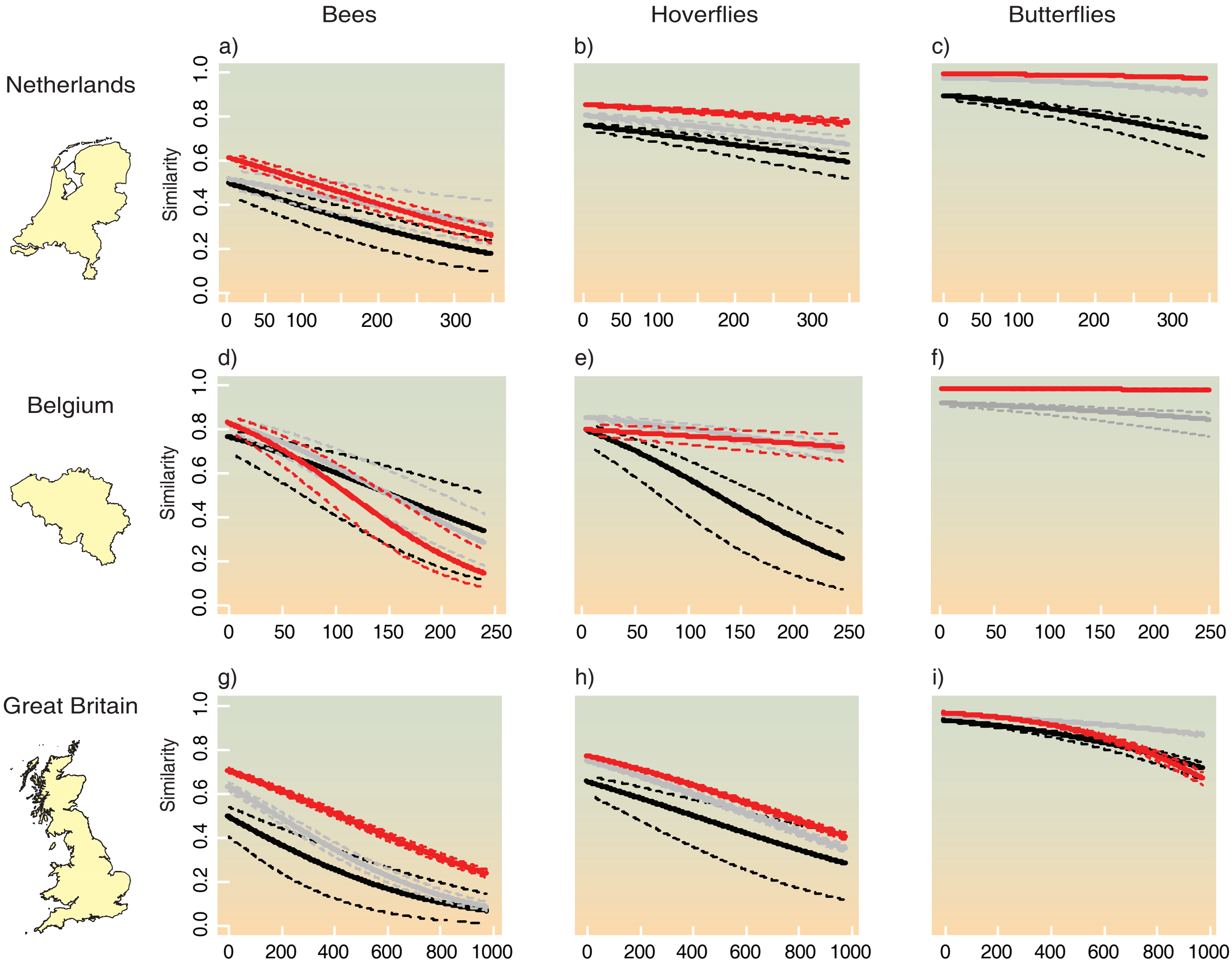

Figure CS3.2. Biotic homogenisation in NW European pollinators. These panels represent the similarity between species composition of samples taken different distances apart (measured using 1-βsim) for three groups of pollinating insects (bees, hoverflies and butterflies) over time. In most cases, recent decades (represented by the red lines) show higher similarity at a given distance than was found in the mid 20th century (black lines).

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Scaling of biodiversity change caused by land-use change

Riccardo Bommarco, Lorenzo Marini

Introduction

Understanding how biodiversity is impacted by land-use change at local to regional spatial scales is an outstanding challenge. First, species' responses to such changes depend on their specific life history traits and dispersal characteristics. Second, land-use changes can vary in nature and take place at contrasting spatial scales. Changed management of a single habitat, such as an arable field, a grassland, or a forest patch, can have an impact on a community by altering habitat quality at a small spatial scale. Habitat loss through land-use conversion and changed management occurring across larger spatial scales can have completely different and potentially severe consequences for biodiversity, where organisms struggle to survive, and find resources and mates in a modified landscape.

The massive destruction or degradation of natural and semi-natural habitats and the consequent increase in habitat isolation, have been identified as major threats to biodiversity and ecosystem functioning. Semi-natural grasslands are particularly threatened by land-use conversion and management intensification. Grasslands harbour an enormous richness of plants, insects, and vertebrates, and are of immense conservation value in human-managed landscapes across Europe. Impacts on biodiversity by habitat loss and isolation have received considerable attention, but this has mainly concerned impacts on populations of individual species. Influence of land use change on communities has also been studied, but with a rather narrow focus only on impacts on overall species richness.

Much less attention has been given to exactly which species in a community go extinct first as a result of land-use changes at the local to landscape scale, and how this depends on species life-history traits linked to local (e.g., competition and persistence) or regional (dispersal) processes (Henle et al. 2004). Such an analysis would also provide information about the relative importance of local *vs.* landscape scale processes in determining community composition, and thereby the spatial scale at which species are most likely to be impacted by land-use change. Such knowledge may enhance the efficacy of conservation efforts.

Another property of communities that has been poorly studied in relation to habitat loss is community

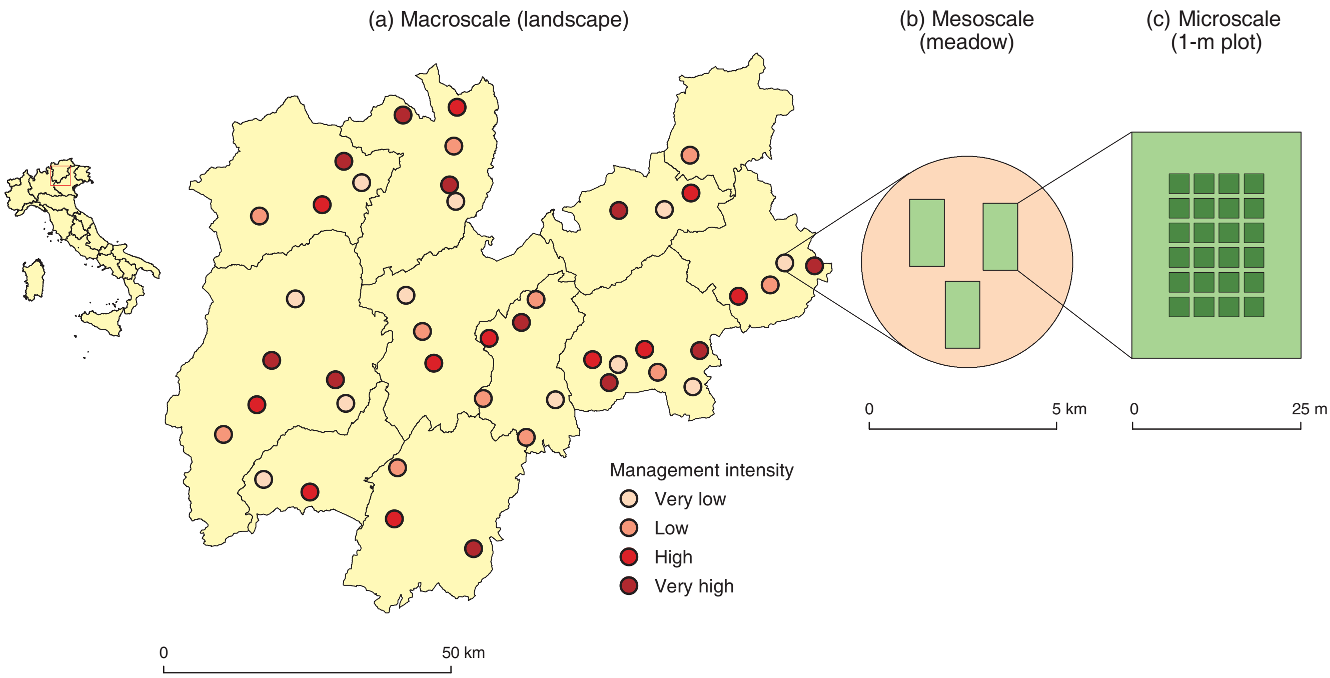

Figure 1. Nested sampling design performed at three spatial scales in montane meadows managed for hay production in the Trento region, Italy: (a) macroscale (landscape), (b) mesoscale (meadow) and (c) microscale (from Marini et al. 2012b).

abundance composition. This is often measured as community evenness, which has been shown to play a fundamental role in governing several ecosystem functions (Hillebrand et al. 2008). However, we still have a very fragmentary understanding of the drivers of community evenness and how it might shift in relation to land use, especially for terrestrial communities of mobile organisms.

The approaches mentioned above are mainly based on comparisons of community composition among local communities (α-diversity) that give information of local community subjected to global change. It is, however, difficult from such studies to discern impacts on the distribution of species at larger scales, and on the turnover in species composition among communities (β-diversity). For instance, two communities may have the same number of species, but have entirely different species in each community. These communities have a high β-diversity with a high degree of turnover between them. Two communities can also have a common set of species, but differ in species richness, i.e., where one of the communities harbour additional species. This will also render a high β-diversity between them, but in this case with a strong nested component since they partly have the same species.

It is eventually the biodiversity within an entire region (γ-diversity) which should be of prime conservation interest. Partitioning γ-diversity into β-diversity components based on richness differences and turnover, and relating these patterns to life history characteristics and drivers (e.g., land use, human population, and climate) across spatial scales, can reveal how land use and environmental changes impact biodiversity distribution across spatial scales. It can also provide critical information for conservation planning. The number of species in a local community (α-diversity) provides some basis for assessing the conservation value of local areas. In addition to this, the turnover of species between local areas, that determines the regional diversity, can indicate the optimal spatial arrangement and management of conservation areas in a region.

In the following we outline a series of studies that address impacts of land use change on community life history traits composition, evenness of abundances, and α-, β-, and γ-diversity distribution among communities across European regions.

Impacts on community traits composition

Local species richness is known to decrease with shrinking habitat area and increasing isolation. However, the risk of extinction is not equal among

Figure 2. Example of extensively managed semi-natural grassland (Val di Fassa, NE, Italy) (photo: Lorenzo Marini).

Figure 3. *Decticus verrucivorus* (photo: Lorenzo Marini).

species and depends on their similar life-history trait attributes (Pe'er et al. 2014). Species with particular traits might be more prone to extinctions compared to other species, and this will result in shifts in community traits composition and depend on the extent of habitat loss and isolation. For instance, species occurrence in a habitat fragment can depend on local persistence or ability to disperse. Low ability to persist and disperse is expected to increase the risk of going extinct as a result of habitat loss. Previous studies have explored such patterns for grassland insects (e.g., Bommarco et al. 2010), but less is understood for the plants.

We collected published information from 19 studies on fragmented forest and grassland plant communities across Europe. We examined whether risk of extinction due to habitat loss could be explained by life-span, clonality, and seed weight. We found that a larger proportion of forest species, compared to grassland species, were affected by habitat loss and increasing isolation. The proportion of long-lived and clonal plants decreased with habitat area. Seed weight, a trait related to dispersal and recruitment, was associated with habitat isolation, but in different directions for the forest and grassland habitats. The results thereby partly challenge earlier views: we found that shrinking habitat size will diminish the number of clonal, long-lived, and large seeded plants in remnant forests and result in fewer clonal, small seeded species in grassland fragments (Lindborg et al. 2012).

We also performed a much more detailed analysis based on primary data from a number of studies on the distribution of plant communities in 300 grassland fragments in five regions in four countries across Europe. To ensure consistency in the data, we standardized each species' taxonomic affiliation and life-history traits. For each fragment we extracted standardized landscape measures from original geographical data, and explored how plant trait community composition was affected by habitat area, degree of isolation, and their interactions across Europe. Plant species richness was consistently negatively affected by habitat loss, whereas impacts of habitat isolation were not evident. However, different species reacted differently depending on their life-history traits. Traits linked to species persistence (competitive ability and annual life cycle) linked to local process, and dispersal (animal aided dispersal) which affects larger scale dynamics of plant communities, emerged as traits enabling species to cope with habitat loss (Marini et al. 2012a). From these results it appears that an efficient conservation strategy is a combination of decreasing the spatial isolation that remnant grassland plant communities appear to suffer from, coupled with conservation and restoration interventions homing in on improving habitat quality for poorly competitive, abiotically dispersed, and clonal perennial species.

Impacts on evenness of abundances in communities

Extinctions due to intensified land use and global change are often preceded by shifts towards uneven communities with few abundant species. These shifts also affect ecosystem functions (Hillebrand et al. 2008). Despite this, only a few studies have explored how land-use change affects evenness of abundances. This is probably because most biodiversity research focuses on measuring impacts on species richness, and there exists a view that species richness and evenness are positively related to each other, where a few dominant species become abundant in communities with few species. This contention has recently been challenged by empirical studies from aquatic ecosystems suggesting that these two components of diversity should be considered separately. For instance, Soininen et al. (2012) have shown that several local processes such as competition, predation, and succession, as well as large scale dispersal, can affect evenness without changing species richness. To assess if this is the case also for terrestrial grassland communities and whether species richness and evenness are similarly affected by habitat loss and fragmentation, we collected data on abundances of flower-visiting wild bees and butterflies from a range of studies in grassland patches across Europe. We tested the effect of habitat area and isolation from other grasslands on evenness and species richness of the flower-visiting insects in each patch. We found that species richness declined with decreasing habitat area. Yet we also found a that evenness decreased with increasing area, and increased with increasing connectivity. A deeper investigation showed that communities in small habitat fragments to a larger extent were composed of mobile and generalist species as compared to those in larger fragments. The more even communities in small habitat islands might be maintained by highly mobile species that move more frequently and easily within the matrix and among patches. The analysis of trait composition indicated the species abundance composition in larger remnant habitat fragments is more determined by local processes, such as competition, whereas the composition of communities in small and less interlinked habitat fragments to a larger extent are determined by dispersal (Marini et al. 2014). The study, thus, showed that the two community measures, species richness and evenness, responded in different directions to habitat loss and increasing fragmentation and are probably driven by different mechanisms, and by a combination of local and larger scale processes. The resulting community composition seems to depend on the relative strength of these processes.

Figure 4. *Brenthis hecate* (photo: Paolo Paolucci).

Species turnover among communities across spatial scales

Building on the understanding that dispersal ability plays an important role in determining community composition, it is important to recognize that only few tests have been performed on the impact of dispersal ability on the distribution and turnover of species across space. In the absence of high resolution biodiversity data and fine scale land-use information that correctly maps habitat suitability to specific groups of species, we empirically investigated communities of grasshoppers inhabiting hay meadows that differed in management intensity (as defined by fertilisation and mowing regime) in the province of Trento in north-eastern Italy. We examined whether mobility of grasshoppers might modify β-diversity along a gradient of management intensity at three nested spatial resolutions: 1 m^2 plots within a meadow, 1000 m^2 meadows within a landscape, and 19.6 km^2 landscapes within the region (Figure 1). Grasshopper community composition varied most when compared over large spatial scales and β-diversity at the landscape contributed the largest part of the regional (γ) diversity. Mobility did explain a large part of the β-diversity, where sedentary species contributed to a greater proportion of β-diversity across all scales as compared to mobile species (Marini et al. 2012b). Our case study confirms that dispersal capacity significantly affects turnover of species across spatial scales. From the point of view of grasshopper conservation, the results suggest that a network of high value meadow habitat should be protected throughout the entire region to maintain a rich pool of grasshopper species in the region.

Concluding remarks

Conservation efforts are generally targeted at saving highly diverse habitats, i.e., those that harbour a high α-diversity, or to the protection of targeted species. Local processes are obviously important in shaping species' communities, but dispersal processes are increasingly acknowledged as critical in explaining the distribution, abundance, and persistence of species across landscapes. Our results show that interventions focused on conserving and restoring local habitat quality need to be complemented with efforts to assess species' turnover and develop the best means to reduce the risk of habitat fragments from becoming homogenized, i.e., hosting the same sets of species. Our studies highlight two critical processes responsible for homogenization: the domination of generalist species with high mobility, and the invasion of species into increasingly specialized habitats. By decreasing the spatial isolation of remnant species-rich habitats, such as permanent grasslands, one may avoid the risk that sedentary species will decline faster. At a larger scale it is also important to ensure the preservation of unclustered habitat remnants, to maintain the natural beta-diversity which occurs across geographic distances. Such efforts have the potential to mitigate large-scale negative impacts of intensified land-use, facilitate species range-shifts, and potentially increase the diversity and with it the resilience of native communities to invasions by alien species.

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The interface between conservation areas and agriculture: Functional spill-over and ecosystem services

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Introduction

Conservation areas harbor a high diversity of plants and insects, including a high proportion of rare and endangered species with special habitat and food requirements, and limited geographic distribution. In central and large parts of northern Europe, most grassland conservation areas originate from extensive management by grazing or mowing and are embedded in mixed agricultural landscapes (Figure 1). The diversity maintained in these conservation areas is threatened by ongoing habitat loss, due local management intensification, abandonment of historical land use, or reforestation, and also large scale land use intensification such as urban development (Steffan-Dewenter and Tscharntke 2002). The consequences for the functional connectivity of conservation areas, extinction debts, and potential negative impacts of agriculture on conservation sites remain largely unknown (Krauss et al. 2010). In the past, conservation efforts mainly focused on maintaining or improving local habitat quality by implementing management schemes and partly also considered connectivity with other conservation areas (Brückmann et al. 2010, Figure 2). However, recently it has become increasingly clear that this perspective ignores important impacts of the surrounding landscape matrix on conservation areas at different spatial and temporal scales (Blitzer et al. 2012, Tscharntke et al. 2012). This is particularly relevant as most conservation areas in human managed landscapes in Europe are small and fragmented and thereby more heavily influenced by the landscape in which they are embedded, compared to much larger conservation areas in other parts of the world. The relatively greater influence of the surrounding landscape on European conservation areas needs to be considered in order to improve the success of current conservation efforts and to meet the targets of the EU biodiversity 2020 strategy. An improved understanding of landscape-scale interactions between conservation areas and surrounding habitats can provide evidence to help direct both the EU green infrastructure programme and the Common Agricultural Policy.

From an animal ecology perspective, landscape-scale interactions can be bidirectional. For instance, species that occur in semi-natural grassland conservation areas may either use additional resources from other surrounding habitats, or the grasslands may be colonized or exploited by organism from the surrounding landscape matrix. Such resource or organism fluxes between different habitat types are termed "spill-over", with important consequences for population dynamics, species composition and ecosystem functions in conservation areas and the surrounding matrix (Blitzer et al. 2012). Landscape interactions can have positive consequences for conservation areas and could be used to mitigate possible extinction

Figure 1. Calcareous grassland in a mixed agricultural landscape in Germany (Lower Franconia) (photo: Jochen Krauss).

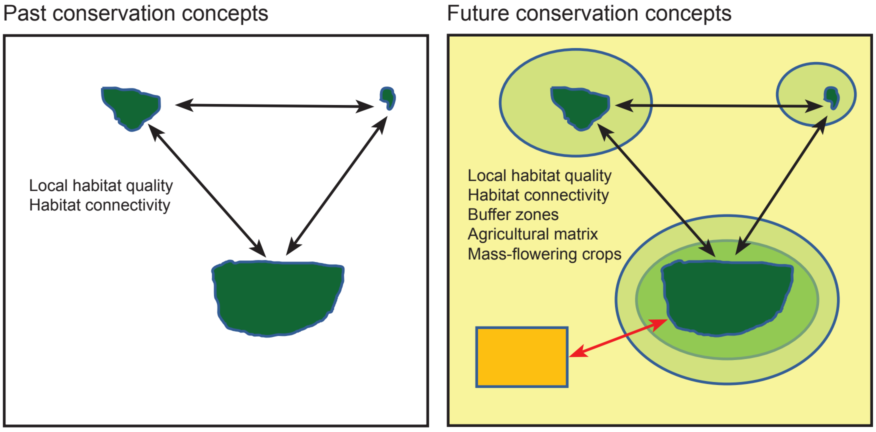

Figure 2. Past and current concepts to manage conservation areas at multiple spatial scales. Green irregular shapes represent conservation areas (dark green – high habitat quality, pale green – low habitat quality. Circles indicate buffer zones with extensified management schemes. Yellow illustrates the agricultural matrix and orange mass-flowering crops. Arrows illustrate the flow of resources and organisms between habitat types.

debts by establishing buffer zones with more extensive grassland management, organic farming, or other targeted agri-environmental schemes (Figure 3). Such interventions can provide food or other limited resources for endangered species, enlarge the available habitat area, enhance connectivity, and reduce direct negative impacts of agriculture (e.g. pesticide drift or nitrogen deposition). However, the interface between conservation areas and agricultural habitats also potentially includes some poorly understood direct and indirect risks for protected areas. For example, mass-flowering oilseed rape provides an attractive additional pollen resource for pollinators in grassland habitats. This can enhance the population size of generalist pollinator species that can exploit this additional resource. Other, more specialized species, might suffer as a result of greater competition for flower resources within grasslands, due to the more abundant generalists after the end of the mass flowering. Thus, the effects of spatial spill-over might vary in time following mass flowering crop phenology. A further poorly studied impact is the possible predation and parasitism of larval stages of rare insect species in grassland habitats. Annual crops provide a huge resource in terms of plant biomass, and thereby build the basis of a food chain with large populations of herbivores and predators. After ripening and harvest of crops generalist and mobile predators could switch to the remaining uncultivated habitat islands and increase top-down predation in conservation areas (Schneider et al. 2013). These impacts might change antagonistic and mutualistic biotic interactions and thereby threaten species with particular conservation importance.

On the other hand, conservation areas and other semi-natural habitats serve as source for beneficial organisms that provide ecosystem services such as pollination and biological pest control in the agricultural matrix (Tscharntke et al. 2005). For example, visitation rates and species richness of pollinators as well as related fruit set and yields of insect pollinated crops decline with the distance from non-arable pollinator habitats. In this case, the conservation of semi-natural and natural habitats in an agricultural matrix can be supported by the economic value provided by the functional diversity of wild organisms. However, it has been rarely questioned whether the management objectives for protected areas are congruent with schemes that aim to maintain or increase ecosystem services for agriculture. Indeed, there may be potential conflicts, as actions promoting ecosystem services will generally prioritise enhancing the abundance of a few common species delivering the service, over protecting rare species of conservation concern.

Thus, future strategies for a successful, long-term management of conservation areas require a landscape perspective that integrates the connectivity of semi-natural or natural areas, the targeted implementation of agri-environmental schemes and the consideration of arable crop dynamics. Furthermore, the reliance of agroecosystems on ecosystem services provided by semi-natural habitats requires a closer link between agricultural and conservation policy instruments and considerations of the economic benefits of biodiversity for maintaining stable and high yields, but also consideration of possible trade-offs between pure conservation and ecosystem services objectives (Bommarco et al. 2013).

Figure 3. Flower strips in the neighborhood of a protected grassland area as an example for possible management schemes in buffer zones of conservation areas (photos: Béatrice Portail).

In the framework of the EU-project SCALES, and in close cooperation with the EU-project STEP (Status and Trends of European Pollinators) we have addressed these issues. First, we assessed whether grassland plant and insect communities were affected by the composition of the surrounding landscape. Second, we assessed the functional consequences of spatial and temporal spill-over for pollination of wild plants and predation of larval stages of insects in grassland conservation areas. Finally, we addressed the spatial and temporal interplay between conservation areas, mass-flowering crops and other habitat elements with a focus on crop pollination and pest control services.

Impacts of surrounding landscape composition on species in conservation areas

Patch area together with degree of isolation from other remnant conservation fragments are well known to affect species richness in grassland communities. There is an increasing interest in assessing how the characteristics of the landscape surrounding the conservation area influence population persistence and species diversity in fragmented landscapes. In SCALES we examined the impacts of grassland fragmentation, focusing on how the land use in the landscapes surrounding the grassland fragments has affected species richness of plants, butterflies, bees, and hoverflies. We gathered information on these organisms from multiple grassland patches of different sizes and isolation, embedded in landscapes dominated either by forest, arable land or a mix of these across Europe. We found that the surrounding landscape type, also often referred to as matrix type, had contrasting effects depending on the taxa. Species richness of plants and butterflies was lowest in patches in landscapes dominated by arable land and highest in forest-dominated landscapes. Hoverflies and bees, on the other hand, were more negatively impacted by habitat loss in forest-dominated landscapes (Öckinger et al 2012a and b). Differences in response to matrix composition probably depend on different ecological life history characteristics and resource requirements among organism groups. Forests, for instance, are likely to provide additional flower and plant host resources for butterflies, and a higher plant diversity in grasslands embedded in forested landscapes can result from habitat generalists invading these patches. An agricultural landscape is likely to be a more hostile matrix for many grassland species due to high disturbance levels, pesticides, and depauperate food and nest resources. On the other hand, some hoverflies might benefit from the presence of agricultural pest insects on which they feed (Meyer et al. 2009). In any case, our results support the view that considering matrix quality is important when developing efficient conservation schemes. Thus, 'softening' the agricultural matrix by adding vital resources or stepping stones, for instance, via targeted agri-environmental schemes or buffer zones, is likely to have a positive impact on the communities in the remnant conservation areas. One important change in agricultural landscapes is the growing area of mass-flowering crops for biofuel production. In the next section we address the functional consequences of this shift in the agricultural matrix.

Mass-flowering crops versus wild plants: Competition for pollinators?

More than 80% of all wild plant species rely on animal pollination with bees and hoverflies as the most important pollinator taxa in temperate climates (Ollerton et al. 2011, Figure 4). On a typical European semi-natural permanent grassland, several hundred plant species co-occur within a rather small area and heavily rely on the pollination services provided by a community of pollinators with different life history strategies, resource specialization, and flight phenology. While previous studies show that pollination limitation is more pronounced in small and fragmented habitats, the impact of mass-flowering crops in the neighborhood of species rich grassland areas has not been considered in the past. There are different possible scenarios

Figure 4. Important pollinator taxa in central Europe. Solitary wild bee species *Andrena hattorfiana* (photo: Ingolf Steffan-Dewenter).

for the effect of mass-flowering crops during and after their flowering period. During the flowering of mass-flowering crops, pollinators could be lured away from wild plants to more rewarding crops such as oilseed rape or sunflower. Thereby, wild plants might suffer more severe pollination limitation; this effect should be strongest in small habitat fragments with small remnant plant populations. On the other hand, mass-flowering crops could also enhance local pollinator densities on grasslands by acting as a magnet that locally increases overall pollinator densities, even for wild plants flowering simultaneously, or by supporting faster population growth of social species. After flowering of oilseed rape, larger populations of social bee species could result in a temporal spillover with higher flower visitation rates of wild plants in grassland areas compared to areas without mass-flowering crops in the surrounding.

A study on the endangered, spring-blooming grassland herb *Primula veris* (Figure 5) indeed shows, that landscape-wide dilution of bee pollinators by mass-flowering oilseed rape results in a reduction of seed set in *P. veris* by up to 20 percent in grasslands with a high cover of oilseed rape in the landscape matrix (Holzschuh et al. 2011). We performed further experiments to address the pollination functions of pollinators for focal wild plant species flowering during or after the bloom of mass-flowering crops. These studies measured pollinator visitation rates in relation to landscape parameters and consequent seed or fruit set. For one focal plant species, the horseshoe vetch *Hippocrepis comosa* (Figure 5), we found a significant reduction in the visitation rates of solitary wild bees and bumblebees during the flowering period of oilseed rape. Thus, in line with the results of Holzschuh et al. (2011), these results imply that competition for pollinators between crops and wild plants plays an important role in different regions and plant families. The flowering period of horseshoe vetch only partially overlapped with oilseed rape flowering, at least in the study year, and thus we could evaluate temporal shifts in visitation rates after flowering of oilseed rape ceased. Interestingly, we found after the end of oilseed rape flowering, a positive relationship between the visitation rates of bumblebees on *H. comosa* and the proportion of oilseed rape in the surrounding landscapes. Accordingly, our data confirm the hypothesis that mass-flowering crops enhance the abundance of social pollinators in adjacent seminatural grasslands. At first glance this seems to be a beneficial effect of crops on conservation areas, however, we found a second less expected relationship for solitary wild bees. In this case, the visitation rates still declined with increasing cover of oilseed rape (after the end of oilseed rape flowering). This pattern could be interpreted as evidence for competitive replacement of solitary bees by social bumblebees. This would imply that mass-flowering crops not only outcompete wild plants in protected areas for pollinators, but also cause changes in the structure of pollinator communities and consequently in plant-pollinator interaction webs. How this might affect pollination services, reproduction and genetic diversity of wild plant populations in conservation areas requires further research.


Figure 5. Studied endangered grassland herb species with potentially enhanced pollination limitation due to mass-flowering crops. Left: Horseshoe vetch *Hippocrepis comosa* (Fabaceae). Right: cowslip *Primula veris* (Primulaceae) (photos: Birgit Jauker).

Impact of predator spill-over into conservation areas

Not only mutualistic but also antagonistic interactions in conservation areas might be modified directly and indirectly by spill-over from the surrounding agricultural matrix. As outlined above, the seasonal dynamics of crops and the huge amount of plant biomass provided by crops might result in directed movements of mobile and generalist predators into grassland conservation areas. In semi-natural grasslands, a large number of endangered herbivorous insect species with specialised food plant requirements can often be found. Increased densities of generalist predators from the agricultural matrix might reduce the survival rates of developmental stages of endangered grassland insect species, and further reduce the population size of species with already small and isolated populations, with potentially negative consequences for their long-term survival. We performed a predation experiment on semi-natural permanent grasslands, where half of the grasslands were adjacent to coniferous forest and half bordered by cereal crop fields. To quantify predation rates of ground-dwelling predators we used eggs of the seven-spotted ladybird beetle (*Coccinella septempunctata*) which were exposed as prey items on the ground. Predation rates were measured in two study periods, one before and one after the harvest of cereal crops (Figure 6; Schneider et al. 2013). In each study period, we found higher predation rates when coniferous forest was the adjacent habitat. However, this result was only significant on cool days, whereas on warm days, prey items were consumed to a

Figure 6. Beetle eggs placed on calcareous grasslands to measure predation rates (photo: Gudrun Schneider).

greater extent which presumably did not allow the detection of possible differences between adjacent habitat types. In contrast to our expectation, we found no enhanced predation rates after crop harvest in grasslands adjacent to cereal fields. Our results indicate, that not only the interaction between perennial and arable habitats, but also edge effects of different perennial habitats, can expose conservation areas to the spillover of antagonistic species; a risk which should be considered in future conservation strategies for semi-natural grassland habitats.

Crop pollination services and biological pest control

Semi-natural and natural protected areas such as semi-natural permanent grasslands are not only the target of spillover from the surrounding landscape, they are also the source of important ecosystem services delivered to agricultural habitats. More than 70% of all major crops rely on animal pollination (Klein et al. 2007), and up to 30% of yields are lost due to antagonistic interactions with herbivores, pathogens and weeds, despite intensive pesticide application (Bommarco et al. 2013). Past research clearly demonstrates that the proportion of semi-natural habitat in a landscape is related to the provision of crop pollination and biological pest control services (Tscharntke et al. 2005). Typically, the delivery of an ecosystem service, such as crop pollination, declines with increasing distance from pollinator habitats.

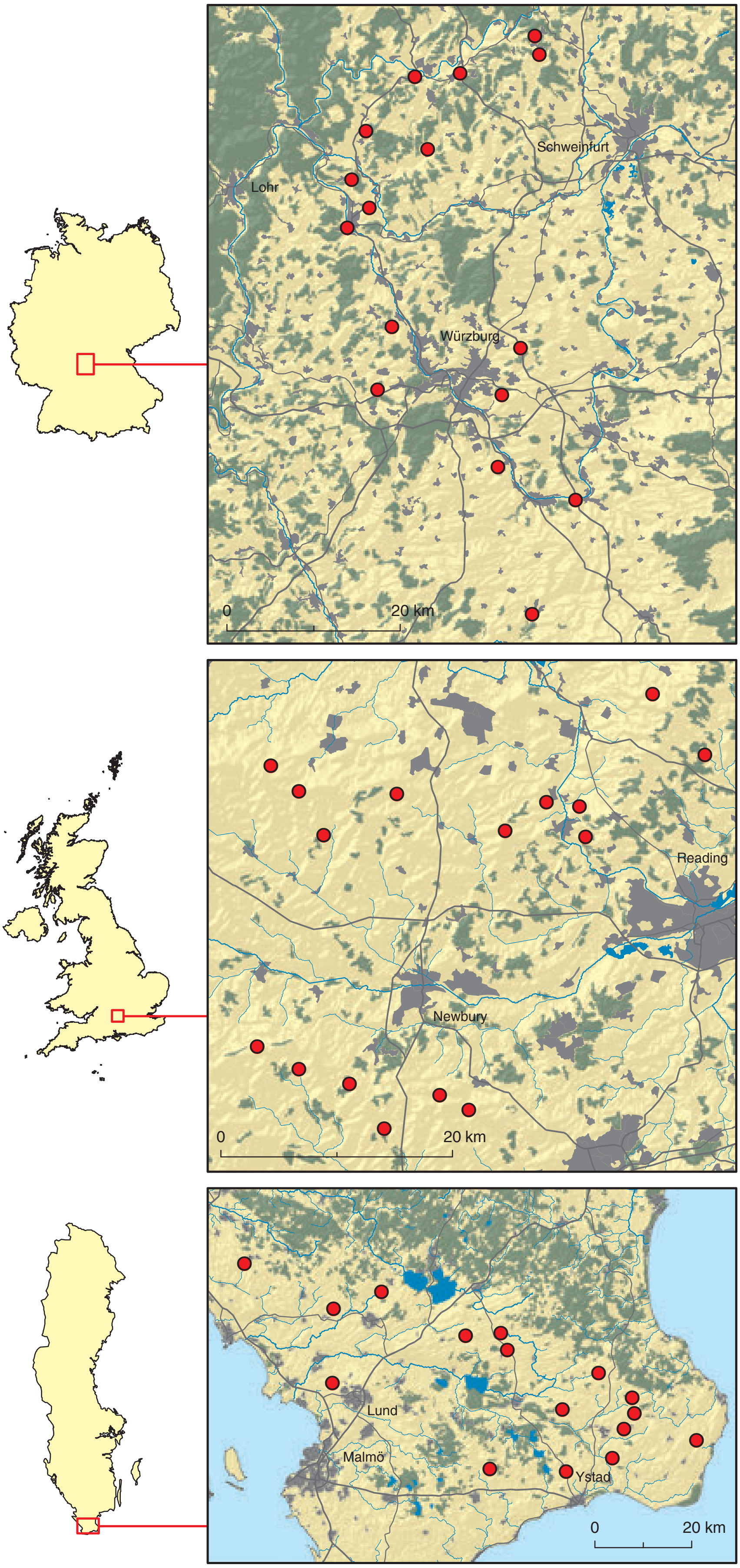

Figure 7. Maps with locations of study sites in Germany (top), UK (middle) and Sweden (bottom). The study region in Germany is located in Lower Franconia; in UK in the South East of England, including the North Wessex, Downs, Berkshire Downs, Hampshire Downs and the Chilterns and in southern Sweden in Skane.

For example, typically about half of pollinator richness and visitation rate can be lost in crops a kilometer from a semi-natural habitat fragment with a parallel reduction of yields and increased spatial and temporal yield variability (Ricketts et al. 2008, Garibaldi et al. 2011). Most importantly, not only visitation rates, but also the species richness of wild pollinators, plays a significant role in yield stability, underpinning the value of natural or semi-natural protected areas, not only for conservation of biodiversity, but also for the maintenance of ecosystems services (Garibaldi et al. 2013). However, a more detailed understanding of how landscape composition and configuration, the seasonal and temporal turnover of resources due to crop rotation at different spatial scales, and the implementation of agri-environmental schemes affect different ecosystem services is currently lacking. In the framework of the EU-projects SCALES and STEP we have established a joint three years study with replicated landscapes in Germany, Sweden and UK to address these issues. The study design allows us to assess in parallel plant-pollinator and pest-antagonist interactions in focal crops, and in relation to multiple landscape parameters (Figure 7). Our results indicate that pollinator dilution in mass-flowering crops is causing a significant reduction of visitation rates in landscapes with a high proportion of mass flowering crops like oilseed rape or sunflower (Riedinger et al. 2014). This indicates that, not only the proportion of and distance to pollinator habitats is important, but also the relation of pollinator habitat area to mass-flowering crop area in a landscape. Thus landscape-wide management schemes for ecosystem services should also take into account the temporal dynamics and areas of arable crops. Similar, we found that biological pest control was influenced by the annual dynamics of crops, as well as the proportion and distribution of semi-natural habitats (Figure 8).

Perspectives for landscape scale management of protected areas and ecosystem services

In conclusion, it is critical that policy and management practices increasingly consider the complex spatial and temporal interactions between semi-natural habitats, cropped areas and conservation interventions. This will improve the management and conservation of protected areas and contribute to the maintenance or restoration of agricultural biodiversity as well as crop-related ecosystem services. This is essential due to the close links and interactions between fragmented conservation areas in central Europe and the mixed agricultural matrix in which they are embedded. In particular, the often ignored, but potentially negative indirect impacts of functional spill-over from agricultural to semi-natural or natural protected areas has to be studied in more detail and accordingly addressed in conservation schemes. Novel and targeted conservation area management schemes could create buffer zones surrounding protected areas to mitigate the impact of intensive agriculture, to enhance the population size of pollinators and pest control agents, and to maintain the diversity of protected and endangered plant and animal species. Possible management approaches supported by spatially targeted incentives could include a higher proportion of extensive grasslands, organic farming, temporal set-aside of arable fields, larger field margins and new linear habitat elements in these buffer zones. For agricultural systems, novel management concepts need to include the provision of ecosystem services by appropriate agri-environmental schemes and by tools that allow farmers to assess the yield consequences of temporal crop rotation patterns. Dual approaches may be needed: those which enhance ecosystem services by managing the commoner species delivering the service and those which focus on protecting rare species. Careful spatial targeting of appropriate interventions will be necessary to achieve win-win situations for nature conservation areas and crop production systems.

Figure 8. Biological pest control in oilseed rape (*Brassica napus*). The rape pollen beetle *Meligethes aeneus* is parasitized by ichneumonid wasps (photos: Gudrun Schneider).

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Conserving different kinds of biodiversity in different sorts of landscapes

Charles J. Marsh, Richard M. Gunton, William E. Kunin

Do we need different kinds of landscapes for different conservation goals?

Biodiversity is a complex concept. It includes the diversity of genes, populations, species and even whole ecosystems. Conservation policy should consider all these aspects, but policy-makers are also increasingly concerned about "ecosystem services" provided by natural systems; for example, the pollination of crops by insects nesting in grassland, or the regulation of water supplies via forests and upland habitats. A network of nature reserves, such as Natura 2000, should ideally help to conserve all the various aspects of biodiversity and multiple ecosystem services simultaneously.

Are all the different aspects of biodiversity and ecosystem services closely linked in practice? If a strategy designed for conserving one will generally do well for all of them, then the job of conservation planners is relatively easy. Best policy will of course be different in every context according to the sites concerned and the ecosystem services of interest, but in general, genetic diversity can help populations survive, and the long-term species-richness of a community or ecosystem relies on the persistence of populations within it. Moreover, the value and resilience of ecosystem services have often been linked to the biodiversity of ecological communities that provide them. But this kind of theorising is controversial (Box 1), and a more empirical approach is needed. While it is costly and time-consuming to test conservation strategies experimentally, modelling techniques allow us to explore options in greater detail and with greater generality. This chapter uses simulation models to explore how the sizes, shapes and proximity of nature reserves might affect the prospects for achieving different conservation goals.

Earlier debates (Box 1) rarely considered the spatial scales at which conservation actions are considered. The best regional strategy may not simply be scaled up to give a global template, or scaled down to give local recommendations. By accounting for the dispersal abilities and habitat requirements of species, we should be able to derive some general, scale-specific recommendations for designing and improving nature reserve networks.

We obtained a set of ecological simulation models that address different aspects of biodiversity (Box 2)

Box 1. Nature reserve design and the SLOSS debate

In 1975 Jared Diamond proposed a series of general design criteria for nature reserves. The best known of these was his suggestion that if we have the money to protect a certain amount of habitat, it is better to preserve one large patch than it would be to have lots of smaller ones of equivalent area (Diamond 1975). This was based upon the Theory of Island Biogeography, where larger islands may be able to support larger numbers of species for longer periods of time, largely because of reduced extinction rates. Other scientists suggested that a set of smaller reserves could sample a wider variety of habitats and species, and the ensuing discussion became known as the **S**ingle **L**arge **O**r **S**everal **S**mall (SLOSS) debate. Diamond's original reserve design recommendations also included a range of other principles (Figure 1) concerning the total area, proximity, connectivity and shape of reserves.

Almost 40 years later, there remains little consensus as to when and where Diamond's recommendations for reserve design are appropriate. Conservationists consider a range of ecological criteria and there may be trade-offs between them. For example, having reserves closer together may be good for conserving populations of mobile animals, as individuals can access resources from lots of reserves, but spreading reserves across a larger area may capture a wider range of habitat variation, and thus more species.

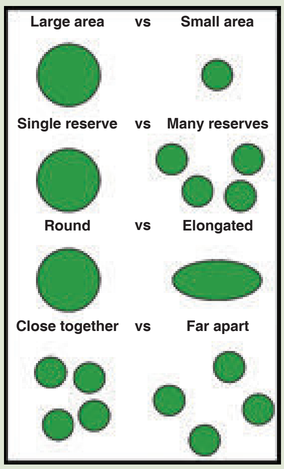

Figure 1. Principles for the design of nature reserve networks as proposed by Diamond (1975). The options in the left-hand column are preferable to those in the right.

Box 2. Simulating ecological processes

We initially used six simulation models to assess four different conservation criteria. In all the models we assume that species require the habitat patches for most of their life-cycles, but can disperse across the matrix between patches.

Table 1. Models used, with the conservation criteria assessed using each model (shaded cells).

Model	Genetic diversity	Population viability	Species richness	Pollination services
MetaConnect				
FunCon				
Spatial neutral model				
Metacommunity model				
Diffusion model				
InVEST				

MetaConnect (http://scales.ckff.si/scaletool/index.php?menu=5) models population dynamics and neutral population genetics for a species (typically animal) in a network of patches. Individuals reproduce and disperse while their genomes mutate and mix through mating; extinction may occur in some patches or whole landscapes, and different genotypes may dominate different patches. For each landscape, we obtained population sizes and total numbers of alleles across 10 loci when equilibrium was reached.

FunCon (Pe'er et al. 2011) models the movements of a mobile (typically bird) species foraging for resources and dispersing in a heterogeneous landscape. Individuals perform central-place foraging and dispersal, and the model determines how many may be supported by the landscape. For each 5-km landscape (a scale at which a bird's use of multiple patches is critical for the landscape's value), we used FunCon to obtain the maximum population of a songbird species it could support.

The **spatial neutral model** of Rosindell and Cornell (2007) simulates the dynamics of species (typically trees) that are ecologically identical. Habitat patches are always filled with individuals, which randomly die and are replaced – normally by offspring of the same species as a nearby individual (randomly chosen according to a dispersal kernel), but occasionally by a newly-evolved species. For each landscape, we used the model to obtain the species richness at equilibrium.

Our **metacommunity model** (Bocedi 2010) populates a landscape with multiple species (e.g. of woodland birds or small mammals) that compete for a resource, each with a niche requirement allowing survival in different parts of the landscape. Species also have differing reproduction and dispersal abilities. We used a textured version of each landscape, representing variation in a temperature niche, and ran the model with a realistic number of species of a given type to obtain the number surviving at equilibrium.

A **diffusion model** of pollination service was designed especially for this analysis. It assumes that wild pollinating insects nest in the habitat cells and diffuse uniformly in all directions, pollinating crop flowers whenever they reach the surrounding matrix. For each landscape we obtained maps of pollinator visitation rates, and converted these to fruit set (proportion of crop flowers producing fruit), for a crop that requires insect pollination, to give an estimate of the potential economic value of pollination service provided by the habitat.

InVEST (Natural Capital Project 2012) is a platform providing a more complex model for pollination service provision. The abundance of pollinators is modelled across a landscape with respect to their nesting sites and foraging resources, and pollination rates are calculated for each crop cell. We specified that pollinators nest in the habitat and forage in both habitat and crop, and for each landscape we obtained overall fruit set values, as with the diffusion model.

For population viability, we used FunCon to assess bird populations at the 5-km scale and MetaConnect to assess bird populations at the 50-km scale and small mammals at the 5-km scale. For species richness, we performed analyses using both the metacommunity model and the spatial neutral model; since these gave similar rankings of the landscapes in most cases, we present results from the metacommunity model as being more realistic. For pollination services, results from our diffusion model correlated very closely with those from InVEST, which we present here.

and used them to assess a wide range of landscape patterns according to four conservation goals.

The landscape patterns (Figure 2) came from a mixture of real habitat maps and patterns that we created to broaden the range of spatial structures. The patterns differed, for example, in the amount of habitat (either 2% or 10% of the total area), the number and size of patches (from a single large reserve down to hundreds of tiny ones), and in how evenly the habitat area was divided up. We also varied patch shapes and connectivity. Furthermore, we interpreted the same set of patterns at different spatial scales, so that they might represent either a 5-km or a 50-km landscape.

The simulation models came from a variety of sources within and outside the SCALES project, and are designed to assess the relative value of landscapes for different conservation goals (Box 2). We assessed three of these (genetic variation, popula-

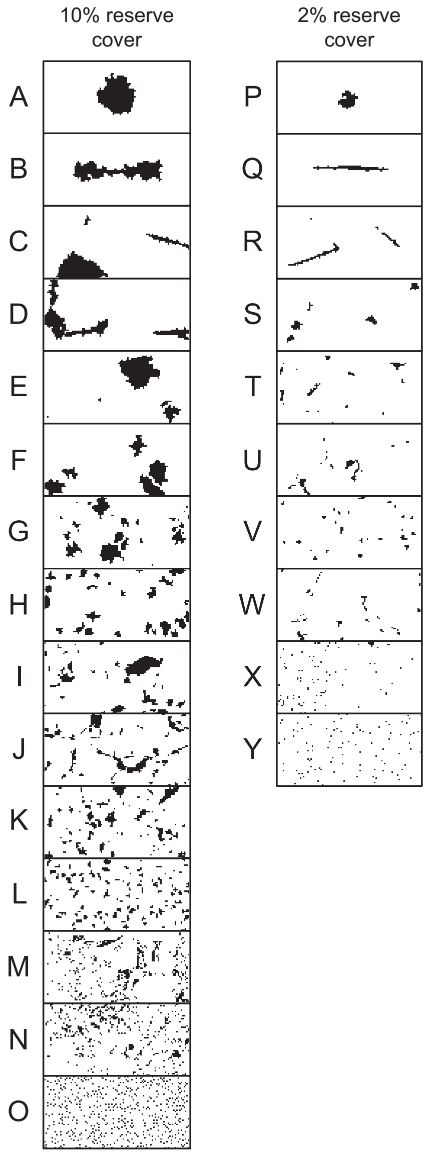

Figure 2. Some of the habitat patterns assessed, ordered by number of patches. The black cells represent nature reserves; in scenarios on the left these cover 10% of the landscape, whilst in those on the right they cover only 2%.

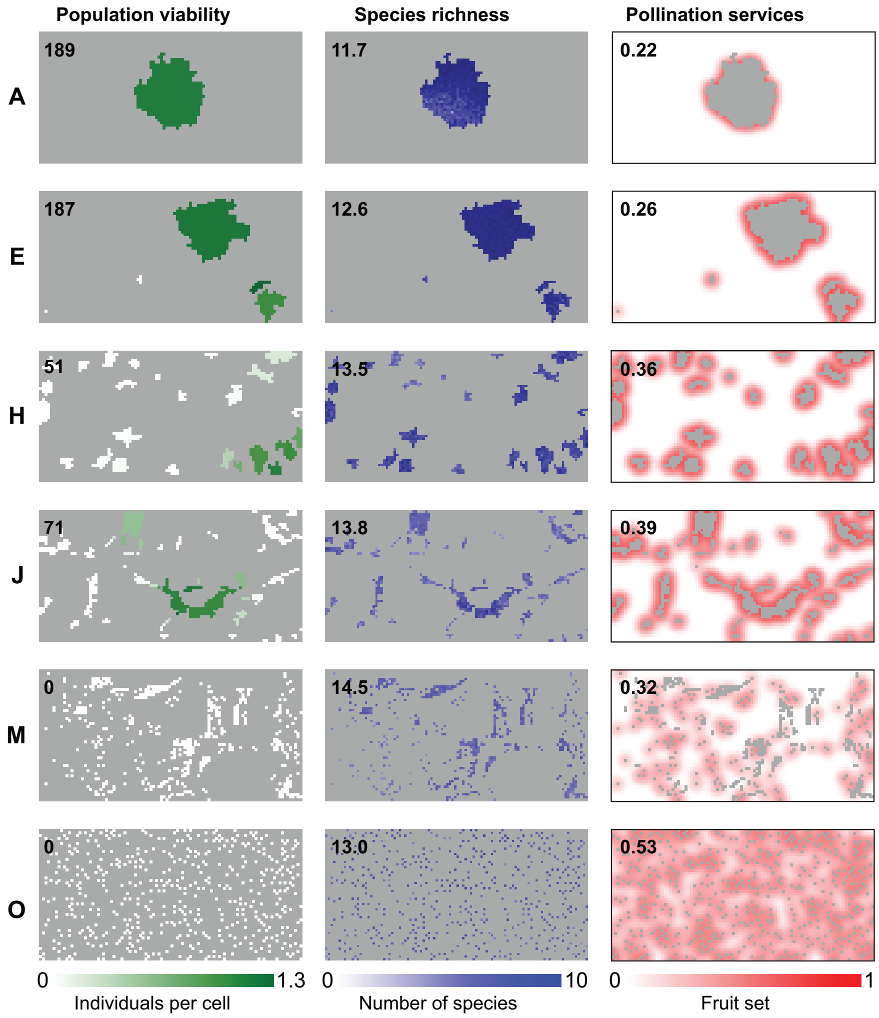

Figure 3. Maps of mammal population sizes at the 5-km scale, and bird species richness and crop fruit set due to pollination services at the 10-km scale, in six representative landscapes with 10% habitat cover, as simulated by the MetaConnect, metacommunity and InVEST models respectively. The overall scores for each landscape are shown in the top-left corners of the maps.

tion persistence and species richness) for diverse sets of organisms: plants, insects, mammals and birds. Pollination service, meanwhile, was assessed for pollinating insects as the focal organism. The simulation results thus enabled us to explore how the differing landscapes compare with respect to these goals, considering a range of different organisms.

Results

While we might hope that the best landscape pattern for one conservation goal would perform well for the others, it turns out not to be so simple. The best kinds of landscapes for conserving different aspects of biodiversity (Figure 3) are sometimes strikingly different: at the 5 km scale landscape E, with 5 patches, for example, scores highly for population viability and genetic diversity and relatively poorly for species richness and pollination service, but these ranks are reversed in the fragmented landscapes H through O. The relative values vary strongly with the spatial scale (Figure 4). At the finer, 5 km scale, if 10% of a landscape could be allocated to habitat, most criteria favoured a “blocky” pattern such as in landscapes E and F – especially for organisms that need a lot of space (e.g., birds and mammals). Indeed, with only 2% habitat cover, the best solution was often to have it all in a single large patch (landscape P or Q). At the coarser, 50 km scale, more-fragmented arrangements (e.g. landscapes L and N) performed better (Figures 3, 4), especially among the scenarios with 10% habitat cover. When only 2% of the land could be protected at this scale, the best scenarios shifted towards intermediate levels of fragmentation, as population viability and genetic diversity plummeted to near zero where habitats were too finely subdivided.

Some clear patterns emerge. The best landscapes usually had neither a single large patch nor a completely fragmented network, but something in between. Natural plant and animal populations tend to be patchily distributed in space – as they were in our metacommunity model, while neutral differentiation builds up in fragmented populations – as in

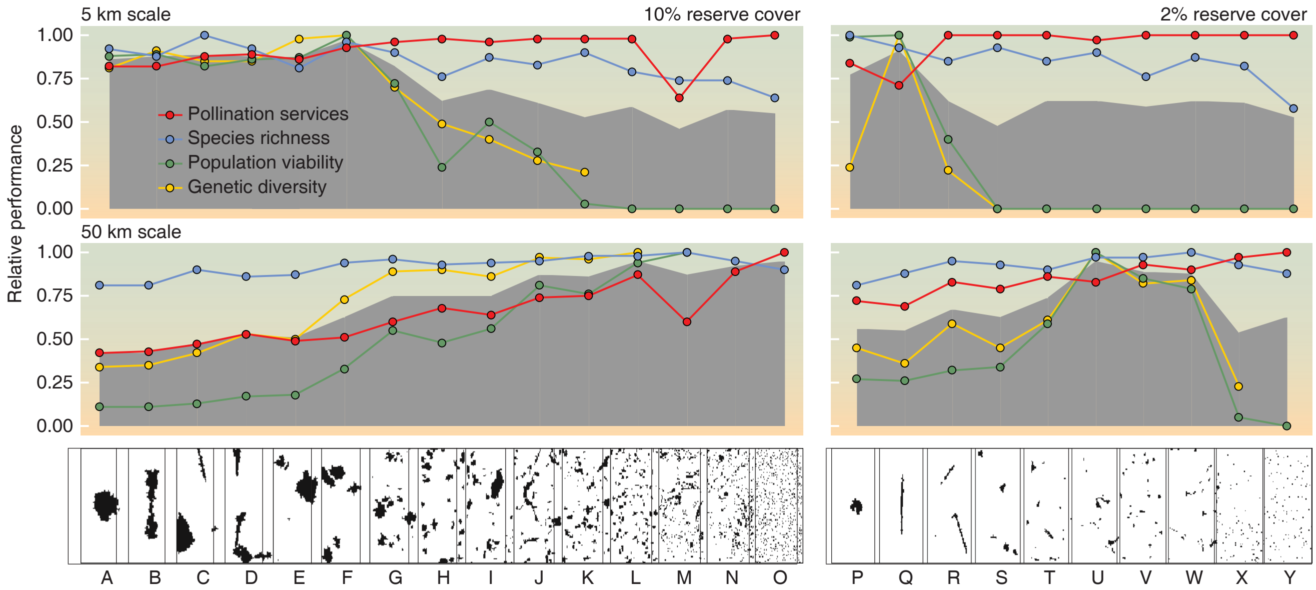

Figure 4. Relative values of four conservation criteria for constrasting scenarios, as assessed by simulation models for landscapes spanning 5km (top) and 50km (bottom). Results for scenarios with 10% reserve cover are shown on the left, and for those with 2% coverage on the right. Each of the four panels shows performances relative to the best scenario for each criterion. The grey shading indicates the average value among scores from the four criteria. Values for genetics, population viability and species richness are for mammals (5-km scale) and songbirds (50-km scale).

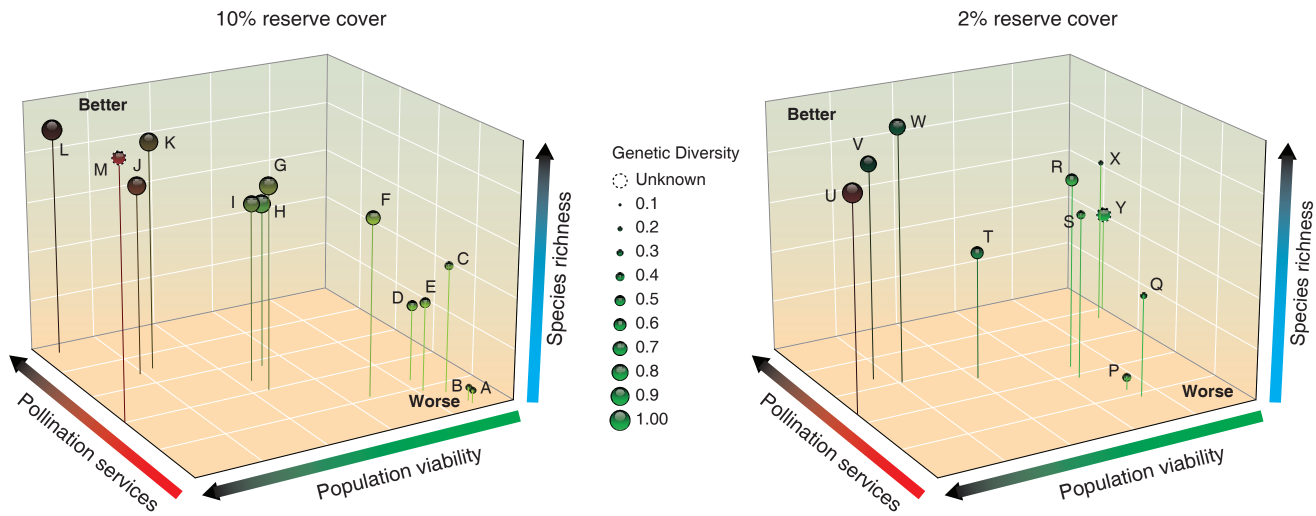

Figure 5. Scatterplots comparing landscapes against all four criteria simultaneously, for scenarios with 10% habitat cover (left) and 2% cover (right). The letters refer to the landscape patterns (see Figure 2) as implemented here at the coarser scale, spanning 50 km. The size of each circle indicates genetic diversity, the height of stalks indicates relative species richness and positions on the horizontal axis indicate metapopulation size – all for songbirds, while positions on the receding (diagonal) axis represent insect pollination service. All four variables are expressed relative to the highest values found for each level of cover.

the MetaConnect model – so that a single large patch could miss many species or locally-distinctive gene alleles. In addition, ecosystem services such as pollination are delivered most efficiently by a fragmented or branching pattern of habitat that brings biota (in this case, pollinating insects) into intimate contact with a surrounding agricultural matrix. On the other hand, very small patches may not be able to support viable populations; many birds and mammals have a territorial requirement, and at very fine scales even insects and plants may fail to persist in tiny habitat fragments.

Besides rejecting the two extremes, our results suggest some landscape configurations that would provide a good compromise among the different conservation goals. For example, the left-hand panel of Figure 5 (50-km scale, 10% coverage) shows a broadly diagonal trend in performance across multiple criteria, from generally poorly-performing scenarios in the lower front region (scenarios A-E, with few large patches) to ones that do quite well on all four axes at the upper rear of the figure (scenarios J, K and especially L), which are fairly fragmented. There is less agreement among criteria in the 2% cover scenarios (right-hand panel of Figure 5); here the moderately-fragmented patterns W and U perform fairly well for genetic diversity, population viability and species richness, but are outperformed for pollination service by more-fragmented patterns.

Conclusions and application

This exercise demonstrates how different conservation goals, applied to different sets of organisms and at different spatial scales, can advocate different conservation strategies. It was not intended to yield specific prescriptions for particular nature reserve networks, since our models use little information on habitat or species distribution. However, our results do suggest some general geometric principles for reserve design that might be validated and developed by future work.

First of all, the best strategy will depend on the amount of land that can be protected in a given area. In general, more-fragmented landscapes performed better when there was more habitat: i.e., for higher rates of cover and when patterns were interpreted at a coarser spatial scale. This reflects a law of diminishing returns: once patches are large enough to hold viable populations of most species present, additional land is better protected at separate, contrasting sites. Consequently, making a single nature reserve as large as possible is unlikely to be ideal unless a very small area of land can be protected. Indeed, it is also clear that there is no single ideal patch size. Different groups of species may be conserved by reserves of differing sizes, including some that are big enough for large animals with low population densities, alongside smaller reserves that broaden the range of environments represented, which may support a range of specialist species and of cryptic genetic diversity.

It is encouraging that, despite the contrasts, some solutions performed reasonably well for a wide range of scenarios. Moreover, some of the best-performing landscapes (e.g. landscapes J, M, U and W) were derived from actual habitat maps rather than patterns created specifically for the analysis. Real conservation planners make decisions based on a complex mix of ecological and other information, ranging from species records and habitat maps through to land availability and cost. The typical resulting mix of reserves of different sizes and shapes, located in diverse environments, may even be a reasonable approximation to the kinds of patterns that more advanced multi-criterion conservation tools, such as described here, will ultimately recommend.

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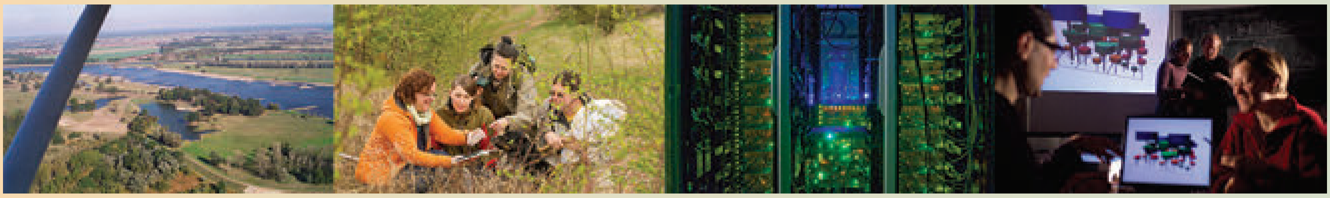
CHAPTER **IV**



Methods and tools

Determining responsibilities to prioritize conservation actions across scales

Dirk S. Schmeller, Yu-Pin Lin, Tzung-Su Ding, Reinhard Klenke, Douglas Evans, Klaus Henle

Limited resources and conservation actions

Conservation actions, such as biodiversity monitoring, wildlife disease monitoring, capacity building or the evaluation and improvement of the effectiveness of current conservation networks in protecting biodiversity, could largely benefit from an intelligible resource allocation (Schmeller et al. 2014). In the past, conservation was prioritized by using the threat status of species assessed by red lists, with the IUCN having a leading role (IUCN 2001). A complementary approach, which has been developed over recent years, determines the conservation responsibilities for an area of interest, which could be any administrative unit, biogeographic regions, or even whole continents (different methods were reviewed in Schmeller et al. 2008c). The approach captures the impact of the loss from the focal region (usually a country) on the global persistence of the assessed species or habitats and determines the international importance of the area of interest for global, regional or national conservation targets (Schmeller et al. 2008a, b). Hence, determining conservation responsibilities could help to inform conservation policy and governance from a national to a global scale, and could assist in meeting the CBD Aichi 2020 targets 5, 10 (Strategic Goal B: Reduce the direct pressures on biodiversity and promote sustainable use), and 17 (Strategic Goal E: Enhance implementation through participatory planning, knowledge management and capacity building). The approach creates hierarchical lists of national responsibility for species and habitats and decision makers could work down that list from very high to high to medium to basic responsibilities, depending on the availability of resources (Schmeller et al. 2008b). All species and habitats with very high and high national responsibilities should be closely monitored following appropriate monitoring programs. The national responsibility approach also helps to identify biodiversity data gaps and therefore could guide capacity building efforts. Without good distribution data on species and habitats, an initial assessment of impacts on those species and habitats and hence their threat status is impossible. Decision makers, i.e. the United Nations Environment Programme (UNEP) or the intergovernmental platform for biodiversity and ecosystem services (IPBES) or the Group of Earth Observations Biodiversity Observation Network (GEO BON), may therefore use the national responsibility approach to determine how much resources need to be set aside to monitor those species and habitats to rapidly complete important biodiversity data.

The national responsibility method

The method to determine national responsibilities comprises three decision steps (Figure 1). Firstly, the assessment unit is defined based on the underlying concepts and definitions chosen by the user. Secondly, the current distribution pattern of a species or habitat is determined, meaning its range within and across biogeographic and environmental regions as an approximation of its adaptability to different environmental conditions. The third step determines the importance of the distribution of the defined assessment unit within a focal area as compared to the total distribution in a reference area, determining the expected and observed distribution, allowing geographic scaling. The distribution pattern and the expected value of occurrence together reflect the importance of a focal area for the global persis-

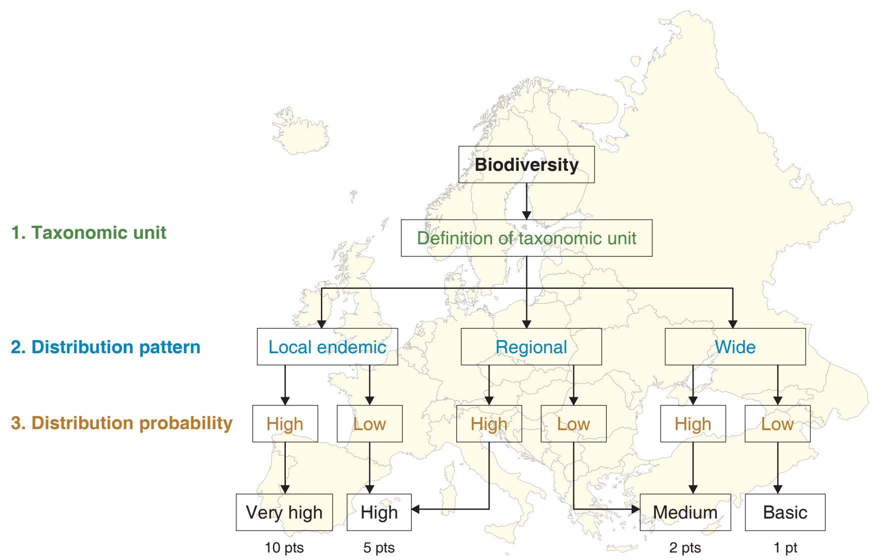


Figure 1. The three steps of the national responsibility approach from Schmeller et al. (2008b).

tence of the defined assessment unit. While the data needs of the method are not very high, several difficulties were recently described, which could hamper the application to biodiversity in general (Schmeller et al. 2014). Most of these difficulties arise from the lack of agreed data standards and harmonization during collection and processing of biodiversity data, but might be overcome quickly by different projects and initiatives building regional and global biodiversity observation networks. Therefore, the determination of conservation responsibilities should be feasible and can already be done for all species for which distribution data is available via the IUCN website. To facilitate the task, the project SCALES has developed a GIS module, for both ARC-GIS and QGIS software, to automate the determination of conservation responsibilities. The automatisation allows the determination of conservation responsibilities for all species and habitats with known distributions across the world. The IUCN Red List database (http://www.iucnredlist.org/) currently covers over 70,000 species, with a steady increase in numbers. For 43,000 species, distribution maps are currently available and for those species conservation responsibilities could be determined, including the well-assessed species groups mammals, birds, amphibians, freshwater crabs, warm-water reef building corals, sharks and rays, groupers, wrasses, lobsters, conifers and cycads.

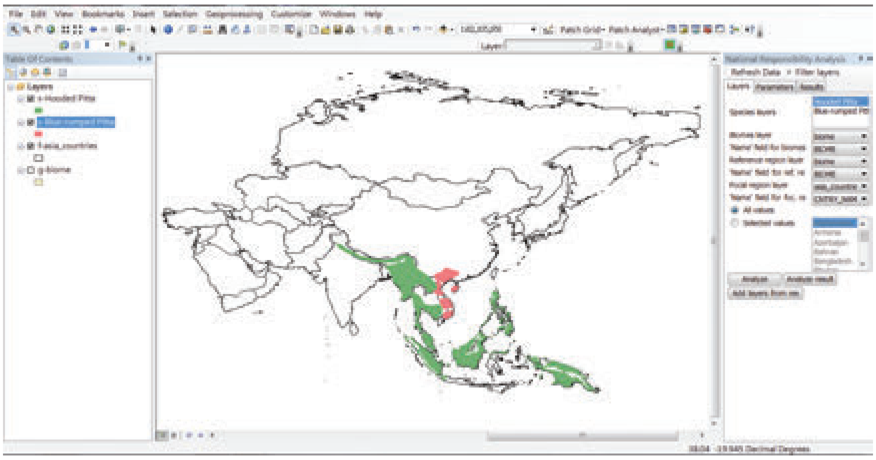


Figure 2. Interface of the National Responsibility Tool (NRT).

The GIS-tool to determine conservation responsibilities

The National Responsibility Tool (NRT, Figure 2) uses a GIS-based approach to determine the international importance of a species distribution area in a focal area following from the work by Schmeller et al. (2008a,b; 2012). The assessment is based on the bioclimatic map developed by Metzger et al. (2013). As input data, the NRT requires a map of the global distribution of the species, habitat or ecosystem, a map of the reference area, and a map of the focal area, usually country borders, in the widely used shapefile format. The NRT ranks the species according to the conservation responsibilities it calculates and allows the results to be displayed as vector maps with a table of the results on a GIS platform, which can either be ARC-GIS (ESRI) or QGIS (open source). The NRT can also combine the conservation responsibility rank with the IUCN Red List status, as suggested by Schmeller et al. (2008a). These complementary assessments would allow determining the conservation priorities of species for nations or other focal areas. Both, conservation responsibilities and priorities can then be displayed in informative vector maps and tabular data, readily usable to inform policy and decision makers in different regions or continents (for an example see Figure 3).

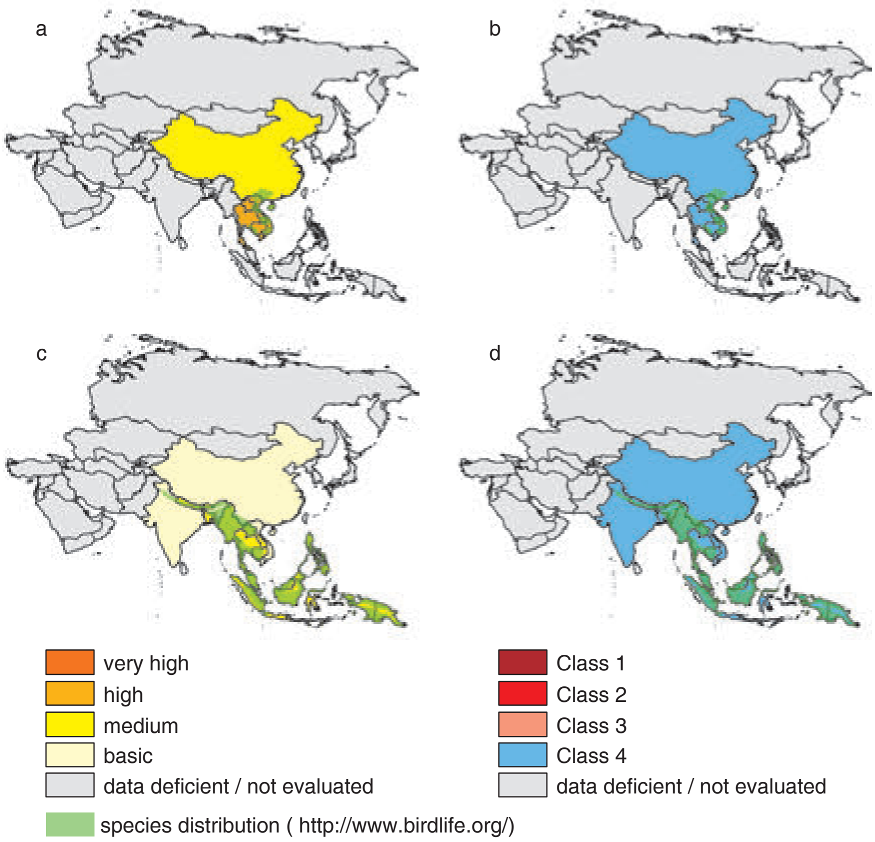


Figure 3. Examples of the output of the National Responsibility Tool for two Asian bird species.
(A) National Responsibility and (B) Conservation Priority for the Fairy Pitta (*Pitta brachyura*), (C) National Responsibility and (D) Conservation Priority for the Hooded Pitta (*Pitta sordida*).

Concluding remarks

The determination of conservation responsibilities could enhance resource allocation to biodiversity conservation and capacity building in regions of urgent need. It also would allow the assessment of efficiency of current reserve site networks, conservation needs in light of emerging pathogens and diseases, or monitoring the impact of genetically modified organisms (Schmeller et al. 2014). With the NRT at hand, regular assessments, usually after updates of currently available biodiversity data, become possible and provide an important information source of global environmental programs, conservation conventions and conservation NGOs.

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A GIS-based spatiotemporal modeling with Bayesian maximum entropy method

Hwa-Lung Yu, Shang-Chen Ku, Alexander Kolovos

Spatiotemporal mapping techniques

Spatiotemporal modeling techniques are often developed independently of GIS, resulting in a rather limited and loose coupling of corresponding tools with GIS environments (Goodchild and Haining 2004). To the end of spatiotemporal interpolation methods, a variety of methods have been proposed that adopt different weighting schemes to elucidate the relationship between the observations and unmonitored space-time locations, e.g., inverse distance weighing, kernel smoothing, and geostatistics. Among them, geostatistical methods account for stochastic dependence among the dataset and can generally provide a better interpolation than non-stochastic methods. A number of software tools exist that provide solutions based on well-known geostatistical methodologies in the literature, such as the family of kriging methods (Chiles and Delfiner 1999). An alternative to the previous mainstream methodologies is the Bayesian maximum entropy method (BME) (Christakos 1990, Christakos 2000); BME is an expansion of traditional geostatistical approaches that operates in a knowledge synthesis framework. This study makes a decisive step to bridge these gaps by proposing a new GIS-based module to be used as a tool for spatiotemporal analysis and prediction.

Spatiotemporal modeling with the Bayesian maximum entropy approach

A knowledge synthesis (KS) framework for spatiotemporal modeling was proposed by (Christakos 2000). It distinguishes two categories of knowledge bases in space-time attributes. One component is included to reflect our level of general knowledge regarding main attribute characteristics and scientific facts that apply in the analysis context. A second component involves the attribute observations and data measurements at specific instances in space and time. This distinction can be considered as a philosophical basis to integrate the subjective and objective information for the modeling of complex space-time processes. In order to formulate the proposed KS framework, BME method was developed to integrate two well-known epistemic approaches, i.e., maximum entropy method and Bayesian method, and to characterize the variations of space-time attributes. The KS concept for spatiotemporal mapping by BME method is illustrated in Figure 1. Based upon the KS framework, for the case of ecological modeling one can obtain space-time species distribution from integrating 1) general knowledge, e.g., empirical laws and ecological theory, and 2) specific knowledge, e.g., species observations or land-use observations in terms of either hard or probabilistic forms. The major distinction of BME approach with standard Bayesian method is the use of maximum entropy method for characterizing the prior knowledge probability function of the study. For a more detailed theoretical presentation and equations of BME and its associated KS framework, readers are referred to (Christakos 2000) and references therein. In the following, we present the STAR-BME software module that brings to GIS the advanced features of spatiotemporal BME prediction as a dedicated plugin component in Quantum GIS.

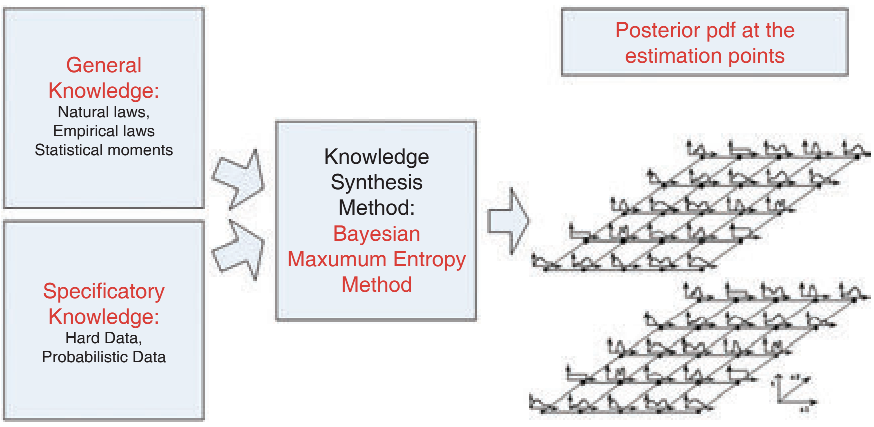


Figure 1. Spatiotemporal modeling flowchart by BME method.

Development of STAR-BME

STAR-BME (an acronym for Space-Time Analysis Rendering with BME) is geared by the BME theory to perform spatiotemporal modeling and analysis, and the STAR-BME development aims at a tight coupling of this advanced functionality to GIS environments. In a different light, STAR-BME provides a GIS-centric tool for the modeling of space-time data. This integration enables GIS software to evolve into a more comprehensive knowledge processing and dissemination platform, as discussed before, by means of the following elements: (i) With STAR-BME, a dedicated GIS tool is now available to process not only spatial data but space-time data, too; (ii) STAR-BME extends the functionality of existing geostatistical GIS tools with unique analytical features powered by the BME theory. Furthermore, the incorporation of STAR-BME into GIS environment can facilitate seamless interaction between spatiotemporal analysis tools and other GIS components; in turn, this can empower GIS users with more tools to access and analyze space-time data. At the present stage, the STAR-BME toolbox has the following major features:

1) Flexible, practical display of space-time data in the GIS environment
2) Integration of multi-sourced space-time data in different formats
3) Incorporation (assimilation) and display of space-time data with multi-sourced uncertainties in various probabilistic forms
4) Analysis of space-time dependence by using empirical and mathematical spatiotemporal models
5) Prediction and validation in space and time
6) Data export and display in multiple formats

STAR-BME is currently available as a plugin for the Quantum GIS (QGIS) open source software (QGIS Development Team 2012), which is freely available at http://homepage.ntu.edu.tw/~hlyu/software/STAR.

An illustrative example

We applied the STAR-BME toolbox in the spatiotemporal estimation of an environmental attribute of concern, e.g. particulate matter in this case. The data were observed at the locations shown in Figure 2. In addition, some secondary data were considered for the purposes of mitigating the biased estimation due to the unbalance distribution of spatial sampling of the dataset. The secondary data were present in the probabilistic form as shown in Figure 3. For detailed descriptions of this dataset, we refer to (Yu and Wang 2013). STAR-BME provides functionality to investigate the temporal distribution and distributional property at a user-specified location as shown in Figure 4, as well as the spatiotemporal trend and covariance estimation of the dataset. Spatiotemporal prediction can be performed at user-specified individual points, grid nodes, or ESRI™ shape files, e.g., polygons. As shown in Figure 5, the spatiotemporal predictions can be presented in either vector or raster formats. Among them, the vector presentation enables

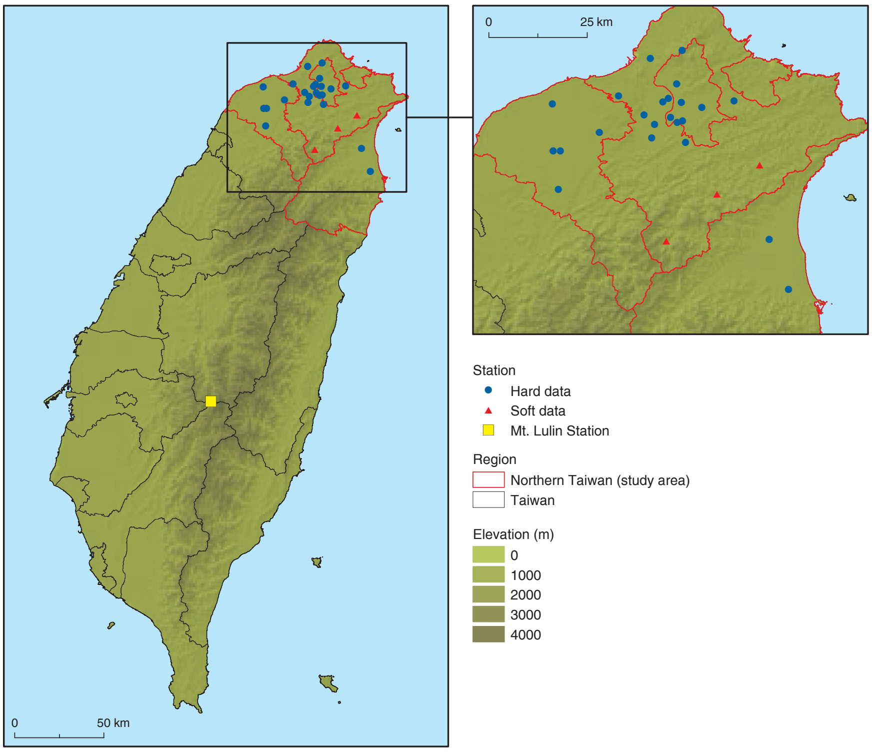


Figure 2. Topographical map of Taiwan with PM10 monitoring stations, i.e. hard data locations, Mt. Lulin station, and the monthly-based PM10 soft data locations.

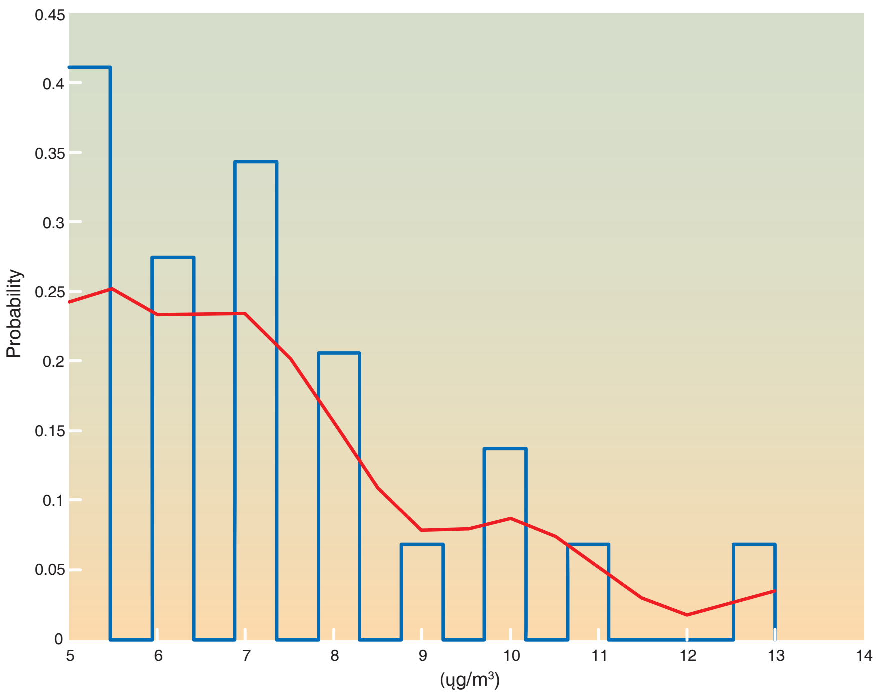
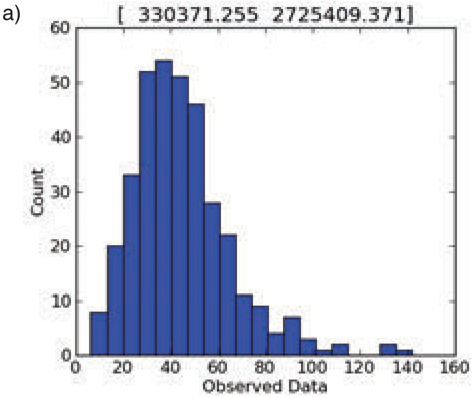


Figure 3. Probabilistic form of the uncertain PM10 daily observations in February at soft data locations.



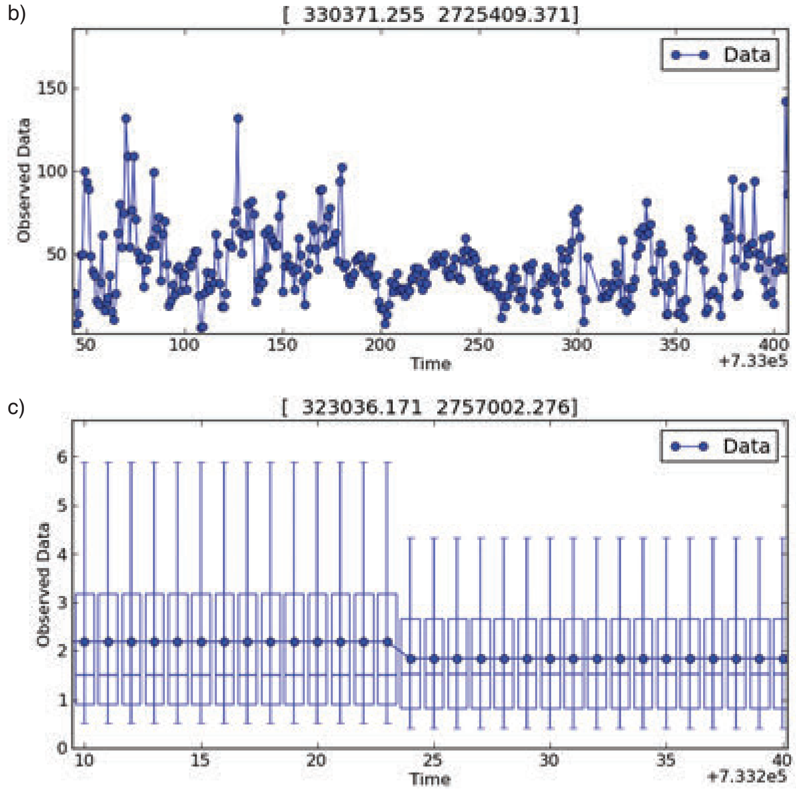


Figure 4. Data views for the space-time data during 1/1/2007-12/31/2007: (a) Histogram view at Dongshan station, (b) Time series view of hard data at Dongshan station, and (c) Time series view of soft data at the spatial coordinate [323036.171, 2757002.276] during 6/17/2007-7/17/2007, where the boxplot is used to depict data quantiles at [5, 25, 50, 75, 95], and the point presents the mean of the uncertain observations.

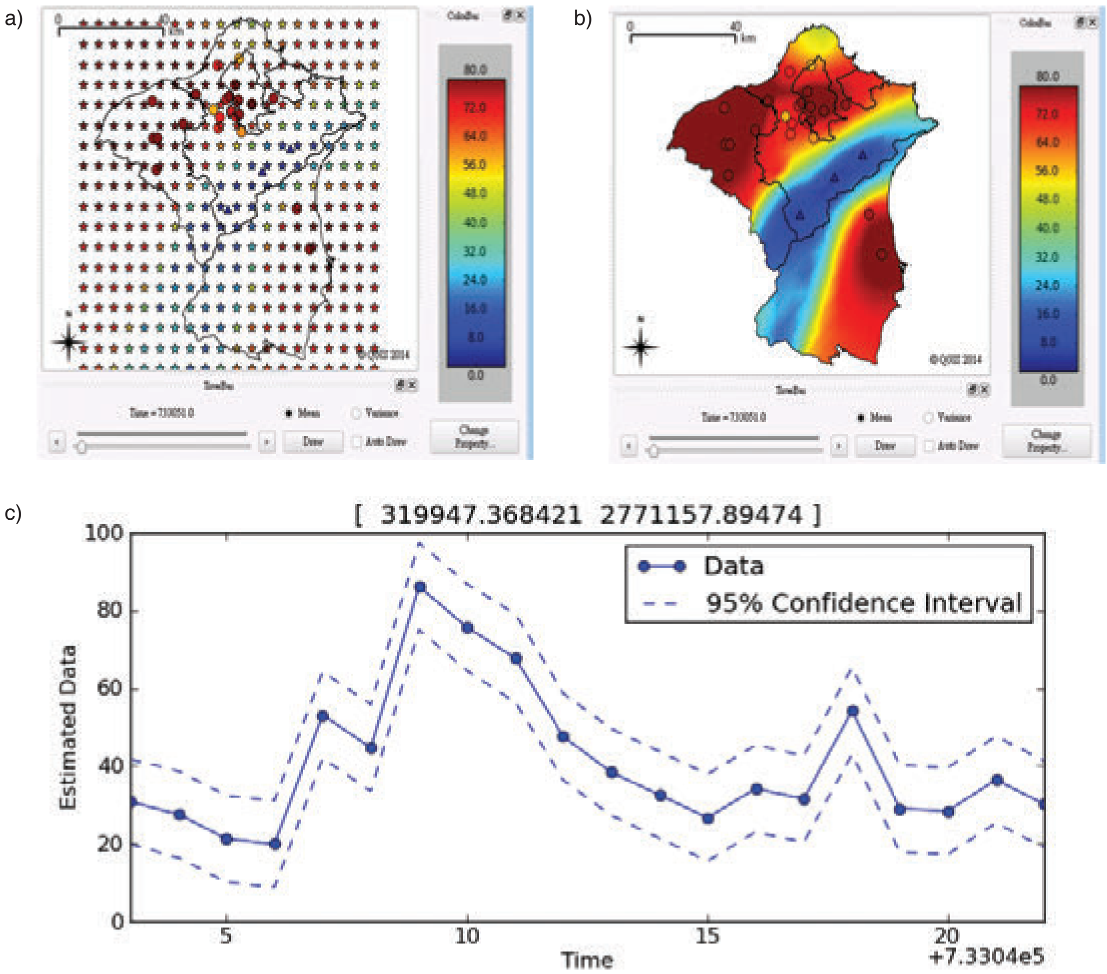


Figure 5. BME PM10 space-time predictions of the study area on 1/9/2007 in (a) vector, and (b) raster formats. (c) The time series of mean and 95% confidence interval of prediction results at a user-selected spatial location from the vector output during 1/1/2007-1/20/2007, i.e., t=[733043,733062].

users to inspect the time series of predicted results and their associated prediction error at a user-selected spatial location. In addition to vector and raster files, prediction results can be also stored in plain ASCII text format so that they can be of further use. Figure 6 shows the seamless integration between spatiotemporal prediction and Web Map Service (WMS) functions under GIS platform with STAR-BME toolbox. It provides an opportunity to interactively inspect the relationships between the spatiotemporal distributions of attributes of interest and other associated spatial information, e.g., land use patterns.

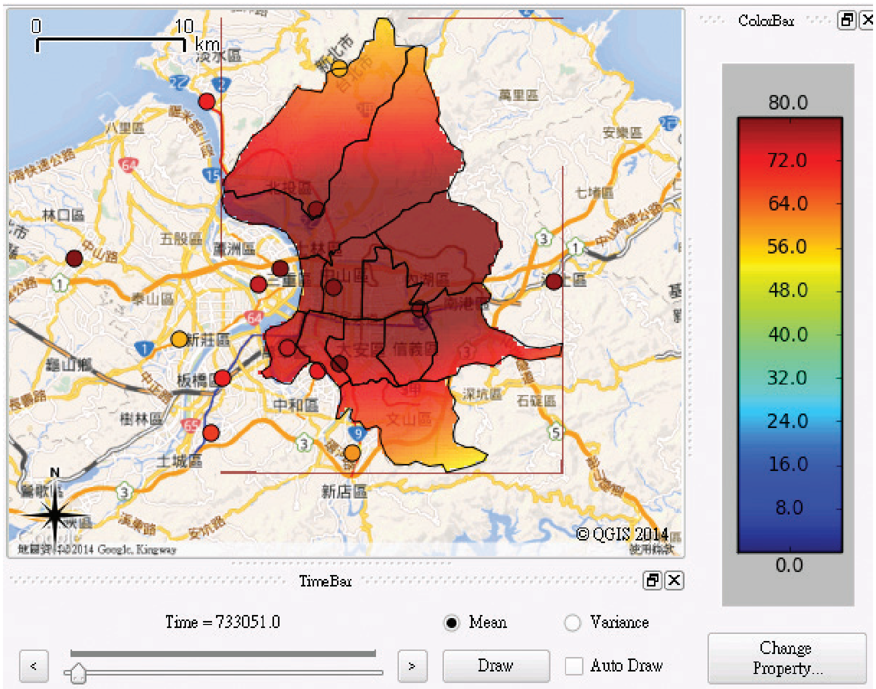


Figure 6. The raster output of BME PM10 prediction means within the focus area of Taipei city overlaying WMS maps at a selected temporal instance, i.e., 1/9/2007 (t=733051).

Concluding remark

This paper introduced STAR-BME, a QGIS toolbox that implements advanced spatiotemporal analysis and mapping functions in a geostatistical context. The analytical foundation of STAR-BME is based on the knowledge synthesis framework and the BME theory for space-time modeling, which, among other features, enable the toolbox to uniquely account for data uncertainties that are very common in studies of natural attributes. STAR-BME also has a variety of mapping features that help it extend the QGIS capacities for multi-perspective space-time data visualization.

In light of these considerations, we have endeavored to provide an easy-to-use analytical GIS component that enriches the present GIS software with sophisticated functionality for space-time analysis. Building on its space-time analysis-oriented character, STAR-BME could help establish the role of GIS software as a comprehensive knowledge processing and dissemination platform, not only for spatial, but for space-time data as well.

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Downscaling climate data to predict species' ranges

Richard M. Gunton, Veiko Lehsten, William E. Kunin

Why study microclimates?

Any species of plant or animal will be adapted to live at certain ambient temperatures, and one reason why climate change matters is simply that individuals will die, and local populations will eventually die out, when ambient temperatures become too unsuitable. Most plants are rooted in the soil, and many terrestrial invertebrates and other animals spend at least part of their life-cycles on or below the soil surface, so soil temperatures are an important feature of their habitat. Soil temperatures may vary widely from one point to another in a landscape because of varying elevation, vegetation cover and exposure to the Sun's rays (Figure 1). This variation will affect the distribution of suitable habitat for particular species at quite fine spatial scales – even just a few metres. Microclimate means the local conditions that affect the distribution of organisms, and soil temperature is an important component of this for many species. The climate warming experienced in Europe and over much of the Globe is normally described as changes in average air temperatures over coarse spatial scales. It will undoubtedly affect local soil temperatures, but the actual changes in microclimate may be complex and difficult to predict. That is the challenge addressed in this chapter.

Downscaling temperatures

Existing global weather models can indicate current and predicted average air temperatures, solar radiation and other climatic factors at coarse resolutions (e.g. 0.5-degree grid cells, corresponding to hundreds of square kilometres; Figure 2). Such scales may be quite appropriate; air temperatures, for example, can be quite uniform over large distances as winds distribute heat over broad areas – although there is still a statistical challenge when comparing alternative climate or weather forecasts with local measurements to decide which model is most realistic. Ground temperatures, however, do not simply match air temperatures, since the ground not only exchanges heat with the air above it but can also be directly warmed by the sun as radiation strikes its surface. It is also well known that the ground radiates heat into the atmosphere according to its own temperature, and that air temperatures themselves tend to decrease with increasing elevation. These processes may depend substantially upon terrain and vegetation cover. Now, because terrain and vegetation maps are available for the Earth's land surface at very fine resolutions, it should be possible to use such maps to create a method for converting coarse-scale climatic predictions into fine-scale ground temperature predictions – what we call "downscaling". Here we describe the development of a tool for doing this across European landscapes, principally by downscaling predicted solar radiation values.

Our downscaling method uses statistical fitting based on preliminary understanding of how the ground is heated and cooled. It improves on purely statistical methods, which may be unreliable under climate change. An alternative approach would be to use a fully-mechanistic model with interrelated equations modelling all relevant quantities, such as soil moisture contents and heat capacities. For example, our input climate data come from a climate reanalysis project, where a climate simulation model (such as that used for weather forecasting) is run

Figure 1. Differential frost cover on uneven ground in sunny conditions, photographed around 10:00 on 6 March 2012 in Leeds, U.K. (photo: Richard Gunton).

for the past and corrected at each time step using observed values, reducing the sensitivity of outputs to erroneous parameter values. Such an approach could be applied at fine scales, but this would require very complex models and high computational effort. Thus we believe our approach provides a compromise between computational effort and accuracy.

Obtaining a predictive formula

Physical laws can give some idea of how soil temperatures should be related to solar radiation, average air temperatures, elevation and vegetation cover, but the system is far too complex for us to attempt microclimate predictions from first principles. Instead, we combined basic geophysical insights with empirical data in statistical models. A year's soil temperature data were obtained from 83 sites in Finland, Poland, the U.K., France and Greece (Figure 3). At each site a tiny (1-cm diameter) temperature sensor was buried at a depth of 2 cm in the soil either in open grassland or under tree cover, and the slope, aspect and elevation of the site were noted. The resulting temperature data were summarised in a variety of ways (calculating monthly means and maxima, or extracting daily afternoon temperatures). Publicly-available hourly data for estimated solar radiation over the same periods at a coarse scale of ½° latitude × ⅔° longitude (Global Modeling and Assimilation Office 2012) were then combined with data on the slope and aspect of each site to obtain down-scaled estimates of solar radiation incident on the site over relevant time periods. We could then fit statistical regression models relating the soil-temperature data to the downscaled solar radiation, along with coarse-scale data for air temperature and wind-speed (Global Modeling and Assimilation Office 2012), plus other basic site data, to derive formulas for predicting monthly mean and maximum temperatures, and daily noon temperatures, in either open-grassland or tree-covered habitats. Because our sites spanned a range of locations across Europe, including elevations from 43 to 1500 m above sea level and slopes from 0 to 45°, our models should be suitable for making predictions for near-surface soil temperatures in a wide range of open and tree-covered landscapes across the continent.

We demonstrate the application of these formulas by generating maps (Figure 4) of estimated monthly mean ground temperatures in a range of landscapes around Europe at the time of the data collection (2009–2010). Similar maps could be produced for future climate scenarios.

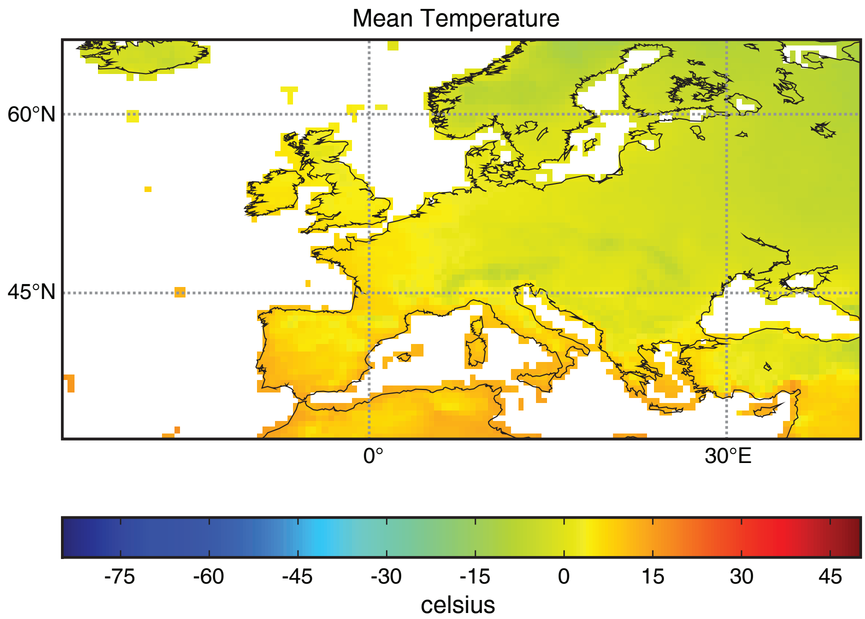

Figure 2. Mean recorded land-surface temperatures over the last decade of the 20th Century in 0.5-degree grid cells of Europe. Climate Research Unit Climatology data (New et al. 1999), obtained from www.ipcc-data.org, August 2014.

Our findings

An example of one of the formulas we obtained is shown in Box 1.

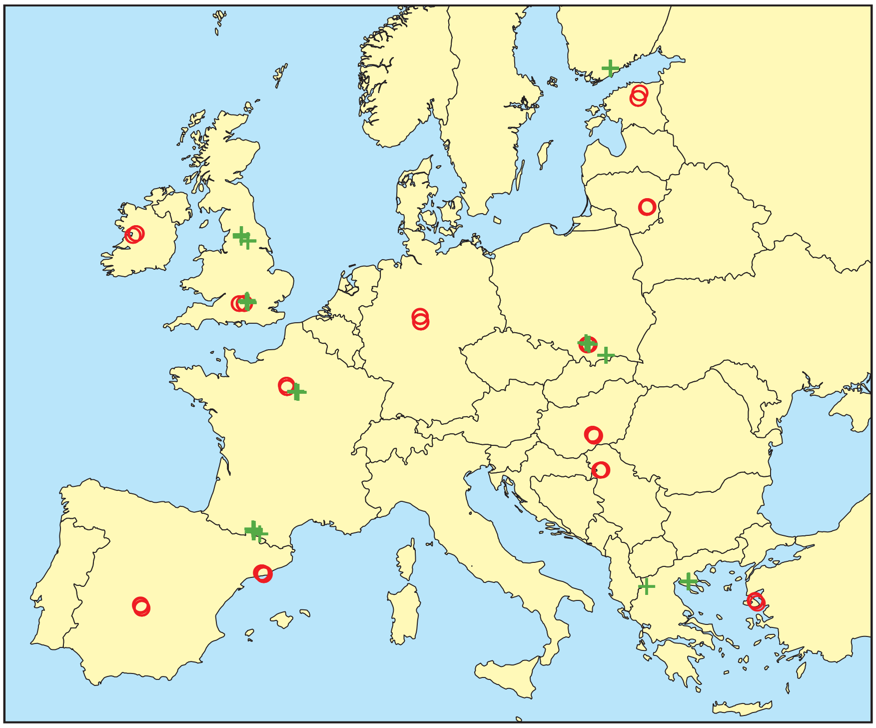

Figure 3. Locations of temperature sensors yielding data used to fit the microclimate models (green crosses) and of those yielding data used to test the models (red circles).

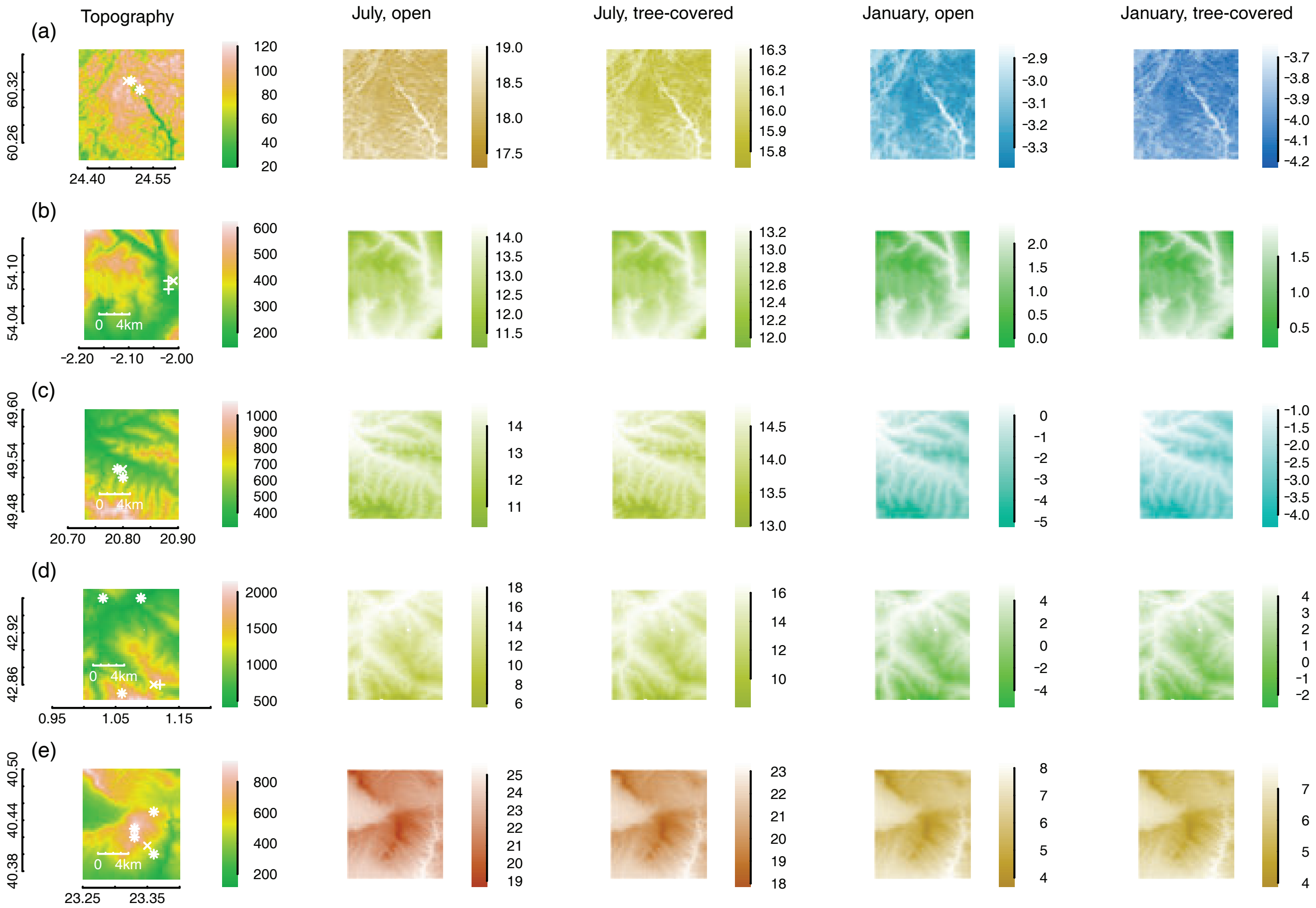

Figure 4. Maps of 5 areas containing microsites where temperature sensors were located, with predicted mean temperatures for the whole landscape. Maps in the first column show topography, with axes indicating longitude and latitude, and crosses at the locations where temperature sensors were buried ("×" for open sites and "+" for tree-covered sites). The following maps within each row show the same areas with predicted monthly mean temperatures, in degrees Celsius. Predictions are made for July 2010, for an open landscape (second column) and a tree-covered landscape (third column), and for January 2010, for an open landscape (fourth column) and a tree-covered landscape (fifth column). The rows cover sites in (a) southern Finland, (b) northern England, (c) southern Poland, (d) southern France and (e) Greece. The resolution of these maps is about 30 m.

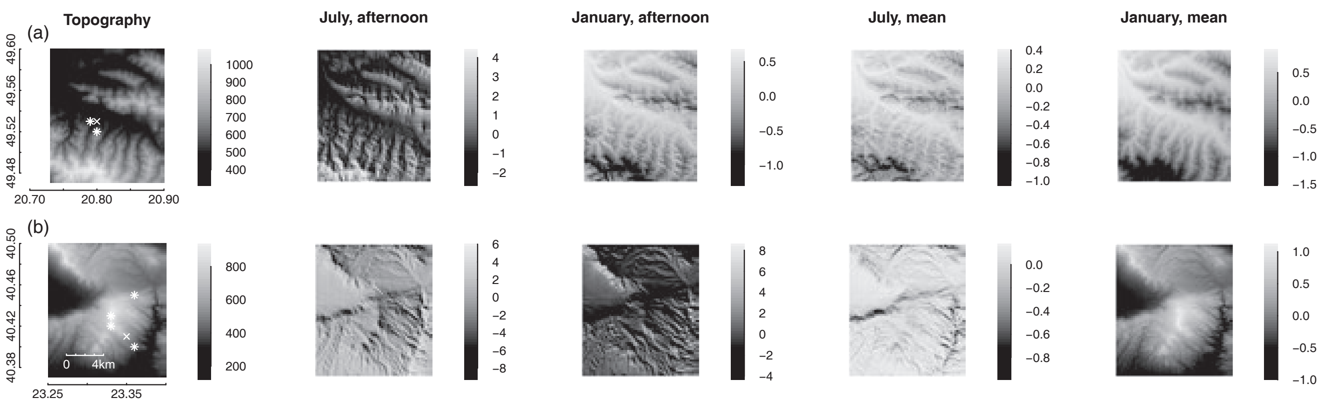

Figure 5. Effects of solar radiation on predicted ground temperatures in open landscapes. Maps in the first column show topography (elevation in m; axes indicating longitude and latitude). The following maps show the same areas with predicted effects of radiation on temperatures (in degrees Celsius) at 13:30 on a sunny day in July 2010 (second column) and January 2010 (third column), and effects on mean temperatures in mid-July 2010 (fourth column) and mid-January 2010 (fifth column). The first row of maps shows a site in Poland (Compare Figure 4c) and the second row, a site in Greece (compare Figure 4e).

The combination of local terrain data with coarse-scale sunshine data enabled effective downscaling of monthly mean, monthly maximum and daily noon temperatures at open sites (Figure 4). Thus, climate data from future climate scenarios could be used to generate fine-scale predictions of which areas will be suitable for species of open habitats whose ranges are limited by mean or maximum soil surface temperatures. Such predictions could be used to model species' ranges and habitat connectivity, which can in turn be used for predicting the

Box 1. A formula for monthly mean temperatures at open grassland sites

The near-surface soil temperature at open grassland sites, in degrees Celsius, is predicted by the formula below. This formula was used to create the maps in the second and fourth columns of Figure 4.

$$11.5 + 0.00014 \times [\text{Radiation} - 97] + 0.18 \times [\text{Temperature} - 11] + 0.31 \times [\text{Convection} - 1.3] - 6.0 \times [\text{Elevation} - 0.41] - 6.8 \times [\sqrt{\text{Latitude}} - 7] - 0.0070 \times [\sqrt{\text{Coast}} - 12] + 0.099 \times [\text{Temperature} - 11] \times [\text{Elevation} - 0.41]$$

where:

- Radiation is the solar radiation incident on the ground surface, accounting for cloud cover and topography and averaged across a month (including night time), in Watts per square metre;
- Temperature is the monthly mean air temperature at 2 m above the ground, in degrees Celsius;
- Convection is the result of multiplying the square root of wind speed (in metres per second, at 2 m above the ground) by the difference between current and annual mean air temperature in degrees Celsius – so it is negative in cold periods and positive in warm periods;
- Elevation is the height above sea level, in kilometres;
- Latitude is in degrees; and
- Coast is the distance from the nearest sea, in kilometres.

Full details of this and the other models are provided in a forthcoming paper, Gunton et al. (in prep).

To make predictions for a given time and place, estimates for the six variables outlined above need to be obtained. For most of these, it will be noted, specific values are to be subtracted to obtain the required model input. These specific values are averages for the sites used in fitting the model, which means that setting any of the model inputs to zero is equivalent to assuming the corresponding variable takes its average value. In order to find the effect of topography on its own, one could set all variables to zero apart fro11m Radiation, and leave out the initial constant (11.5); the resulting predictions would be deviations due to the amount of sunshine falling on a particular slope. The Radiation variable is the most complex for the model-user to calculate, since it must account for the angle of incidence of solar radiation on the ground. Tools for implementing this and the other models are available at www.microclim.org.uk.

potential for migration and changes in biodiversity. In contrast, temperatures at tree-covered sites were poorly predicted, which is not surprising since sunshine should have less of a warming effect on shaded ground.

We tested the accuracy of our method with data collected from different European locations in a different time period (2007, Figure 3). Predictions for these sites confirmed that monthly means could be predicted quite accurately (explaining 86% of variation in the data), whereas daily afternoon temperatures were predicted with less accuracy (explaining 72% of variation).

Outlook

We found that the variation in mean July near-surface soil temperatures can be as much as 15°C across a few kilometres in an open hilly landscape in southern Europe, while January temperatures in such landscapes can span 12°C (Figure 4). This is partly due to variation in elevation, but we predicted the effect of terrain, affecting intercepted solar radiation, to exceed 2.5°C in some landscapes in January (Figure 5). This means that microclimate variation between contrasting slopes within a single landscape can be comparable to the average latitudinal change expected over hundreds of kilometres, or the elevational change expected over a 5000 m ascent. In practical terms, this suggests that a species threatened by warming temperatures in undulating terrain may be more likely to survive by colonising more shaded slopes rather than migrating northwards or upwards. Moreover, any populations that are already restricted to shaded slopes by a requirement for cooler temperatures will be especially threatened as temperatures continue to rise, since the nearest suitable "temperature niches" are likely to be very sparse and/or remote.

We conclude that microclimate downscaling is an important component in the modelling of species' ranges. Coarsely-averaged temperatures will not be good predictors of species' distributions when a plant or animal may track its temperature niche by exploiting the microclimates of particular slopes and elevations in a rugged landscape, and this has serious implications for the conservation of such species.

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Connectivity: Beyond corridors

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Introduction

What is the first thing that comes to mind when reading the word 'connectivity'? Is it perhaps 'corridors'? For some people the answer might be 'yes', because corridors can easily attract one's attention, especially in the innovative form of 'ecoducts' or 'eco-bridges' (Figure 1). However, there has been a running debate for several decades regarding the cost effectiveness of corridors, asking what would be their best design and management, and in which circumstances do they really maximise the ecological benefits for species (Simberloff and Cox 1987, Beier and Noss 1998). To understand the debate, one must acknowledge that connectivity encompasses a much broader range of elements than just corridors. In this chapter we examine the concepts of 'structural' connectivity, 'landscape' connectivity and 'functional' connectivity, explore different measures of these, and discuss the usefulness of such concepts and measures when scaling up from small to large areas, and from single to multiple species. We describe some important challenges and suggest guidelines for researchers and practitioners to help enhance consideration of connectivity in conservation policy and management.

Differences between structural and functional connectivity

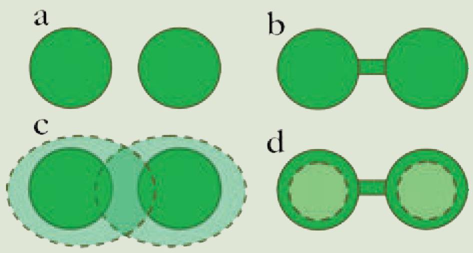
The movement of individuals across landscapes can affect many ecological processes across scales, from individual survival, through the viability of populations and metapopulations, to community dynamics, the resilience of ecosystems, and wider biodiversity (Jeltsch et al. 2013). Species distributions and their shifts (e.g., in response to climate change), depend on species' movement capacities and yet are mediated by landscape structure. The loss of connectivity, due mainly to the unprecedented expansion of anthropogenic infrastructure, is an increasingly central driver of the global biodiversity crisis. Yet to enable policy and management to maintain or enhance connectivity, consensus is needed on what connectivity means and how to measure it in an appropriate way.

One way to look at a landscape is by examining 'structural' connectivity, namely, looking at landscape structures regardless of any biological or behavioural attributes of organisms interacting with them (Tischendorf and Fahrig 2000, Kindlmann and Burel 2008). Alternatively, Taylor et al. (1993) introduced the term 'landscape connectivity' to define "the degree to which the landscape facilitates or impedes movement between resource patches". A later derivation of this term is the concept of 'functional connectivity', which focuses on the landscape from the perspective of the species, namely, the outcome of interactions between individuals and landscape structures in accordance to their needs, perception, and response norms of species or individuals (Box 1). This term, which strongly adheres to the Movement Ecology paradigm in focusing on individuals and

Figure 1. An example of a wildlife bridge, or 'ecoduct', in Banff National Park, Canada. (photo: Adam T. Ford, WTI-MSU).

Box 1.

To exemplify the concept of 'functional' connectivity, imagine two disconnected habitat patches (a), or alternatively two patches that are structurally connected by a corridor (b). A species that can move beyond the boundaries of patches and into the non-suitable environment might perceive the disconnected neighbouring patches as functionally connected (as marked by the dashed, pale green area (c). Yet a core-habitat species, which avoids habitat edges, may not move into the corridor, and hence structurally connected patches may remain functionally disconnected (d).



their response to their environment (Nathan et al. 2008), has become dominant in landscape ecology.

How to assess connectivity

In the following, we briefly present a non-comprehensive selection of the many measures and tools used to assess structural and functional connectivity. For extensive reviews, we recommend Kindlemann and Burel (2008) and Uuemaa et al. (2009), and for scale-specific aspects Simova and Gdulova (2012) and Schindler et al. (2013).

'Structural' connectivity indices are often used for large-scale assessments of connectivity, such as an evaluation of fragmentation across Europe by the European Environmental Agency (EEA). Examples of such indices are the number of patches in a given landscape or the distribution of patch sizes. Other commonly used metrics are Landscape Shape Index and patch cohesion (Schumaker 1996), both of which are based on the ratio between patch perimeter and area. Hundreds of indices can be calculated using the freeware *Fragstats* (www.umass.edu/landeco/research/fragstats/fragstats.html), GIS, and other tools (e.g. www.geo.sbg.ac.at/larg). Since many of these indices correlate strongly with each other, a small subset may suffice to cover the specific needs of the user (Calabrese and Fagan 2004).

Moving beyond a simple description of landscape structures, graph theoretical approaches enable consideration of the movement capacities of species and the landscape's resistance to such movements. The basic principle is to treat the landscape as a network of habitat patches ('nodes'), with certain movement probabilities between them ('links'). These probabilities can then be derived as a function of Euclidian distance, cost distance or least-cost paths, or alternatively, by simulating movements as an electric current in a resistance network, building on Electric Circuit theory (McRae et al. 2008). Connectivity measures derived by graph-theory approaches enable, for instance, ranking the contribution of different patches to connectivity by testing the effect of their removal. Notable applications of this approach are in projects aiming to identify optimal locations for corridor protection or restoration. For useful programs, see *Conefor Sensinode* (www.conefor.org/), *Circuitscape* (www.circuitscape.org) or GIS-based tools at www.conservationcorridor.org and www.corridordesign.org.

To consider the complexity of 'functional' connectivity as the outcome of individual-landscape interactions, one can use individual-based simulation models (IBMs). By simulating the response of individuals to landscape structures, one can consider their internal state, perceptual range, or social interactions leading to density-dependent emigration and immigration. One can simulate movements as random or correlated-random walks, but more sophisticated algorithms also consider habitat suitability and perception. As an outcome, one can calculate immigration and emigration rates within and among patches, or assess connectivity across entire landscapes. An example of such IBMs, *FunCon* concentrates on assessing functional connectivity as an outcome of either home-range (everyday) movements or dispersal (namely, movements result in depositing offspring elsewhere) (Pe'er et al. 2011). Other interesting applications of IBMs to evaluate connectivity involve efforts for species protection or reintroduction, e.g., for carnivores like bears or lynx in Europe.

While many approaches and tools have been developed and applied to assess connectivity, it is less clear which level of detail is appropriate for which question, landscape, or species, and which constraints result from data availability. Thus, a major challenge for connectivity research is to assess the array of available methods, and identify the appropriate methods and metrics for a given objective or application.

Scaling up from small to large spatial scales

Functional connectivity is ultimately an up-scaling process by itself. The interaction of individuals with their environment, at the local scale, affects processes and patterns at much larger scales (Jeltsch et al. 2013). The question is, however, how does the detailed knowledge obtained so far from movement studies and available tools, contribute to our understanding of connectivity at large scales? Such questions become particularly relevant in the context of Green and Blue Infrastructure, or other initiatives seeking to maintain connectivity across entire continents (Figure 2).

Figure 2. Established in 2003, the European Green Belt initiative connects 24 countries alongside national and regional initiatives. It is a backbone of a Pan-European ecological network and renders a significant contribution to European 'Green Infrastructure'. It builds on the former Iron Curtain, which once separated human societies, but now has the potential to connect ecological communities by forming a viable ecological network.
Source: © European Green Belt Initiative/ Coordination Group

For spatial up-scaling, the following are important considerations:

1. In working with maps covering larger spatial extent, one usually needs to present data at a coarser grain. Yet this comes with a risk of losing important landscape features and processes (Bocedi et al. 2012). Extreme caution is therefore required in selecting a procedure for scaling up, considering how the resulting maps and other outputs should be interpreted.

2. The most cost-effective measures to enhance connectivity differ across scales. On the local scale, the protection of a given species in a certain region may require ensuring the presence of corridors or stepping stones to enhance its short- and intermediate-term viability. Yet on larger temporal scales, the protection of species depends not only on the distribution of current habitats but also ensuring that species can shift their ranges under climate change. This requires tackling trade-offs between habitat protection and connectivity (Hodgson et al. 2011) and allocating conservation efforts and budgets both in and outside protected areas – using limited resources and considering a multitude of stakeholders that need to be involved in the process.

Scaling up from single to multiple species

A shift from local to large scales also entails shifting the conservation focus from particular species to wider biodiversity. This requires an understanding of the effects of connectivity and species interactions on higher ecological levels, such as communities and ecosystems. The challenge emerges from the fact that not only species, but even individuals and populations of a given species, differ from each other in habitat requirements, movement modes and capacities. Such variability and specificity can lead one to conclude that cross-taxon estimations are almost impossible to obtain. However, there are several ways to overcome this difficulty.

First, simple landscape measures – which focus on differences between landscapes – can predict functional connectivity with acceptable performance for groups of species, as long as these species share at least somewhat similar habitat requirements. This is because, if species are sufficiently similar, the effect of landscape structure tends to dominate over interspecific differences (Box 2). Yet caution is needed: not all metrics perform equally well, and not all aspects of functional connectivity are equally easy to predict (Box 2).

Secondly, clustering species and habitat types into groups, or functional types, can potentially address the needs of a broad range of species and in the process identify important trade-offs.

Finally, landscape mosaics, where a range of (semi-)natural habitats intermingle with little isolation between them, can potentially support the movements of multiple species. We therefore recommend approaches that seek to enhance landscape permeability as a whole, rather than focusing on just the physical connection between specific elements. Such an approach is particularly useful in hyper-fragmented landscapes, where no single link would suffice to support overall connectivity.

Conclusion and recommendations

Connectivity has recently become a key component of biodiversity conservation. Important natural habitats have been protected (e.g., within the Natura 2000 network in Europe), yet the connection between them often remains limited. In the meantime, pressures on biodiversity outside protected areas have grown due to continuing habitat loss, fragmentation, and intensification of land-uses. We therefore encourage a continued shift to network-thinking, and cross-scale initiatives, such as Green and Blue Infrastructure, in order to secure connectivity across scales and ecological processes.

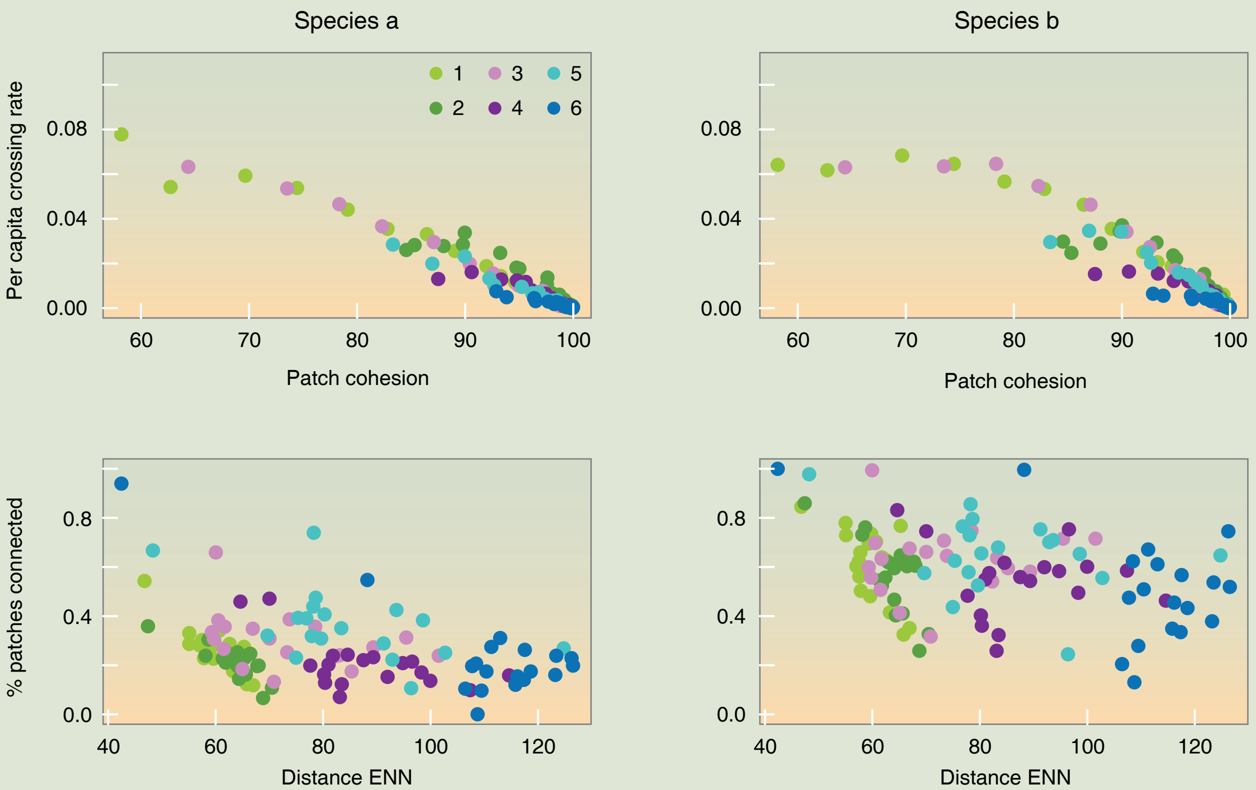
To support such initiatives, **researchers** should ensure that empirical studies on movement and dispersal are repeated across species and landscape structures, and wherever possible communicated in a standardized way that would support meta-analyses. Simulation models can aid in obtaining further generalizations on 'functional' connectivity beyond single species. Researchers should also engage more closely in science-policy dialogues to feed the vast and expanding knowledge on connectivity into biodiversity research and conservation.

For conservation **practitioners** who seek to consider connectivity in policy, planning and management, we recommend the following sequence of actions:

1. Define the aim: What do we need connectivity for?
2. Define the spatial scale of relevance: Does your aim require a local, landscape, continental, or multi-scale approach?

Box 2.

To assess the potential usefulness of simple landscape metrics for predicting 'functional' connectivity, we plotted metrics against outcomes of the *FunCon* model (Pe'er et al. 2011) across landscapes and species. Here we show exemplary results for two virtual forest bird species, one of which moves up to 30 m (left panels) and the other moving up to 600 m (right panels) beyond forest borders and into the hostile matrix. Simulations were performed over >100 virtual landscapes differing greatly in their degrees of fragmentation (scored from 1 to 6, where higher numbers represent greater fragmentation). As an example for good performance, 'patch cohesion' (Schumaker 1996) performed well in predicting the per capita crossing rate of birds during home-range movements (= number of inter-patch movements, divided by individuals and simulation steps; upper subplots). Results were also robust to fragmentation level (represented by different colours) and species (left versus right panel). By contrast, the average Euclidean Nearest Neighbour (ENN) between patches (distance, in meters) performed poorly here, as shown by the percentage of patches receiving immigrants during dispersal simulations (lower subplots). Thus, careful selection of landscape metrics can aid predicting (or ranking) connectivity across species and landscapes.



3. Identify barriers and conflicting interests: Different focal scales will entail different conservation targets and the involvement of different stakeholders.
4. Before selecting the metrics and model to use, try to view the landscape from the perspective of the species, considering their modes of perception and movement. Particularly, consider potential elements that orient or disorient individuals, such as light pollution (affecting many nocturnal animals), noise pollution (affecting e.g. marine mammals using acoustically-based orientation mechanisms), or air turbulence caused by wind turbines (affecting e.g. birds and bats).
5. Select the approach and tools for assessing connectivity based on the level of complexity of the question and the availability of relevant data. If complex interactions or trade-offs are likely to occur, consider addressing functional connectivity in detail by applying a suite of models and careful sensitivity analyses.
6. Consider the outcomes across scales. Small-scale elements may be critical to promote large-scale connectivity, but a large-scale overview would help identifying and designing even small-scale elements to alleviate habitat isolation. The challenge, then, would be to balance the various ecological needs and examine the cross-scale effectiveness of potential alternative options.

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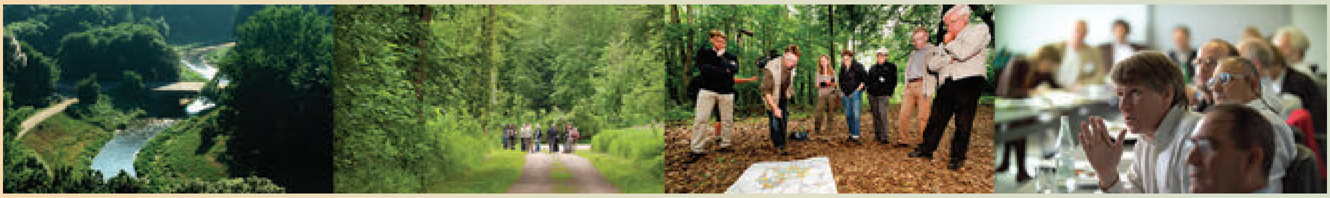
CHAPTER V



Scaling in policies and management

Systematic site selections beyond Natura 2000

Raphaël Mathevet, Pascal Marty, Jukka Similä, Riikka Paloniemi

Policy challenges

Environmental governance relates to the efforts to address environmental problems or conflicts by creating, changing or endorsing institutional arrangements. Face to the lack of effectiveness of many environmental policies aiming to conserve biodiversity within EU, multi-level institutions or multi-level governance approaches had received much attention in recent years. But like many concepts, they have attracted different meanings and implications, ranging from being a challenge for democracy to the key to achieve more sustainable environmental management.

Adapting the administrative level and spatial scale of governance to the characteristics of a specific biodiversity conservation issue and improving the participation of local and/or non-state actors in policy design and implementation are seen by many scholars and practitioners as key in the success of their effectiveness (Folke et al. 2007). The ecological functional unit is more and more considered as the unit for good planning, conservation and management, but at the same time managers still have to deal with a growing number of vertical and horizontal levels of various governance agencies and, of course, their growing interactions. This network of interlinkages between agencies, tasks, policies, territorial jurisdictions, and ecological units is highly complex and often dynamic. In this context the decentralization within policy implementation (i.e. through the key principle of subsidiarity of the EU) seems appropriated and polycentric governance should lead to long-term effective natural resource management policies (Ostrom 2005). By expecting to improve the quality of the diagnosis process, the solution suggestions and the decision-making process, the integration of lay knowledge with scientific knowledge may lead to social learning, commitment and engagement of stakeholders in the efficient implementation of the policy making outputs on the ground. However multi-level and participatory governance may lead to some major difficulties such as lack of change due to power asymmetries or high level of complex social-ecological interactions that have multi-scalar impacts (Vimal et al. 2012).

Since the mid-20th century, the creation of various kinds of protected areas (e.g. Nature Reserves, National parks, Regional Natural Parks, Natura 2000 sites), agro-environment programs, and new direct regulation (like requirements to protect specific features directly by law) has been contributing to the conservation of European biodiversity in human-made landscape and wild or re-wilding postindustrial areas. Face to the lack of effectiveness of conservation policy worldwide and in EU, the site-selection and multi-level governance are key components of a biodiversity conservation strategy. The Natura 2000 programme has been a key element in policy change towards new governance arrangements in EU. The implementation of the network completed the previous existing protected-area network. However, the implementation caused long and serious conflicts where many land-owners and natural resources users (farmers, foresters, hunters, fishermen) challenged the legitimacy of nature conservation and any new biodiversity conservation policy for many years in most European countries. The topdown policy approach used in the selection of Natura 2000 sites and the implementation of the sites' management were extremely criticized (Alphandery and Fortier 2001). Nevertheless, these conflicts have encouraged considerable changes for biodiversity conservation approaches. Public or stakeholders' participation has become a kind of mainstream in environmental governance, and as a part of this change, voluntary approaches for site-selection are emerging (e.g. participatory mapping and management plans). In truly voluntary scheme, the land-owners have power to make initiatives and final decisions, which may distort systematic selection of sites to be protected and, as consequence, affect the effectiveness of biodiversity policy. Hence, a key challenge is to combine the strengths of voluntarism and systematic decision-making.

In this chapter, we explore how polycentric governance system containing a significant degree of voluntarism, that includes several agencies and levels of governance, has evolved. We draw on the SCALES project's outcomes and, in particular, on the experiences from two case study countries, France and Finland. The two cases present innovative polycentric governance solutions found for policy challenges beyond Natura 2000 network. Solutions cover dealing with social issues through zoning and combining scientific and lay knowledge. Finally, we will conclude with a set of policy recommendations.

Responses to policy challenges

French case study: Dealing with social issues and zoning

An Environment Round Table ("Grenelle de l'Environnement") was held in 2007 by the French government. This national consultation of key stakeholders on ecological and sustainable development issues strengthened and completed the "National Strategy for Biodiversity" (SNB) created in 2004. The SNB addresses both protected areas with the "Strategy for the Creation of Protected Areas (SCAP)" and ecological connectivity with the "Trame verte et bleue" (TVB or ecological network). The SCAP aims at identifying 2% of the French metropolitan territory needing to be under strong protection which will be integrated in the TVB as "biodiversity reservoirs" linked to each other by ecological continuities. The Environment Round Table aimed at decentralizing the design and implementation of policy instruments addressing environmental issues by promoting a participatory approach with both regional state services, regional and local authorities and local stakeholders.

In this context, The "Regional Strategy for Biodiversity" (SRB) is the equivalent of the SNB at regional-scale (i.e. sub-national level). Thus, both the SCAP and TVB designed at national-scale are specified and implemented regionally. Concerning the SCAP, the state defined national priorities for the protection of biodiversity by identifying species important for biodiversity and which distribution is more or less covered by protected areas. However, this list is then refined at regional scale based on the specificities of the region. Furthermore, regional authorities are in charge of identifying new protected areas and defining the most appropriate regime of protection and their type of governance.

Regarding ecological continuity, the Regional Scheme for Ecological Coherence (SRCE) is the regional implementation of the TVB defined at national-scale. The state provides a framework to determine ecological continuities and criteria for national consistency. However, the regional authorities and decentralized state services are jointly leading the spatial identification of the SRCE. Furthermore, the choice of the method is decided by a regional committee for the TVB. Finally, the TVB/SRCE is implemented by local administration which account for it in their land planning strategies.

Unlike the SCAP, the TVB/SRCE is planned to be accounted for and integrated within existing strategies of land and urban planning. Furthermore, areas of interest for ecological continuity will not be submitted to a strong regulatory regime. In terms of participatory approach during the policy making process (e.g. still in progress in 2013), local stakeholders were mainly involved in thematic and technical committees whereas government and national agencies mainly deal with overall guidelines, methodological consistency and objectives among local territories. In this participatory process, most difficulties are due to (i) the multiplicity of group of stakeholders which tend to drag the policy design in different or antagonistic directions; (ii) scientific illiteracy of most stakeholders that should need more time to share and discuss scientific evidences and suggestions. This iterative process among the steering committee and local committees may lead to some tensions between project leaders, scientists and actors, thus to local decisions deviating from national guidelines or objectives. Overall, the main concern of participants of local committees is to know whether the SCAP or TVB will lead to regulation affecting economic activities such as farming or forestry.

Finnish case study: Dealing with social issues by combining scientific and lay knowledge

Since mid-1970s until mid-1990s in Finland, the key instrument for site selection has been the Nature Conservation Programmes. These programmes are top-down policy efforts, where the state has indicated the location of future protected areas and these plans have been implemented by environmental authorities. In the late 1990s the implementation of Natura 2000 was another top-down procedure. Citizen were in conflicts with the ways how nature conservation programs were implemented; and therefore no new nature conservation programs were adopted since mid-1990s. Instead, in order to reshape practices of nature conservation new institutional arrangements have emerged. The most significant new policy instrument is the National Biodiversity Program for Southern Finland ("METSO"). The adoption of this scheme has been a significant policy change in principle, although still now, the acreage of those protected areas established under top-down schemes exceeds that of the new approach. Still it is likely that the balance will change in near future.

METSO is a voluntary forest conservation program for both state and privately-owned lands. METSO was launched by the Finnish government in 2002, and it was first piloted during the years 2002–2007 and then converted into instrument covering most forest areas in Finland for years 2008–2016 (Government of Finland 2008). The programme focuses on the conservation of biodiversity of forest habitats all over the country except Northern Finland, where thanks to the existing wide national parks and wilderness areas the need for new protected areas is not so urgent.

The major policy changes are a shift from forced to voluntary conservation and the assignment of more powerful roles of the regional and local levels actors coming from both environmental and forestry sectors. Under the scheme of METSO, the authorities are not allowed to use compulsory purchase for conservation purposes. Instead the programme relies on incentives, including both permanent and temporary contracts for new sites to be protected. Not surprisingly, many forest owners consider voluntary conservation more legitimate in comparison to forced action. During the implementation process, forest owners' willingness to engage in voluntary conservation has increased (Paloniemi and Varho 2009).

The policy shift towards voluntary approaches with a number of stakeholders involving in the governance process has significantly increased legitimacy of site-selection procedure. The challenge is to ensure that sites selected for conservation processes within voluntary approach are reasonable in biological conservation terms. This challenge is responded by setting ecological criteria and developing a decision-support tool for prioritization. In circumstances, where conservation budget is the key constraint, this response is adequate. Still the shift towards voluntarism has its price. The average sizes of protected areas have decreased and the locations of the sites are often problematical in terms of their connectivity, despite developing decision-support tools. Would the enthusiasm of landowners to offer sites for conservation significantly decrease, picture would become even worse. This stresses not only the importance of incentive structure, but also the need to take into consideration other historical, institutional and political factors affecting motivations of landowners.

Policy recommendations

Our case study results suggested – as other scholars have concluded from empirical studies (Ostrom 2005) – that the historical, institutional and political contexts in which the site selection design and implementation take place are crucial. Taking care of multi-levels institutions acknowledges that conservation is shaped by political aspects, norms, values and power relationships issues. While this is not really new for social scientists, it is still an important progress for many conservation scientists. Integrating multi-institutions in policy designs and implementation should help in building links between science, policy and people. We still need to better understand which contextual or process factors make efficient conservation policy and site selection in order to successfully learn from them. To do so, we suggest taking into account a set of policy recommendations when building policy mixes for site-selection:

- ***More diverse and adaptive site-selection methods: Learning to learn***

The change in site selection process towards voluntarism have resulted in some countries in a situation where the average size of new sites is small, although the total number of areas selected each year has increased. A research program should facilitate the monitoring and study the consequences of this development both from ecological and social perspectives.

Many signs of progress from conflict to legitimate conservation procedures, strategies, and practices were found and should be reinforced: (i) Bottom-up development of site-selection mechanisms that increase perceived legitimacy among land-owners and govern conservation of biodiversity that exists between jurisdictional scales; (ii) site-selection mechanisms with various time scale that encourage land-owners to take pro-conservation action and learn from their own efforts, and (iii) site-selection mechanisms that take into account small- and large-scale ecological processes.

- ***Managing a place for both science and public***

Participatory and multi-level governance process in site selection should distinguishing approaches solely based on scientific frame and approaches based on co-construction with state and non-state actors. A discussion with all relevant stakeholders from all relevant decision-making levels is necessary. It is particularly important to discuss and agreed upon the quality of datasets, the uncertainties in biodiversity distribution and functioning, the working hypothesis, especially when using modelling approach.

The site selection decisions result from the evaluation of a range of – sometimes contradictory – interests (in the broad sense), of costs and consequences, whose relative weights depend largely on information available and the way in which it is treated (Mathevet and Mauchamp 2005). Thus, it seems that the knowledge of the decision-making mechanisms and its implementation is at least as important as the knowledge of the functioning of the biophysical system. The best natural science databases might not give answers to the raised questions (that usually concern human-made or dominated landscapes) if they do not associate socio-economic databases.

- ***Improving communication on scientific analysis and suggestion***

Having one or several bridging persons dedicated to communication of scientific knowledge to a broad public would most likely facilitate communication and sharing of ideas among stakeholders. Besides, defining the biodiversity policy targets at local-scale tend to focus on local concern. Consistency among regional methods and decisions could be enhanced by a better communication among these regions.

- ***Integrate a critical reflexive dimension in participation practices and deliberative decision making***

Any site selection design and implementation process is embedded in a specific political and institutional environment where any scientific suggestion or evidence analysis compete with political dimensions such a power relationships dynamics, political support or economic and social costs relatives to PAs management or creation. Any authority or management body in charge of site selection should (i) create space for deliberative experiments mixing scientists from different disciplines, inhabitants, users, technicians and also decision-makers in the same arena to present and discuss scientific findings and enrich their suggestions; (ii) monitor with both a critical social-ethnographic perspective and social engineering perspective the whole process to help the leaders to adjust the method and the arena's composition; (iii) implement social activities to develop social trust, political commitment and action.

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Governance of network of protected areas: Innovative solutions and instruments

Małgorzata Grodzińska-Jurczak, Agata Pietrzyk-Kaszyńska, Joanna Cent, Anna V. Scott, Evangelia Apostolopoulou, Riikka Paloniemi

Introduction

Although governance of protected areas (PAs) is, most commonly, realized regionally and locally, it is expected to improve the state of biodiversity at national and European levels. Therefore linking regional governance, local implementation and Europe-wide evaluation, requires a careful consideration of various scales and interactions between them. As functioning of PAs is often in conflict with current socio-economic development, their governance needs to be considered in economic, political, cultural and historical contexts as well as in relation to competition and cooperation with other sectors. PAs governance also becomes more compound as new tasks such as systematic monitoring, effective science-policy interface and scientific knowledge incorporation should be undertaken. Newly established institutional settings are often required and integration of various instruments (e.g., financial instruments, coexistence of traditional and innovative conservation ideas) is inevitable, yet not often practiced (Cent et al. 2013, Paavola et al. 2009, Paloniemi et al. 2012).

In the present chapter, we analyse selected innovative solutions of institutional and societal character recently developed and practiced in three case study countries: Greece, Poland and the UK. The cases are selected to present the diversity of scale challenges of PAs governance the case countries have been or are still facing. We explore current institutional and societal challenges of PA governance and a new, but short and simplified, analytical framework to link cases to each other. Institutional challenges cover, in our analysis, a search for governance innovations to "fit" ecological needs as well as socio-economic, cultural and historical contexts. Neither environmental governance nor management of PAs exist in a "policy vacuum", so they incorporate a selection of policy instruments (not only those targeted specifically to PAs management) and consequently create opportunity for potentially innovative ideas (Paavola 2009). In the case of societal challenges, we consider the fact that increased establishment and management of new PAs, such as Natura 2000 (N2000) sites, covers a variety of land categories with different ownership status, types of land use and levels of human activity. Both governance and management should therefore respect land use issues and conservation of biodiversity and ecosystem services while assuring participation in decision making (e.g. in designing the PAs borders, developing management strategies or planning conservation measures) (Gibson et al. 2000, Rauschmayer et al. 2009, Tikka and Kauppi 2003), and just and fair distribution of costs and benefits of conservation (such as distribution of direct and indirect costs, access to nature, opportunity costs, etc.) (Balashenko et al. 2005, Piper 2005). Conservation measures need to not only incorporate rules and restrictions, but to be efficient, as already proved in many EU member states, by emerging cooperative actions at the local level (Alphandéry and Fortier 2001, Chmielewski and Krogulec 2008). Effective governance should subsequently address biodiversity conservation issues inside and outside strictly designated PAs, the latter to be highly interrelated with societal challenges.

Responses to institutional and societal challenges

Governance as an arrangement of governing beyond-the-state is defined by Dingwerth (2004) as "*the socially innovative institutional or quasi-institutional arrangements of governance that are organised as horizontal associational networks of private (market), civil society (usually NGOs) and state actors*". These new articulations between state, market and civil society generate new governance forms combining the three 'moments' of society in new and often innovative ways (Swyngedouw 2004, cited in Swyngedouw 2005).

In Greece, along with the increase in number and types of protected ecosystems, the variety and number of involved actors has increased. Thus, especially since the 90s, there have been a number of new governance arrangements towards the above direction through the emergence of an expanded role for non-state actors, sharing of responsibilities and establishment of partnerships between the state and representatives of the private sector and the 'community', mainly in the context of EU funding schemes and in implementing EU directives (Apostolopoulou et al. 2012).

The institutional changes in Greece enabled a broader participation of non-state actors in PAs management. Apart from the management agencies several multi-sectoral and multi-level cooperation networks have been created (e.g., National Committee of Governmental Planning and Sustainable Development Policy, National Board of Planning and Sustainable Development, Committee Nature 2000). Moreover, in the context of Community Support Frameworks (CSFs), Life-Nature projects and the operational program Environment, several NGOs, actors from the local administration (such as development agencies, municipalities, prefectures and regions, research institutes, universities, and management agencies) participate in the implementation of conservation policy by conducting environmental studies (including Specific Environmental Studies necessary for the designation of the majority of PAs), monitoring schemes, management measures and plans.

Simultaneously, the recently implemented Kallikratis – a state spatial restructuring plan – which promoted a new governance architecture of regional and local administration (Greek Law 3852/2010) led to a wide-ranging reorganization of all governance levels towards a new governance architecture of regional, local and decentralized administration. The latter gave a significant administrative and budgetary autonomy to regions by transferring powers from central government to the regional authorities including their overall development strategy. It is worth mentioning that, inter alia, the General Secretary of the Decentralized Authority is now responsible for the selection of certain types of protected areas following a less "strict" approach than the one described in the main Greek environmental law 1650/86. In particular, instead of a Specific Environmental Study (SES) now a so-called "special report" just describing the ecological importance and the protected values of the area can be sufficient for the designation of some types of protected areas.

Despite the increased involvement of various stakeholders in Greek governance, this rescaling of governance in practice transforms existing power geometries (Swyngedouw 2005). The contradictory character of many of these innovative governance arrangements is related to the fact that they have often been guided not by the need for meaningful cooperation and coordination between and within governance levels, but these have been rather pushed by political or economic interests (Apostolopoulou and Pantis 2010, Apostolopoulou et al. 2012). A characteristic example is the establishment of management agencies in priority N2000 sites which in some cases has contributed to the extension of the state's power, has been linked with explicit privatization efforts, or has facilitated private funding for protected areas. In other cases, management agencies contributed to the creation of new arenas for negotiation of conservation goals by directly involving through official procedures state and non-state actors in conservation politics and allowing them to negotiate the boundaries and size of PAs as well as the restrictions imposed. However, in most cases the outcomes of these negotiations, the terms of participation or the selection of stakeholders who could participate, were determined by powerful interests (Figure 1).

In the case of Poland, implementation of N2000 triggered changes in organization of public services for nature conservation. On-going decentralization strengthened regional level administration, followed by creation of separate highly independent bodies (Regional Directorates of Environmental Protection, RDEP) less prone to political influence. Delegation responsibilities for N2000 to the regional level resulted in closer and more functional co-operation between the regional and local level, mainly due to incorporating more broadly local actors (e.g. local governments, NGOs, leaders, etc.) into management activities. Institutional changes have strengthened conservation bodies by providing independence from regional administration and financial resource allocation. A drawback of this reorganisation is the fact that it has and is still weakening the role of landscape protection areas (landscape parks). Together with a decrease of employees in administration, the responsibility for this types of protected areas and a large part of decision making power were shifted to the regional and local authorities, which may cause more development pressure on the protected landscapes. Priority given to the implementation of N2000 and fulfilling the EU obligations towards nature conservation resulted in the state giving too little attention to the development of effective instruments to support landscape protection. Instead, main financial and human resources have been allocated to habitat and species conservation (Figure 2).

History and evolution of public participation in Poland is not as rich and long as in the EU-15 mainly due to the historical and societal factors (e.g., a considerablly late Europeanization of policy, a limited communication between government and public, obligatory involvement activities during the socialistic regime)

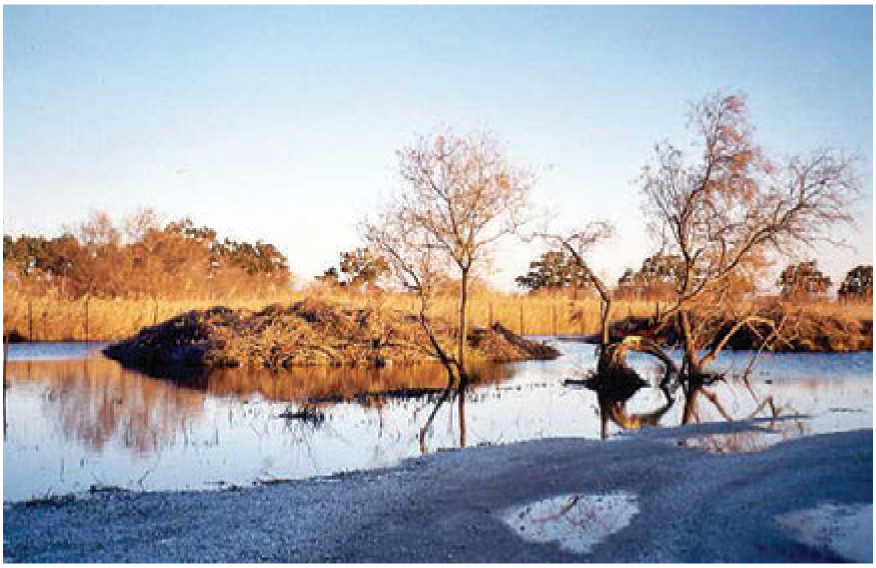

Figure 1. Schinias National Park (Attica, Greece). A case of biodiversity governance based on public-private and multi-stakeholder partnerships (photo: Evangelia Apostolopoulou).

(Hicks 1996). Public participation is regulated legislatively right now and regarded as a needed and useful tool for PAs governance (Mitter 2003, Lazdinis et al. 2007, Bell et al. 2008). The first systematic public consultation scheme concerning designation of protected sites within N2000 started in Małopolska – a southerly located region of Poland in 2008. Primarily, it was aimed at gaining public opinion on the sites selected by the experts, and then was broadened to include of local communities in development of separate conservation plans for each site (Grodzińska-Jurczak and Cent 2011). Considerable success was achieved practically from the very beginning of the consultation scheme paralleled by gaining detailed information on the local conflicts over N2000. Positive experiences from the program resulted in growing importance of public consultation activities undertaken by conservation institutions outside the region (Figure 3). Since then, public consultation has been organized as open meetings accompanied by establishing local cooperation groups of various stakeholders (experts, local authorities, NGOs, citizens), and they have become obligatory while developing management plans for N2000 sites (Grodzińska-Jurczak et al. 2012, Cent et al. 2013).

In the UK a strong tradition of protecting important sites for environmental or aesthetic reasons has been observed since the Nature Conservancy Council was formed under the National Parks and Access to the Countryside Act in 1949. Since this time, there have always been statutory nature conservation bodies in the UK. The biggest institutional changes and innovations came with the devolution of responsibilities to separate nature conservation bodies for each UK country in 1990; currently these are Natural England, Scottish National Heritage, Countryside Council for Wales and Northern Ireland Environment Agency. The devolution of these organisations provides more regional control over nature conservation, while the Joint Nature Conservation Council provides an overarching service for the whole of the UK, collating information and designating European protected sites. All of these bodies are funded by grant-in-aid from the UK or devolved governments.

Public participation in UK PAs governance is more common than in Greece or Poland. Local land-owners and communities are consulted over plans for nationally and internationally protected sites, and planning policies are particularly inclusive, offering consultation with members of the community, and potentially allowing the designation of locally important green spaces by local communities.

Despite the overall regulatory character of the UK's environmental policy, Europeanization of national environmental policies has pushed towards the adoption of new instruments. Thus, the UK has developed several instruments to address the societal challenges of PAs governance. A high proportion of land is owned by individuals, who have a substantial influence on the protection and management of designated sites. Therefore partnerships are often seen in the management of protected sites, with Environmental Stewardship being one of the most important and common mechanisms for this. 90,000 ha of land in England was in Higher Level Stewardship between April 2011 and March 2012 (Natural England 2012). Economic instruments also encourage governance of protected areas by third parties. The UK is currently considered as one of the EU leaders in adoption and innovation in this area. For example, the Conditional Exemption Tax Incentive Scheme (or Heritage Relief) is used to support the management and protection of heritage land and property by private owners. If land is proven to be of outstanding scenic, historic or scientific interest (e.g., designated as an Areas of Outstanding Natural Beauty (AONBs) or a Site of Special Scientific Interest (SSSI) then private owners can receive conditional exemptions from capital gains or inheritance tax. In return for tax exemption owners are required to maintain or manage the land, preserve its character, and allow reasonable public access. Gaining these kinds of tax relief is difficult and statutory nature conservation bodies such as Natural England monitor such land to ensure the conditions are being adhered to. These solutions are not easily available in most other European countries; in most cases landowners are not given financial compensations (as tax reduction or in any other form) due to

Figure 2. Management of local car traffic related to presence of large carnivores in Poland. The sign saying "Slow down, bears in the area" is a part of an awareness campaign of WWF Poland on large carnivores. The roads in the area are local, of low importance, and have been recently renovated – this example shows that conservation does not necessarily interfere with development (photo: Joanna Cent).

Figure 3. Consultation meetings in Poland. Regional Directorate of Environmental Protection, Kraków first informs and explains principles of N2000 program, then engages local stakeholders in discussion on management of the sites (photo: Magdalena Szymańska).

inclusion of their land into the sites, they are required to undertake environmentally-sound activities in order to receive funding, like in the agri-environmental programme. Claiming compensation is especially difficult for owners of small land parcels, for example agri-environmental schemes where cost at the entry (while preparing application) hinders participation of small owners in the program.

Our study clearly illustrates that existing policy instruments, their implementation and innovative solutions developed within each case study country differ due to political and socio-economic context. Although solution to PAs management have been addressed to fit national and regional regimes, all investigated countries still face the problem of governance effectiveness. This challenge, its causes, scope and context are complex, in general they all refer to the issues of public participation.

Policy recommendations

A wider inclusion of public participation approach in PAs governance emerged as innovative, in all three case study countries, although in different ways and considering different aspects of PAs governance. Slowly but steadily, there is a gradual shift from top-down to more complex decision-making processes based on multilevel governance approaches.

- Localized innovations are needed for promoting a process towards sustainability

Innovations are not at end but a means to promote sustainability. Although they do not give a promise of immediate improvements, they should, particularily in the EU context, be designed and implemented using a complex socially and environmentally meaningful mode, providing at the end a basis for smart, sustainable and inclusive growth targeted in the EU 2020 Strategy (Van den Hove et al. 2012). Hence the innovative character of described solutions in the PAs governance within case study countries is influenced by, and in turn influencing, both social and environmental contexts with far from simple or self-evident consequences. The nature of current innovations is undoubtedly controversial as they are aiming to respond to scale challenges while, in many cases, producing new ones.

- Seriously taken public participation should be organized

Innovations towards enhancing public dialogue and promoting new participatory and fair arrangements in environmental governance and PAs management should be encouraged. For example, organizing open meetings supporting cross-level and cross-sectoral cooperation between stakeholders and authorities working at local and regional levels should be fostered. Making it obligatory when developing management plans for N2000 sites might be desirable to assure sufficient scope (in terms of both areas and invited participants) and allocation of resources, as well as to enable social learning. Good experiences from such efforts have been received from Poland.

- New institutional arrangements needed for governing PAs management should be supported

The establishment of official institutions consisting of a variety of actors from different governance levels with the responsibility for PAs management could be beneficial for dealing with scale challenges. However, as the example of management agencies from Greece has showed, such governance arrangements should be based on clearly defined goals towards promoting social-ecological resilience, be supported from the state with funding and qualified staff and be carefully designed in order to ensure the equal involvement of all relevant stakeholders and especially of local community groups.

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Ecological fiscal transfers: A policy response to local conservation challenges

Rui Santos, Irene Ring, Paula Antunes, Pedro Clemente, Thais Ribas

Introduction

Intergovernmental fiscal transfer schemes redistribute public revenues from national and regional governments to local governments with the dual aim of providing the latter with financial resources to fulfil their local public functions and helping to reduce fiscal inequalities (Boadway and Shah 2007). The redistribution of public revenues to lower levels of government is usually based on socioeconomic indicators, reflecting the acknowledged relevance of the associated public functions. However, intergovernmental fiscal transfers can also be an effective instrument to support the local provision of ecological goods and services (Ring 2002, Köllner et al. 2002, May et al. 2002, Ring 2008a, b).

Ecological Fiscal Transfers (EFTs) are distributed on the basis of ecological or conservation-based indicators and are allocated in the form of either lump-sum or specific-purpose transfers (Ring et al. 2011). They can take into account spillover benefits and can also offset opportunity costs (e.g., resulting from land-use restrictions) and/or local public expenditure on conservation activities. For these reasons, EFTs are widely considered to be an innovative instrument that not only have the potential to offer an incentive to local governments to enhance the quality of conservation areas within their territories but also, in doing so, provide ecological benefits that flow beyond municipal boundaries (Ring 2008a, TEEB 2011).

With the 2007 amendment of the Portuguese Local Finance Law (LFL – Law 2/2007, 15th January), Portugal became the first European Member State to integrate EFTs within the system of annual fiscal transfers from the national to the local level (municipalities). Article 6 of this law, which focuses on the promotion of local sustainability, establishes that *"the financial regime of municipalities shall contribute to the promotion of economic development, environmental protection and social welfare"*. This general objective is furthered by means of several mechanisms relating to the redistribution of public revenues from central to local governments, including especially positive discrimination towards those municipalities with land classified as part of Natura 2000 Network or other national protected areas (Santos et al. 2012a,b).

This positive discrimination is implemented by applying ecological criteria (classified areas) as part of the set of indicators used to determine the distribution of the General Municipal Fund (FGM) (see Figure 1), which is allocated to municipalities as follows:

- 5% is distributed equally to all municipalities;
- 65% is allocated as a function of population density (weighted in order to benefit less populated municipalities) and of the average number of stays in hotels and at camp sites;
- 30% is distributed on the basis of the municipalities' area, topography and land located within conservation areas:
  - in municipalities with *less* than 70% of their territory located within conservation areas, 25% is distributed in proportion to the area weighted by elevation levels and 5% proportionally to the land classified as part of Natura 2000 or other protected areas;
  - in municipalities with *more* than 70% of their territory located within conservation areas, 20% is distributed in proportion to the area weighted by elevation levels and 10% proportionally to the land classified as part of Natura 2000 or other protected areas.

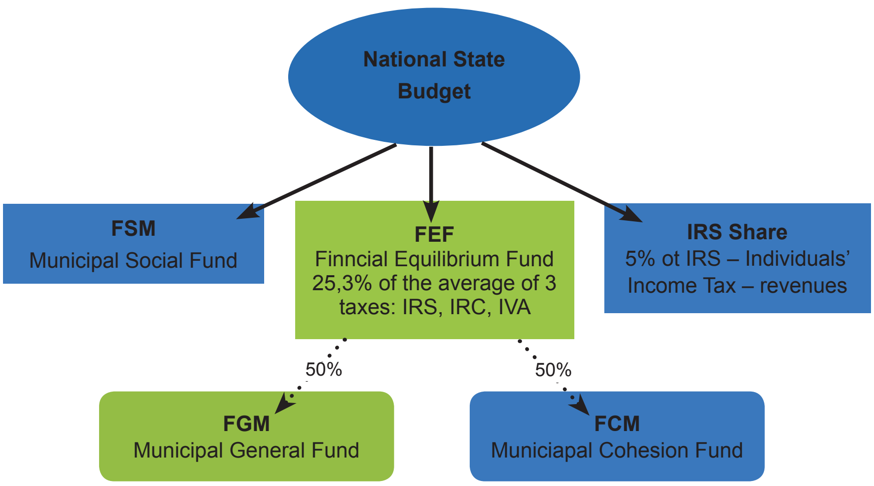

Figure 1. Allocation of State funds to municipalities according to the Portuguese LFL

The principle adopted for this intergovernmental fiscal transfer is non-earmarking, meaning that beneficiaries (local governments) are free to decide how they will use the funds transferred. The only ecological criteria in play in this law are the total area under protection and the percentage of municipal land designated as protected area.

Importance of fiscal transfers

Intergovernmental fiscal transfers from the central government are an important source of revenue for Portuguese municipalities, providing, on average, 60% of total municipal revenues. Alongside fiscal transfers, municipal revenues come from various other sources, such as direct taxes (e.g., property taxes – Imposto Municipal sobre Imóveis) and indirect taxes/tariffs (e.g., water and sanitation).

In the majority of Portuguese municipalities, fiscal transfers represent more than 75% of total municipal revenues both for 2008 and 2009, as shown in Figure 2 (Santos et al. 2012a,b). However, the importance of fiscal transfers for municipal revenues differs significantly among the municipalities. In 2008, for example, fiscal transfers accounted for 25% of municipal revenues in Lisbon and for 97% in Barrancos. The importance of fiscal transfers is greater for inland municipalities than for coastal municipalities, as the latter are typically more populated and developed, having other significant sources of revenue such as property taxes, a municipal tax on vehicles, water supply and sanitation tariffs, waste management tariffs, and licensing fees.

In this context, the changes contained in the Local Finance Law regarding the criteria for fiscal allocation may have a significant impact on the funding of municipalities, not least with regard to the development strategies adopted by those municipalities with a high level of dependency on fiscal transfers.

Ecological transfers

A more detailed analysis of the ecological incentive given by the new LFL shows that ecological transfers are very significant in terms of both total municipal fiscal transfers and ecological fiscal transfers received per hectare of classified area, particularly in municipalities with more than 70% of municipal land designated as classified area for biodiversity conservation (Figure 3) (Santos et al. 2012a and Santos et al. 2012b provide more detailed data and analysis on this subject).

In 2008 the ecological transfers for this group of municipalities represent on average 24% of total municipal transfers and 18% of their total municipal revenues. For example, in the municipality of Castro Verde, this dependency is particularly high: the ecological component was 37% of the total fiscal transfer and 34% of total municipal revenue (Santos et al. 2012a,b).

The ecological transfer in 2008 was 49 Euros per ha of protected area for municipalities with more than 70% of their territory covered by protected status. In the remaining municipalities with less than 70%, the value is 25 Euros/ha. The spatial representation of this indicator per municipality across Portugal is shown in Figure 3b (Santos et al. 2012a,b).

Policy recommendations

To date, Ecological Fiscal Transfers have typically been designed to compensate principally for the opportunity costs faced by public actors, as is the case with the Portuguese scheme (Santos et al. 2012a,b) and the ICMS-Ecológico in Brazilian states (May et al. 2002, Ring et al. 2011). However, this instrument can also be designed to provide compensation for the costs of managing public conservation areas – implying an active role on the part of public actors in implementing conservation measures – or to compensate local governments for expenses incurred in providing spillover benefits to areas beyond their boundaries (e.g. ecological values and ecosystem services). These different objectives should be clearly stated and rendered transparent to all relevant stakeholders at the design stage.

The indicators used for the allocation of Ecological Fiscal Transfers need to accurately reflect the objectives established for the instru-

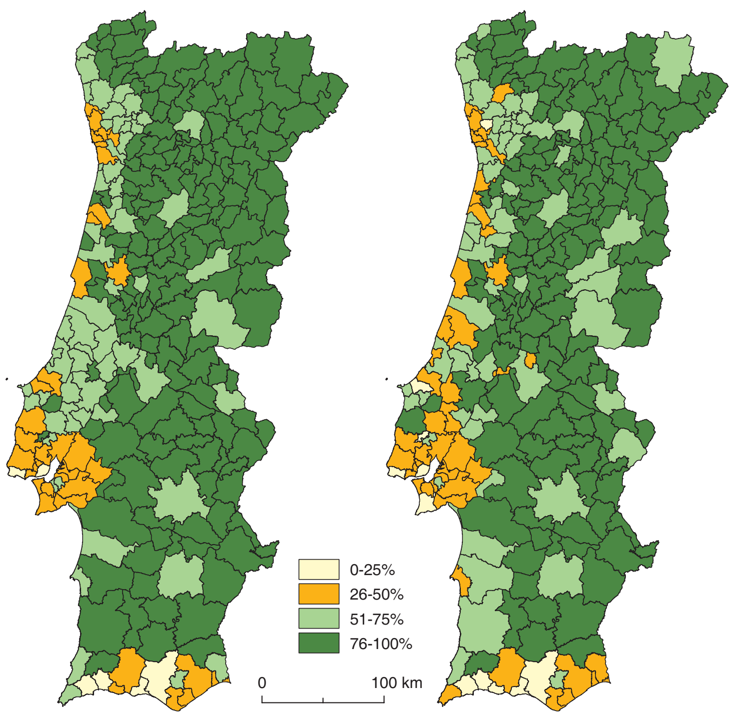

Figure 2. Direct fiscal transfers shown as a proportion of total municipal revenues in 2008 (on the left) and 2009 (on the right)

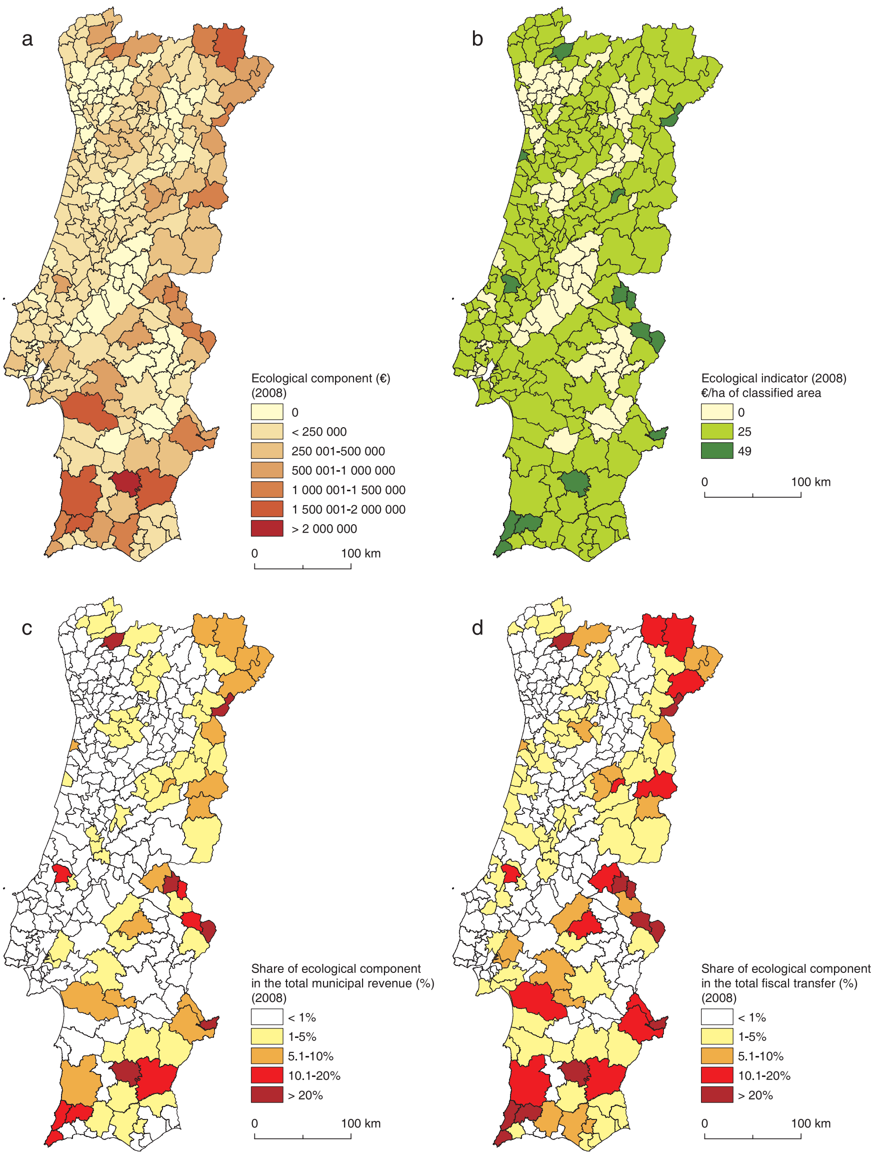

Figure 3. Ecological component in euros (a), ecological indicator in Euros per hectare of classified area (b), share of the ecological component as a proportion of total municipal revenue (c), and share of the ecological component as a proportion of total fiscal transfers (d).

ment. Several indicators, or sets of indicators, are currently used in EFT schemes. One common approach is to use a quantitative criterion (i.e., amount of protected area per municipality) as a proxy for the importance of the ecological functions provided by a given territory. However, additional criteria are needed in order to reflect the quality of management and the relative importance of different conservation areas subject to different levels of protection and land-use constraints. The objective should be to strengthen the incentive to create new conservation areas while at the same time maintaining and managing existing protected biodiversity values and services. Regardless of the option chosen in each case, it is critical that the selected indicator(s) are aligned with the conservation policy objectives and the goals of the instrument, whether they are to compensate for opportunity/management costs or to include recognition of spillover benefits.

Currently, several EFT schemes, like the Portuguese one, do not take into account the quality or level of protection of different categories of protected areas or the ecological goods and services provided by areas outside nature conservation networks. This is acceptable in a first stage of implementation, in order to avoid schemes that are too complex and to allow for gradual change in the mindset of decision makers and other relevant stakeholders. However, it is important to analyse the opportunities available for introducing changes that can improve the effectiveness of the scheme as well as its performance regarding other relevant policy criteria (e.g. efficiency, in terms of financial compensation linked to the positive externalities each municipality provides to society, and equity).

For example in Portugal, municipalities without classified areas on their territory do not receive the ecological fiscal transfer. However, the fact they do not encompass protected areas does not mean that they are not contributing to or investing in ecological aspects. Ecological services and values are not restricted to protected areas or Natura 2000 sites. We have explored elsewhere the introduction of additional ecological indicators that better reflect ecosystem services and their values provided by the territory and that can incentivise all municipalities to engage in conservation activities (Santos et al. 2012b). The use of an indicator accounting for the provision of cultural, regulating and supporting services would imply that all municipalities received funds, reflecting that they all provide ecological services at some level. However, ecological transfers received by municipalities would then be very low and more uniformly distributed, when compared to real transfers in 2008 (Santos et al. 2012b). This would dilute the presently acknowledged importance of protected areas by local actors and therefore may be counterproductive for achieving biodiversity conservation objectives.

One of the main aims of EFTs is to compensate local authorities for costs or benefits linked to biodiversity conservation or ecosystem services. The schemes can be implemented by means of lump-sum transfers or by earmarking (a proportion) of the ecological fiscal transfers for specific purposes. To date, the principle adopted for fiscal transfers in Portugal is non-earmarking, meaning that beneficiaries (local governments) are free to decide upon their use. By earmarking EFTs, however, it is possible to create a causal link between municipal conservation measures and the ecological indicators adopted. The ecological transfer could be allocated – either partially or completely – to specific measures implemented or coordinated by local authorities. This change would significantly improve the conservation effectiveness of the instrument. However, earmarked transfers tend to have a powerful influence on municipalities' freedom to decide their own priorities. For this reason they represent the main argument (from a public finance perspective) in favour of most fiscal transfers taking the form of lump-sum payments. Earmarking fiscal transfers for conservation purposes implies a deviation of funds that may be needed to the fulfilment of other important public functions and therefore this issue should be carefully discussed with interested actors, in particular municipal authorities.

When EFT earmarking is not allowed due to fiscal rules, it is still possible to design an EFT scheme that provides an additional reward to those municipalities that voluntarily take the initiative to dedicate resources from their municipal budgets to support the implementation of specific conservation measures, whether undertaken directly by them or by private local actors. This reward scheme could be based on an additional criterion drawing upon historical and validated information on direct conservation costs supported by each municipality, for instance. It can also be articulated with other incentive instruments oriented for private actors (e.g. agri-environmental measures, PES, forest code). However, the introduction of additional criteria and rules in an EFT scheme increases its complexity, which is not desirable, in particular in early stages of implementation.

Some of the conservation costs that are incurred by municipalities and district governments – local public actors – are frequently neglected. For example, conservation may mean missing out on development opportunities, thus leading to a reduction in municipal budgets as a result of forgone local taxes. This may generate a negative response among local authorities. EFTs can help to turn local opposition to protected areas into active support by internalizing the positive externalities of protected areas and other conservation measures, thus providing an incentive and creating in local authorities a new mindset more favourable towards biodiversity conservation. To do so, it is necessary for all relevant stakeholders to be actively involved in the design of the EFT scheme from the very beginning of the process. In addition, the instrument must be transparent and easily understood by all stakeholders. In the Portuguese case, despite the ecological component positively discriminating municipalities with a high percentage of classified area, the significant number of changes simultaneously introduced by the new LFL made it difficult for the stakeholders affected to fully understand the ecological component. One particular difficulty here was the presence of several crossover effects which effectively hid the financial incentive offered to municipalities by the ecological transfer.

EFTs are capable of being integrated in a flexible way into the current policy mix of each country. For example, a proportion of EFT funds can be channelled by the receiving local authorities towards private sector actors, for instance to support conservation programmes and activities. This provides an opportunity to align the incentives for local public and private sector actors and to increase effectiveness and efficiency.

Furthermore, intergovernmental fiscal transfer schemes can be combined with poverty alleviation objectives (OECD 2005), an important characteristic for designing policies in developing countries. They may also play a role in the implementation of international programmes at a national scale, linking climate mitigation and biodiversity conservation policies (Irawan et al. 2014, Ring et al. 2010).

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EU Green Infrastructure: Opportunities and the need for addressing scales

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Green infrastructure: A new EU policy concept

Despite on-going efforts, widespread losses of biodiversity, ecosystems and ecosystem services are continuing in the EU, with associated detrimental economic and social impacts. Following the failure to meet the target of halting the loss of biodiversity by 2010, a number of policy shortcomings have been identified, several of which are, either directly or indirectly, linked with the inability of the EU policy framework to effectively address the scale-related challenges of biodiversity conservation (European Commission 2010a, 2010b). These scale-related issues include, for example, failing to safeguard functional linkages between individual Natura 2000 sites across the wider landscape, and not being able to effectively ensure biodiversity conservation beyond protected areas, especially in agricultural and forest ecosystems, the management of which requires the involvement of a range of administrative levels and stakeholders (Paloniemi et al. 2012, Primmer et al. 2014).

The EU policy framework for green infrastructure – still under development – aims to promote more holistic policy solutions within the Union, including supporting the integration of scale-related issues into decision-making as a means of facilitating the success of the new biodiversity goals for 2020 (European Commission 2013a). Green infrastructure, as understood in the EU policy context, is a multifunctional concept that combines biodiversity conservation with the maintenance of ecosystem services, i.e. the contribution of ecosystems to human well-being (Box 1). It builds on the recognition that ecosystems, including ecological networks, require functional connectivity to maintain ecological processes and enable species to disperse and migrate where necessary (Tischendorf and Fahrig 2000, Mazza et al. 2011, Marsh et al. 2014 this book, Pe'er et al. 2014 this book). While the essential contribution of green infrastructure in mitigating fragmentation and unsustainable land use is still an integral part of the EU framework for green infrastructure, the role of ecosystems in providing wider multiple benefits, including maintaining ecosystem services, has more recently been afforded greater emphasis. Therefore, EU green infrastructure policy acknowledges that protected areas and ecological networks alone are insufficient to maintain biodiversity, ecosystems and associated services, and there is a need to maintain and restore wider

Box 1. Definition of green infrastructure in the context of EU biodiversity policy

Green infrastructure (GI) is a strategically planned network of natural and semi-natural areas with other environmental features designed and managed to deliver a wide range of ecosystem services. It incorporates green spaces (or blue if aquatic ecosystems are concerned) and other physical features in terrestrial (including coastal) and marine areas. On land, green infrastructure is foreseen to be present in both rural and urban settings (European Commission 2013a). In addition to the provision of ecosystem services, the aim of green infrastructure is to enhance those aspects that underpin the delivery of these services, including ecosystem resilience and preservation of biological diversity. This contributes to the overall efforts of conserving biodiversity (European Commission 2013b).

The types of physical features that contribute to GI are diverse, specific to each location or place, and vary also in accordance to scale (European Commission 2013b). In general, GI is considered to consist of the following main elements: 1) core areas, 2) restoration zones, 3) sustainable use / ecosystem service zones; 4) green urban and peri-urban areas, 5) natural connectivity features and 6) artificial connectivity features (Mazza et al. 2011, Green Infrastructure Working Group 2011b).

areas of the landscape through proactive, strategic and coherent actions across all policies that influence land use (Mazza et al. 2011).

This chapter explores the scale-related aspects of green infrastructure implementation in the EU. We identify different scale-related requirements for successfully implementing EU biodiversity policy and highlight how the EU framework for green infrastructure can help to address these requirements. Furthermore, we assess the current level of integration of green infrastructure into biodiversity policies in a number of EU Member States. Based on the results, we draw preliminary conclusions on the integration of green infrastructure into EU biodiversity policy up to 2020, with a specific focus on the applicability and added value of the concept in addressing scale-related issues.

Methods

An EU-level policy assessment was carried out to determine whether – and to what extent – the integration of the concept of green infrastructure into the EU 2020 biodiversity policy could improve the scale-sensitivity of conservation within the Union. In this context, the term 'scale-sensitivity' was used to refer to the design of a policy instrument, describing the ability of a given policy instrument to recognise and address different scale-related issues. The assessment focused on exploring the role and importance of five different types of scales in biodiversity conservation: spatial, functional, societal, jurisdictional and temporal scales (Primmer et al. 2014, Pe'er et al. 2014) (Box 2).

The assessment of EU green infrastructure policy was based on a number of wider policy analyses, the results of which are outlined in Kettunen et al. (2012). All these analyses built on a comprehensive review of relevant EU policy documents. Firstly, scale-related requirements for implementing the EU biodiversity policy to 2020 (European Commission 2011) were identified. Secondly, an assessment of the existing EU policy instruments for conservation was carried out to determine to what extent the existing EU policy framework was able to address the scale-related needs of EU biodiversity policy up to 2020. Based on these two analyses, a range of gaps and shortcomings in the current EU policy framework were identified (see Kettunen et al. 2012 for detailed results). Finally, the potential of – and scale-related requirements for – a dedicated green infrastructure policy to successfully address these shortcomings was assessed.

In addition to the EU level, dedicated assessments of existing policy frameworks supporting green infrastructure at national level were carried out in five different EU Member States (Finland, France, Greece, Poland and England in the UK). These assessments were based on a review of national policies relevant for biodiversity conservation. The insights of these national analyses, including opportunities and gaps in national frameworks, were used to identify key aspects that should be taken into consideration when further developing the EU policy on green infrastructure.

Results

Scale-related opportunities and needs of EU green infrastructure

The concept of green infrastructure can help to address all different types of scales considered in the context of this study, including interlinkages between the scales.

Spatial and functional scales: A key aim of an EU green infrastructure policy framework is to support the maintenance and restoration of areas that are important for both biodiversity and provisioning of ecosystem services for human wellbeing (European Commission 2013a, Green Infrastructure Working Group 2011a). Improving the quality of wider landscapes by increasing the area of sustainably managed and/or protected areas is foreseen to be essential to the EU framework for green infrastructure (European Commission 2013a, Mazza et al. 2011, Green Infrastructure Working Group 2011a). Similarly, the maintenance and restoration of ecosystem functions and related services, in particular on a landscape scale, also play an important role in the policy. This can be achieved, for example, by improving the management of ecological networks. Consequently, it is expected that the implementation of a green infrastructure policy at national, regional and local level will, to a large extent, be carried out through inte-

Box 2. Definition of scales used in the context of the study

The term 'spatial scale' was used to describe the different geographic dimensions related to the distribution of species, habitats and ecosystems within the broader landscape.

The term 'functional scale' refers to the way different species use the landscape within their life cycle and/or how ecosystem processes take place on different scales within the physical environment.

The term 'societal scale' applies to the different stakeholders that play a role implementing biodiversity policy in the EU.

The term 'jurisdictional scale' refers to the different jurisdictional instances relevant for sustainable use and conservation of biodiversity. These include the different levels of governance (from local to international) and the interaction between these levels influence the conservation of biodiversity.

The term 'temporal scale' was used to describe the duration of different ecological and anthropogenic processes relevant for biodiversity conservation.

grated spatial planning, taking spatial and functional scales of conservation better into account.

Societal scale: Given the need to address ecosystems and their functions at a landscape level, it is also likely that a significant part of those areas relevant to maintaining green infrastructure will be located on private lands. Consequently, carefully designed participatory approaches, involving all relevant stakeholders and beneficiaries of ecosystem services, are required to secure the effective implementation of EU goals for green infrastructure (cf. Paloniemi et al. submitted). For example, a range of envisaged initiatives supporting green infrastructure, such as introducing payment schemes for ecosystem services (PES) to bring areas under sustainable management, are foreseen to be based on (voluntary) partnerships between different stakeholders. In general, the concept of green infrastructure equally recognises both private and public values associated with an area and thus also supports more inclusive and socially sustainable decision-making processes and practices. Consequently, it is foreseen that the societal scale – i.e. relationships between different stakeholders – is envisaged to be an integral part of the future EU framework for green infrastructure. Scale-related considerations are also required to determine how policy measures at the EU scale translate into concrete decisions on land use planning among local stakeholders.

Jurisdictional and time scales: The existing EU policy framework for green infrastructure does not yet provide direct or explicit provisions related to functioning and cooperation of different jurisdictions for biodiversity policy (i.e. jurisdictional scale) (European Commission 2013a, Green Infrastructure Working Group 2011c). Similarly, it does not explicitly address issues related to the duration of ecological and anthropogenic processes relevant for biodiversity conservation (i.e. temporal scale). Therefore, it remains to be seen how, and to what extent, these considerations will be integrated into the future framework. The results of our policy assessment highlight the importance of addressing these two scales. Aspects related to jurisdictional scale and land use planning in the EU, including links between relevant policy sectors and issues related to EU and Member State competence, need to be clarified, addressed and overcome when further designing the EU framework for green infrastructure. Finally, as with all aspects of biodiversity conservation, consideration of both short- and long-term impacts will play an important role in implementing practical green infrastructure measures.

Interlinkages between scales: ccording to our assessment, the concept of green infrastructure can help to address some important interactions and linkages (e.g. conflicts and synergies) between different scales. For example, the concept makes explicit links between spatial and functional aspects of conservation through the concept of ecosystem services. It also takes into consideration benefits provided by well-functioning ecosystems to different individual stakeholders and the general public, thus acknowledging that societal and ecological aspects need to be addressed in a more coherent manner.

Our assessment shows that green infrastructure is a cross-cutting policy concept that aims and, in order to be successful, needs to simultaneously address several actors, sectors and scales. This requires making links with existing policies and instruments that govern the use of land and resources within different ecosystems. Consequently, the uptake and implementation of EU green infrastructure needs to be supported by several existing policy instruments, including mainstreaming the concept into relevant sectoral policies, both EU and national, and earmarking funding to support these initiatives. For example, it could be envisaged that EU Directives for Environmental Impact Assessments (EIA), Strategic Environmental Assessments (SEA) and Environmental Liability could play an important role in mitigating negative impacts on green infrastructure. Similarly, several EU funding instruments, such as instruments available for funding rural and regional development, are expected to be crucial in promoting the uptake of green infrastructure initiatives at national and regional level.

Green infrastructure in the national context: Insights from Member States

Our national level analysis reveals that none of the five Member States considered in the context of this assessment (Finland, France, Greece, Poland, England) have a dedicated national policy framework or instrument in place that would be fully compatible with an EU framework for green infrastructure. Furthermore, the existing national frameworks for green infrastructure appear fragmented, consisting of multiple policies that are primarily oriented towards biodiversity conservation or the conservation of key natural resources (i.e. agriculture, forests and water).

In France, a national framework for green and blue infrastructure exists (Trame verte et bleue – TVB). However, the definition of TVB focuses on biodiversity and ecological connectivity only, with no direct reference to ecosystem services. In England, green infrastructure has been a policy priority for the past years, its emphasis moving towards landscape connectivity and maintaining the functioning of ecosystem services. However, dedicated policy instruments operationalizing the concept have been adopted only recently: a significant number of local councils are currently developing green infrastructure strategies as suggested in the 2012 National Planning Policy Framework. In Finland, Greece and Poland no dedicated policies on green infrastructure exist and the most concrete references to the concept can be found in specific projects implemented at regional or local levels.

The 'dual focus' of green infrastructure on biodiversity and ecosystem services still needs to be integrated into policies both at national, regional and local levels. In most of the studied countries, the endorsement of ecosystem services takes place mainly on a strategic level. While several instruments seem to provide flexibility to address ecosystem services (e.g. by referring to supporting good environmental status and/or preventing environmental harm) only a few explicitly target both biodiversity and ecosystem services. Furthermore, those instruments that include a direct reference to ecosystem services are mainly focused on preventing negative impacts of activities on ecosystem services, rather than specifically recognising different economic, social and/or cultural benefits of these services and pro-actively supporting them. The importance of nature in supporting ecotourism, recreation, education and research (i.e. so called cultural services) is often acknowledged. However, this acknowledgement is rarely followed by dedicated legislative instruments. Furthermore, synergies between different sectoral policies, plans and measures are often not accounted for, such as the maintenance of wetlands for avoiding habitat loss, improving water quality and mitigating climate change.

The most relevant EU instruments – current and future – foreseen to support green infrastructure at national level include the Nature Directives, Water Framework Directive, and agri- and forest-environment schemes under the Common Agricultural Policy (CAP). In addition, EU Regional Policy (supported by European Regional Development Fund – ERDF) is envisaged to play an important role in the future EU policy on green infrastructure.

As regards the current level of integration of scale-related aspects in green infrastructure, our assessment indicates that spatial and functional requirements for conservation are the most commonly addressed scale-related aspects at national level. There are, however, debates on how well the existing ecological corridors and networks work in practice to support functional connectivity (e.g. France). As for jurisdictional scale, cooperation between the EU, national, regional and local levels is often taken into consideration in the design of policy instruments, for example by establishing dedicated responsibilities and identifying opportunities for cooperation between jurisdictions and policy sectors. However, shortcomings, such as a suboptimal information flow from local to EU level and different policy sectors adopting non-synergistic goals, are identified in this cooperation in practice (e.g., Paloniemi et al. 2012, Paloniemi et al. submitted).

Societal scale is also addressed, usually through public consultation in the planning stage of implementation. The involvement of stakeholders is, however, often limited to consultation only instead of adopting more cooperative approaches. In England, an interesting trend can be observed towards the devolution of environmental responsibilities and shifts to a networked governance model that incorporates more NGOs and civil society institutions, thus increasing the sensitivity to societal scale.

As at the EU level, conservation needs related to temporal scale appear to be less explicitly covered with only a few instruments, such as the Greek marine strategy, providing clear references to the need for investigating and taking into account temporal aspects related to the dispersal and life cycle of marine biodiversity. The same appears to be true with cross-scale linkages, for example with limited consideration given to link ecological dynamics of conservation with understanding of the social systems (e.g. roles and interactions between stakeholders) in the long-run. Furthermore, only few existing instruments were found that aim to address several levels of spatial scale and adopt multi-species approaches.

Conclusions

According to the assessment carried out by Kettunen et al. (2012), in order to effectively address the 2020 biodiversity goals the EU policy framework would need to consist of instruments that more systematically enable targeting different levels of spatial and jurisdictional scales – from local to the European and global level – while capturing the wide range of functional relationships between different levels. Furthermore, the need to consider issues related to the societal scale, such as the roles, needs and responsibilities of different public and private actors, is constantly increasing. In addition, it will be crucial to ensure that the different policy instruments are able to detect and adapt to changes in time, e.g. integrating impacts of climate change.

Our results indicate that an EU-wide policy framework for green infrastructure can be considered as a promising, new concept supporting the integration of scale-related considerations into EU biodiversity policy. However, in order to address scale-related complications, more comprehensive and explicit considerations of scales need to be integrated into the framework. References to different scale-related aspects of conservation, including ecological, jurisdictional and societal scales, would need to be made more systematic and explicit. For example, the framework should cover aspects related to the quality and functioning of broader ecosystems and their services, including scale-related requirements for their conservation. If possible, the changes in conservation needs over time should also be addressed. The future framework should also provide a strong basis for the integration of relevant institutions and actors into the policy processes at all levels of EU decision-making.

Strategic frameworks alone will not be able to improve the scale-sensitivity of EU policy: the real success of green infrastructure depends on effective tools for implementation, especially mechanisms for mainstreaming the concept into relevant sectoral policies including polices on agriculture, fisheries and rural and regional development. To support this, the EU could encourage and facilitate the implementation of green infrastructure initiatives into different sectors, for example, by providing guidance, creating funding opportunities and promoting research and sharing of best practices. Reforming existing EU policy instruments to include elements supporting the creation and maintenance of green infrastructure is also needed. The development of new dedicated instruments addressing, for example, environmental planning or maintenance and restoration of ecosystem services within different sectors could be considered. Finally, all interventions mentioned above need to build on a solid understanding of the status of and trends in biodiversity conservation and the maintenance and availability of ecosystem services, including interlinkages between the two (e.g., see Marsh et al. 2014, Pe'er et al. 2014 this book).

Within the EU-wide framework, a significant part of the concrete implementation of green infrastructure policy will take place through initiatives carried out at regional and local levels. Consequently, spatial planning is predicted to play an important role in implementing the EU policy in practice. Given the lack of direct EU competence in this area, national policy frameworks and instruments will play a central role in the implementation of green infrastructure. However, a well-designed common framework is needed to offer a more integrated and strategic approach for spatial planning within the EU, improving cooperation between EU and/or Member States and ensuring coherence of set objectives at EU, national, regional and local levels. Such a framework could also provide a useful tool for integrating scale-related requirements for conservation into land use, with a view to ensuring that actions taken at the local and regional level also yield benefits at the EU-level.

Securing funding from the EU and Member States' budgets is crucial in order to ensure green infrastructure implementation in practice (Green Infrastructure Working Group 2011d). There are various EU funding programmes (e.g. Structural and Cohesion funds, agri-environment schemes under CAP, and the LIFE Programme) that can be utilised in financing green infrastructure initiatives. With shrinking public budgets, however, new ways of mobilising investments are also needed. In this regard the subsidy reform and the elimination of environmentally harmful subsidies could play a key role in redirecting existing resources to green infrastructure. Furthermore, different economic tools such as environmental charges, fees and taxes and their potential to finance investments in green infrastructure should be considered. Green infrastructure can provide a cost-effective alternative or be complementary to 'grey' infrastructure and intensive land use changes. Consequently, rather than creating costs, financing the implementation of green infrastructure should be considered as a sustainable investment yielding a high level of benefits over time.

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Conservation strategies across spatial scales

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Introduction

Interventions to conserve biodiversity should consider scales because the patterns of biodiversity are shaped by many factors acting at different spatial and temporal scales. Despite this recognition, conservation strategies often fail to explicitly consider the scales at which different factors work. This can lead to mismatches between ecological factors shaping biodiversity patterns and conservation actions designed to counter the loss of biodiversity (Cumming et al. 2006). Mismatches occur in two ways: either broad-scale ecological processes are wrongly addressed by fine-scale policies (e.g. global climate change at the level of city governments) or fine-scale ecological processes are wrongly addressed by broad-scale policies (e.g. drying of a bog at the level of national government). Such mismatches can lead to a failure or low efficiency of the intervention, which can result in a waste of valuable conservation resources. It is thus important to understand whether and how conservation strategies depend on spatial or temporal scales (Henle et al. 2010).

In this chapter, we present results from a review (Lengyel et al. in preparation) of studies that (i) specified their spatial scale (extent) explicitly, (ii) applied any of four major conservation strategies that encompass a wide range of conservation actions around the globe and (iii) were published in one of eight leading journals in conservation biology and applied ecology (*Animal Conservation*, *Biodiversity and Conservation*, *Biological Conservation*, *Conservation Biology*, *Ecological Applications*, *Journal of Applied Ecology*, *Journal for Nature Conservation*, *Restoration Ecology*). We focused on four conservation strategies and four biodiversity levels to find commonalities in the specificity of conservation strategies and biodiversity levels to spatial scales (Figure 1).

Methods

We first searched the ISI Web of Science database for articles that contained the word "scale" in their title, abstract or key-words. We then filtered articles based on three criteria: they had to (i) report on results of applying at least one of the four conservation strategies, (ii) explicitly study one of the four levels of biodiversity, and (iii) explicitly identify the relevant scale(s) of the study. We tallied the number of studies in each combination of conservation strategy, biodiversity level and spatial scale and tested whether some associations were more frequent than others. For the identification of spatial scale, we used the scale category as specified in the original articles.

Patterns in scale-related conservation

A total of 233 studies, published between 1993 and 2011, met our search and selection criteria. We identified ten scale categories ranging from the local to the global scale (local, local/landscape, landscape, landscape/regional, regional, regional/national, national, supranational, continental and global) based on the authors' designation. The number of studies increased considerably with time, especially since the year 2000. Studies conducted at the regional and local/landscape scales were most frequent. Conservation planning was the most frequent strategy, followed by habitat management, restoration, and species-based protection. The majority of studies were from terrestrial ecosystems, only 10% were from freshwater and 3% from marine ecosystems. The species level was studied most often, followed by the population level, whereas studies at the genetic and ecosystem levels (beyond habitats and communities) were rare.

When biodiversity levels were considered, we found that conservation planning was most frequent at the ecosystem level as this activity typically deals with protected areas. Habitat management and restoration were both most frequent at the species level as they often study effects of interventions on species, thus species richness or diversity was of concern. Species-based approaches were mostly associated with population level and with species status. Particular conservation strategies showed clear associations to certain biodiversity levels, supporting the view that strategies are more or less specific to biodiversity level.

When spatial scale was considered, we found that conservation planning was most often associated with regional scales, followed by the mixed local/landscape scale, although national and supranational examples were also found. Habitat management was linked mostly to the local/landscape scale, then to the landscape and the local scales separately. Restoration activities were the most confined as they were almost exclusively local in spatial extent. Species-based protection activities were rather evenly distributed across spatial scales compared to the other strategies. A statistical comparison confirmed that there was a significant deviation from an equal distribution of strategies along the spatial scale categories ($\chi^2 = 80.41$, df = 9, $P < 0.0001$), confirming that conservation strategies show specificity to certain scale categories.

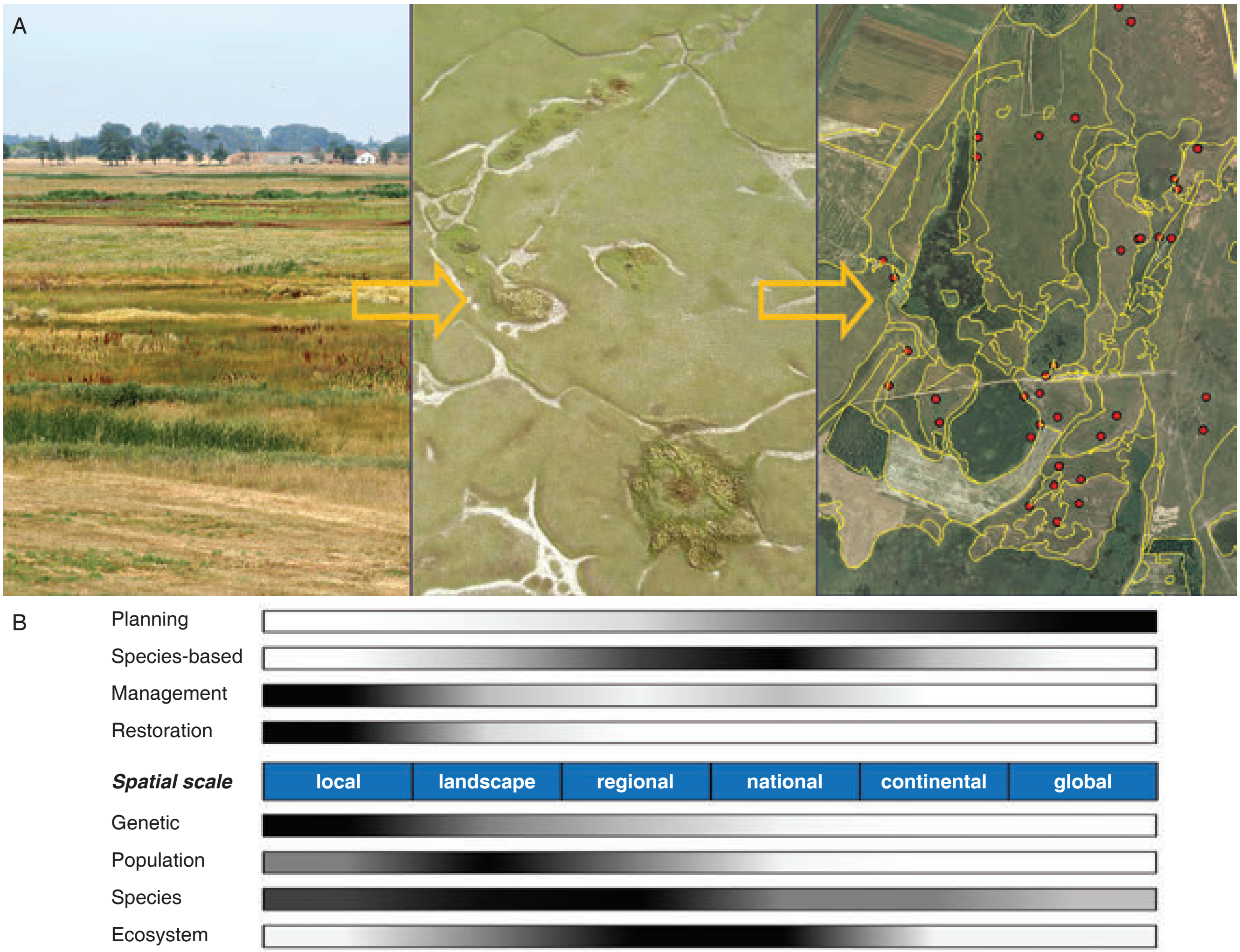

Figure 1. An example for local, local/landscape and landscape scale patterns in vegetation diversity, the conservation of which requires different strategies (A), and schematic scale-dependence of conservation strategies (above scale bar) and biodiversity levels (below scale bar), where darker colour indicates higher relevance (B). B is adapted from Lengyel et al. (under revision).

Scale issues in conservation strategies

Conservation planning

Conservation planning was defined as any strategy that aims at the spatial allocation of conservation effort. It includes approaches such as reserve design, management plans, protected area networks, hotspots analysis, scenario modelling for future changes, threat analysis for regions, and surrogate or higher-taxon approaches. Systematic conservation planning (SCP), for example, applies systematic principles and quantitative algorithms to identify areas of high conservation value and to find optimal networks of protected areas that meet pre-defined conservation targets at pre-defined costs (Margules and Pressey 2000). Current methods are able to incorporate habitat information, area effects, connectivity patterns and economic measures, such as opportunity costs and constraints (Moilanen et al. 2009). Although a handful of studies showed that the outcome of SCP depends greatly on the spatial scale chosen (both in extent and the scale at which information is aggregated), we know little about temporal aspects, i.e. the dynamics of colonization and extinction. Because of the dependence of SCP on spatial scale, any such exercise should carefully evaluate multiple scales to optimize conservation actions across spatial scales.

Species-based protection

Species-based approaches, such as Red List status assessments, threat and vulnerability assessments for species, or population viability analysis, are important traditional tools as they inform about conservation needs in a straightforward way. Our survey suggests that the number of species-based approaches has generally increased, although we did observe a slight recent (2009-2010) decline. Current scale-related efforts in this area include the integration of scale-explicit approaches, such as species distribution modelling or large-scale biodiversity databases with species-based methods to identify hotspots of threatened species. Another example is scale-explicit population viability analyses (PVAs) that typically use a local (subpopulation) and a landscape or regional (metapopulation) scale to estimate species persistence and apply upscaling of species demography from local to larger scales. Such spatially explicit PVAs can also be used in conservation planning, for example, to identify areas in which the survival of subpopulations can be increased by management.

Habitat management

Habitat management includes activities related to land use and classic management (e.g. grazing in grasslands) but also incorporates agri-environmental schemes and adaptive ecosystem management. Habitat management started out at the local scales, and increasingly extended to management of several species and natural habitat types or entire ecosystems, leading to the concept of adaptive ecosystem management, which is typically conducted at large spatial and temporal scales (Groom et al. 2006). Nevertheless, the majority of management actions still takes place at the local or landscape scales. Recent studies of the effects of agri-environmental subsidy systems regularly combine these two scales, assuming that the impact of local management will depend on the neighbouring landscape. For example, management will likely be more successful in landscapes rich in natural habitat types than in landscapes rich in artificial elements.

Habitat restoration

Restoration activities are actions to enable the recovery of degraded, damaged or destroyed habitats or ecosystems. Small areas or short time periods available for the restoration often limit the scales at which it can be carried out. In many restorations, socio-economic constraints play a large role in these limitations. Small projects usually lack enough funding to establish and operate a monitoring system, and, as a result, many restoration projects are not monitored properly or not monitored at all beyond the end of the project. A long-term effect of this is that we know little on the effectiveness of restoration at moderate to large spatial and temporal scales.

Recent theoretical advances in restoration ecology, however, may remedy this situation. A central question in restoration is how local communities build up from landscape-scale or regional pools of species through ecological filters. Studies of these questions ought to look beyond the local scale if they are to quantify regional species pools and mechanisms operating at scales larger than the local. Our survey attested that the extension of restoration studies to larger scales has already started.

Trends, gaps and recommendations

Our results suggest that the importance of spatial scale has been increasingly recognized in conservation research as the number of scale-explicit studies has been rising since 2000. The trends also show that this increase is similar for each conservation strategy, i.e., that scale-specificity has grown to similar extents in the various subdisciplines. Our survey also detected important differences from our expectations. For example, the regional scale was the most important in two of four conservation strategies. This may be partly because the "regional" scale is probably the broadest of the scale categories identified. Conservation planning and species-based protection were most frequent at the regional scale, followed closely by the local/landscape scale, whereas habitat management and restoration actions were most frequent at the local and local/landscape scales. Interestingly, there was no such clear relationship between spatial scale and biodiversity level such that higher levels of biodiversity did not automatically mean larger spatial scales of study.

Some of the gaps identified in our survey were the absence of large-scale approaches in habitat management and restoration. This gap may exist partly because many of the management and restoration projects (e.g. under the EU LIFE-Nature programme) are not reported in the primary conservation literature surveyed or are reported in the "grey literature". The low number of studies found from freshwater and marine ecosystems could be explained that many such studies are published in journals not covered by our review. There were only a few studies relating habitat management or restoration to the genetic level of biodiversity or to ecosystem services (Dixon 2009). Restoration and classic management studies rarely apply multi-scale approaches although this has become quite a custom in studies of agri-environmental schemes. Finally, we found very few examples of the links between spatial scale on one side and either conservation genetics or freshwater systems on the other, although it may be that we did not cover journals that publish such studies.

A logical recommendation from our survey is to encourage conservationists to approach problems at multiple scales. Just as cell biologists use different magnifications when looking at a cell organelle through a miscroscope, conservation biologists should also use a multi-scale approach to gain a better understanding of the system and the actions designed to benefit it (du Toit 2010). There are several good examples of the multiscale approach for habitat management and conservation planning. Studies that defy the conventional scale specificity of conservation strategies outlined here would be particularly fruitful because they would help in evaluating whether or not researchers have consciously or unconsciously settled on the scales they believe is "right" for each ecological process. Adaptive ecosystem management is one such example where management is optimized over large spatial and temporal scales. Finally, rarely studied combinations of biodiversity level and conservation strategy could provide valuable new insight into the operation of strategies at unconventional scales. For example, studies applying conservation planning to benefit ecosystem services or genetic diversity or studies applying restoration to increase population viability would greatly add to our understanding of scale-specificity of conservation strategies.

Finally, because conservation actions cannot be separated from their socio-economic context, it is highly important to increase awareness of scale issues among conservation managers, stakeholders, policy makers and the general public. One way of doing this would be to incorporate costs into conservation decision-making, for which the approaches of systematic conservation planning are highly appropriate. Another possible way is to increase the visibility of the cross-scale linkages of the impacts of policy instruments (Henle et al. 2013), e.g. the EU Birds and Habitats Directives in protecting European biodiversity and to focus on conservation successes instead of failures at different spatial and temporal scales (Sodhi et al. 2011).

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Biodiversity monitoring and policy instruments: Trends, gaps and new developments

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Introduction

The monitoring of species and habitats is essential to assess whether biological diversity is increasing, decreasing or stable. A central issue in monitoring is whether the methods used to collect and analyze data are adequate to detect the trends in biodiversity in a scientifically sound manner. Despite a recent surge of interest in the theory and practice of biodiversity monitoring (e.g. Lengyel et al. 2008, Marsh and Trenham 2008, Schmeller et al. 2012), the links between policies governing monitoring and the way monitoring is actually conducted have remained unexplored. This is particularly interesting because monitoring programmes guided by policy instruments at different scales (European, national, sectoral or regional) and implemented by different actors (Schmeller et al. 2009) may differ in their ability to detect trends in biodiversity. An evaluation of biodiversity monitoring practices in light of the policies shaping them can inform us about the suitability of policies and programmes to actually achieve monitoring goals, i.e., how well policies guide and support programmes to actually detect trends in biodiversity in a scientifically appropriate manner.

This study had two aims. The first goal was to assess the effectiveness and limitations of monitoring programmes to detect the status and trends in biodiversity in light of the policy instruments that guide monitoring. This is important because certain policy measures may be more effective than others at initiating monitoring activities that are well founded scientifically. Here we evaluated how monitoring practices reflect recent scientific advances by focusing on six EU Member States (Finland FI, France FR, Greece GR, Hungary HU, Poland PL, United Kingdom UK) that provide a geographical cross-section and for which metadata on monitoring practices were available from the EuMon database of monitoring schemes (http://eumon.ckff.si) (Henle et al. 2010). We used three measures in this evaluation, which estimate the ability of monitoring programmes to detect trends in biodiversity. We assessed how advanced the methods of data collection are by scoring various aspects of sampling (sampling design). We further estimated the effort allocated to data collection by an index for sampling effort. Finally, we recorded how data are analyzed to 'translate' them to the language of decision-makers and the general public (level of data analysis). We then linked monitoring programmes to policy instruments and evaluated whether the three measures differed either by policy instrument, type of policy, administrative level (European/national) or by geographical scope of monitoring.

The second goal was to assess the potential use of methods newly developed in the SCALES programme in improving monitoring practices. Here we reviewed 24 research reports incorporating 43 studies conducted in the SCALES project to assess whether and how recent scientific advancements can be used to improve monitoring policies and practices. This chapter summarises results from a study of monitoring and policy instruments.

Results: Monitoring schemes in light of policy instruments

We found that species monitoring programmes guided by different policy instruments were generally similar to one another with regard to their sampling design and data analysis. Although sampling effort varied among different policy instruments, the differences were not related to policy types or administrative level (European/national). Species monitoring programmes, however, differed between countries because design was more advanced in Hungary and effort was higher in the United Kingdom than in other countries. Programmes also varied by their geographical scope because programmes with larger geographical scope had more advanced designs and higher effort than smaller-scale programmes (Figure 1). Advanced statistics were applied in 26% of the programmes, mainly in programmes guided by national legislation on protected areas. Lack of data analysis was common in programmes guided by 'other' national policy instruments.

In habitat monitoring, programmes guided by European policies generally had more advanced design and higher effort than those guided by national policies. In addition, programmes governed by French and UK landscape laws and by national park laws in other countries had more advanced design. Likewise, programmes governed by the UK landscape and French and Polish conservation laws had higher effort than programmes guided by other policy instruments. Habitat monitoring programmes in Hungary

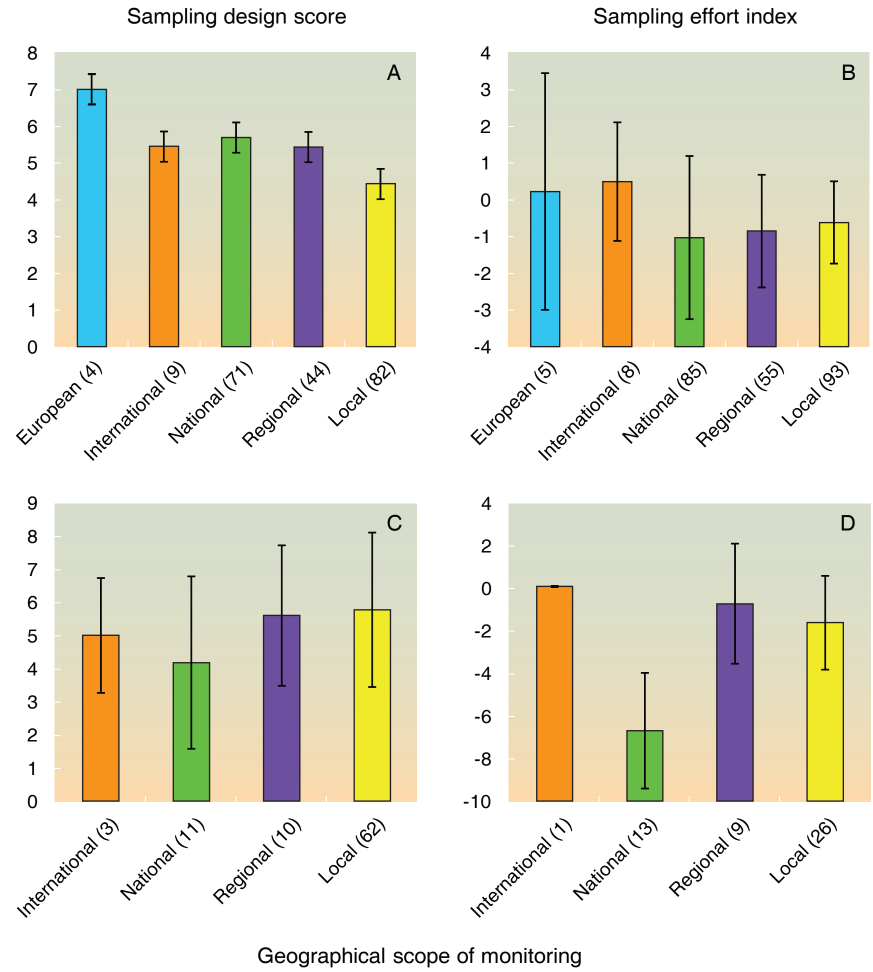

Figure 1. An example of the assessment of programmes monitoring species (A-B) and habitats (C-D) at different geographical scopes based on their sampling design (left-hand column) and sampling effort (right-hand column). The number of programmes in each category is shown in parentheses.

performed lower both in design and effort than those in other countries. Habitat monitoring programmes did not differ in design by geographical scope, although regional and local programmes had higher effort than national programmes (Figure 1). Finally, advanced statistics were used only in programmes guided by the main national conservation law. Most programmes, especially those guided by European policy, analyzed data by "other statistics". The major patterns found are summarised in Table 1.

Discussion

No single policy instrument stood out as one which was always associated with good sampling design, high effort and adequate data analysis. Thus, no general programme can be recommended as the best. Programmes guided by some policy instruments performed better at one or two measures while others were better at other measures. In general, monitoring programmes guided by various policy instruments did not differ greatly in species monitoring but they did so in habitat monitoring.

The performance of monitoring programmes is influenced by both the policy instruments guiding them and the underlying process of development, discussion/negotiation and implementation. The successful implementation and improvement of monitoring schemes on different administrative levels and spatial and temporal scales will then depend on the societal background, i.e., the general awareness of biodiversity conservation and the degree at which a society values biodiversity. It is thus not surprising that the differences found here could be linked to the advancement of biodiversity conservation in the case study countries. Monitoring programmes often had more advanced design or higher sampling effort in socio-economically more developed countries that had longer traditions in biodiversity conservation (UK, France) than in the other countries (Greece, Hungary, Poland). Large-scale, European policy instruments can thus be very important in reducing this gap in monitoring and in biodiversity conservation that still exists between northern/western and southern/eastern countries within the European Union.

Recent advancements to improve monitoring

We found that recent scientific advancements can improve biodiversity monitoring in four main ways: (i) identifying priorities with regard to what species or habitats should be monitored, (ii) achieving a more optimal sampling design, (iii) finding a more optimal way of allocating sampling effort in space and time, and (iv) utilising advances in data analysis to enhance the flow of information from the data to decision-makers and the general public. We recommend the implementation of these measures in monitoring activities to further improve monitoring practices. We also recommend some of the new methods to be integrated in national and European policy instruments that guide biodiversity monitoring. The following part discusses how the different advancements (in Italics) can be used to improve monitoring practices.

Priorities in selecting species and habitats for monitoring

- To *increase the number of species and habitats monitored*, several options are available: (i) monitoring more species, (ii) monitoring common or surrogate species or habitat types if their changes reflect the changes of rarer species or habitats that are more difficult to monitor, (iii)

Table 1. A summary of measures of scientific assessment in species and habitat monitoring programmes guided by different types of policy instruments in six case study countries.

Monitoring	Type of policy	Sampling Design Score[a]	Sampling Effort Index[a]	Data analysis
Species	EU Birds Directive	intermediate	high but variable	basic
	EU Habitats Directive	intermediate	high but variable	basic
	EU Water Framework Directive	intermediate	low	basic
	National main conservation law	intermediate	intermediate, variable	basic/advanced
	National parks law	intermediate	high	basic/advanced
	National, other protected area law	intermediate	low, variable	advanced/basic
	National hunting/forestry law	intermediate	low, variable	basic/advanced
	National other	intermediate	high	basic
Habitat	EU Birds Directive	high	N/A[b]	N/A
	EU Habitats Directive	low	high	basic/other
	EU Water Framework Directive	high	intermediate	N/A
	National main conservation law	intermediate, variable	low, variable	advanced/other
	National parks law	high	high	N/A
	National, other protected area law	intermediate, variable	high	N/A
	National, hunting/forestry law	intermediate	intermediate, variable	basic/other
	National other	low	high	N/A

[a] "*intermediate*" means that values did not differ from the average for all policy instruments, and "*high*" and "*low*" means values consistently higher or lower than the average values, respectively; "*variable*" was when the standard deviation was large relative to the group mean.
[b] N/A – data not available or sample size too low to allow categorization.

monitoring many species or habitat types at the same time by using low-intensity data collection methods, such as species lists followed by list length analysis to quantify the detectability of species (Roberts et al. 2007).

- The knowledge of *dispersal ability and distances* can help in (i) prioritising species and survey units for monitoring because species with limited dispersal often have higher risks of extinction and should be monitored more closely, (ii) allocating effort to species that represent either very low and very high abilities of dispersal to predict changes in their habitat or (iii) allocating effort to those species and habitats that are more likely to be affected by global change.
- The knowledge of *sex-related differences in dispersal* informs us about sex-related differences in the use of habitats and about different detectability of individuals. For example, if females of a species are more secretive, they will be more difficult to monitor than males. Sex-related differences in dispersal can also lead to a skewed sex ratio, which is often a sign of low population size and high extinction risk. All of this information should be considered in the design of species monitoring.
- Quantifying the *responsibility of administrative units in conservation* is important to allocate monitoring to those countries/regions/localities which bear the overwhelming responsibility of conserving a given species or habitat.
- Finally, different policy instruments can be served simultaneously by a recently developed temporal and spatial prioritisation programme that is based on common priorities in policy instruments (Henle et al. 2013).

More optimal sampling design

- *Modelling the geographical range of species across scales* can provide information on the areas that are poorly known and which thus should be monitored more closely (Rocchini et al. 2011). The modelling of ranges can also predict where a species of conservation importance is likely to occur (Elith and Leathwick 2009), which may also help in identifying areas that should be monitored.
- Understanding *beta diversity*, or the degree to which the flora and fauna are similar or different over larger geographic regions, can help in identifying the proper size of monitoring units. If the local floras or faunas are unique, smaller survey units are appropriate, whereas if the local floras or faunas are largely similar, larger survey units will suffice.
- The determination of an optimal sampling design and effort should be done at the planning phase of monitoring programmes. The aim of optimisation is to find a balance between the need to monitor many species over large areas and long time periods and the constraints regarding the resources available for monitoring. The underlying principle should be to design monitoring programmes that are really able to detect changes in species and habitats over time, otherwise, monitoring will lead to a waste of resources. One way to help this step is to conduct *prospective (a priori) power analysis*, which can give quantitative information regarding the number of samples, sampling sites, or the necessary length of time that is necessary to detect changes in species and habitats. Such power analysis should be part of the design phase for any monitoring programme. Several online and freely downloadable tools are available for this purpose (Nielsen et al. 2009).

More optimal sampling effort in space and time

- The knowledge of *how the habitat requirements of species change with scale* (Altmoos and Henle 2010) can be used to give priority to species or habitat types that are more sensitive to changes of scale. For example, for species that select their habitats at large scales (imagine an eagle), surveys of low local intensity but extending over large areas are necessary. In contrast, for species selecting habitats at lower scales (imagine a sparrow), higher local intensity but smaller survey areas may be necessary. Such a distinction can help to more optimally divide monitoring effort between larger-scale species and smaller-scale species.
- In some cases, species numbers or populations can change in similar ways over large areas and can simultaneously decrease and go locally extinct or can increase and colonise local habitats. When such *spatial autocorrelation in extinction and colonization* happens, it can be used to identify synchronously changing spatial units of populations. Monitoring can then be optimised because if such correlated changes are strong, survey effort may be reduced to one or a few synchronously changing unit and extended over larger areas (Giraud et al. 2013).
- *Upscaling* is the process when changes in a species over a large area (i.e., a country) are inferred from local observations (i.e., from cities). A comparison of upscaling methods showed that the number of sampling sites is more fundamental in estimating the number of species over a large area than the number of samples collected at one site. This result implies that *local survey effort may be reduced*, without committing large errors in estimating the number of species and the effort saved can be allocated to other areas or habitats. Sampling intensity may thus be sacrificed for monitoring larger areas. For example, collecting information on the presence of a species in an area is much easier than to reasonably estimate the number of individuals (abundance) of that species in the area. The effort or resources freed up by the easier local survey can then be allocated or extended to areas or time periods that were previously not monitored.
- *Reduced-effort monitoring* can be implemented in several ways. Pilot surveys can evaluate whether possible sites for monitoring vary in their potential to provide meaningful information (identification of information 'peaks' vs. 'valleys'). In case they vary, sites representing information peaks can be sampled intensively, whereas others (valleys) can be sampled with a low intensity. Alternatively, it is possible to balance the monitoring in space and time if not all sites need to be surveyed every year. For example, in a split panel design, some sites are revisited, some are newly visited every year, and the proportion of new vs. revisits can be adjusted based on previous knowledge or pilot study.

Optimising data analysis

- *Downscaling* is the process when changes in a species or population at local scales are inferred from changes in a larger area. Methods of *upscaling and downscaling* are now advanced enough to provide reasonable estimates on population sizes and can now be considered for use in monitoring. For example, downscaling can be used to identify incompletely surveyed areas, which is useful when monitoring has gaps. Downscaling is also promising for the integration of data collected at different spatial scales (e.g. some data from local monitoring and some from landscape-level or regional monitoring). Bringing such data from various sources in a top-down approach to the same scale can provide joint trend estimates at finer resolution for large areas. *Upscaling* is also relevant in monitoring, where one of the greatest challenges is how to draw conclusions at large scales from local-scale trend estimates (bottom-up approach). Recent upscaling methods use small-scale species-area relationships to predict large-scale species richness with acceptable accuracy, therefore, their use is recommended in the integration of monitoring.
- For many species, the geographical range overestimates the area that populations of the species actually occupy in reality because not all areas within the range are suitable as habitats for the species. Thus, concepts, such as the area of occupancy, and metrics, such as the fractal dimension of the geographical range, have been used to pinpoint/refine the areas where species occur in reality. A recent study of the *scale-dependence of area of occupancy and fractal dimension* of the geographical range (Clobert et al. 2012) showed that occupancy (whether a species is present in a pre-defined survey unit) can provide basic data to evaluate (i) whether a population is declining or increasing and (ii) to evaluate the relationship between scale and area of occupancy. Such basic information can be used to judge tendencies in populations or species, which can provide important additional information to traditional monitoring.
- Habitat monitoring is inherently spatial, i.e., it measures the extent (area) or spatial configuration of habitats. *Landscape metrics* are thus obvious tools to measure changes in the size, shape, configuration and structural connectivity of habitats within a landscape over time. Additional knowledge of *functional connectivity* can provide information on whether the structural connectivity actually functions in reality, i.e., whether individuals use the corridors between neighbouring habitat patches. Information on functional connectivity may also help in identifying thresholds (e.g. minimum number of individuals getting from one patch to the other) that are necessary to maintain popula-

tions. Populations close to this threshold value then should enjoy priority in monitoring.

- *New analytical tools* are now available which are capable of involving more than one scale in the analysis. Hierarchical Bayesian multi-scale occupancy models are just one example, which also has the advantage that it allows the inclusion of previous knowledge (e.g. from small-scale monitoring) to be included, which makes the results more robust. In any case, data collected in the field should be analyzed, ideally by using more sophisticated, readily available methods, such as general linear mixed-effects models, to obtain more information from data already collected. The gaps in spatial or temporal coverage can also be filled in by analytical tools, such as upscaling/downscaling methods or the TRIM software (http://www.ebcc.info/trim.html).

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Biodiversity monitoring and EU policy

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Introduction

Effective policy-making requires a well-functioning system of monitoring if its objectives are to be met (Balmford et al. 2005). Although the EU has a reasonably robust regulatory framework on biodiversity – at the heart of which lie the Birds and Habitats Directives – in order for these policies to be effective, policy-makers need to be able to assess the progress towards these objectives (EEA 2012). This in turn requires the organisation of biodiversity data in a form that can be easily represented and understood at a European scale.

The ability of the EU to influence the type of data collected across its territory is limited and the monitoring of progress towards EU-level biodiversity goals is therefore heavily dependent on the frameworks already established within Member States and the capacity of the monitoring institutions to carry out these functions over the long-term (Donald et al. 2007, EuMon 2011). With the adoption of the EU 2020 Biodiversity Strategy, with six dedicated targets aimed at halting the loss of biodiversity and restoring ecosystems where possible by 2020 (EC 2011), it remains uncertain how progress towards these priorities will be assessed. This is particularly the case for a range of new, dedicated objectives related to protecting the functioning of broader ecosystems and delivery of ecosystem services.

The monitoring of biodiversity across the EU is a combination of state- and NGO-funded schemes and carried out by varying proportions of volunteers and professionals. The assessment of the data at the EU level is complicated by the disparity in quality of the data provided from across the Member States. In general, biodiversity monitoring is most comprehensive in northern and western Member States, whilst there is a significant lack of data from southern Member States, particularly in the Mediterranean region. Data are also lacking from several eastern European countries, which are very rich in biodiversity but where few or insufficient systematic monitoring programmes exist (EuMon 2010, Bell et al. 2011).

Within the SCALES project, a case study was carried out to explore the extent to which the existing national monitoring institutions are capable of supporting EU policy requirements in light of the EU 2020 targets. More specifically, this involved examining the motivations and manner in which monitoring is carried out, identifying the barriers to responding to policy changes, what is the current state of affairs with respect to emerging priorities and what are the opportunities for improving biodiversity monitoring relevance for policy making.

Approach

The study undertook a literature review to assess current needs for biodiversity monitoring in the EU and to identify remaining gaps. The study then investigated the issues related to meeting policy needs in more detail using an in-depth case study of the UK and three general case studies (on Finland, Greece and Poland), to provide an overview of the likely issues related to monitoring across EU Member States. The research for the UK was based on semi-structured interviews with 22 organisations representing monitoring institutions, NGOs and statutory agencies responsible for data reporting. A further seven interviews were undertaken with representatives of institutions in Member States beyond the UK: four organisations in Greece; two in Poland, and one in Finland, representing in all cases the official national authority responsible for reporting at the EU level, and where possible a representative of a related NGO and/or academic institution.

The analysis and recommendations of this study therefore reflect the views of those involved and the results of the literature review. We present here the issues that are likely to effect the relevance of biodiversity monitoring for policy making and the ideas identified for improvement based on these sources of information.

Motivations and manner in which monitoring is carried out

Drivers

In the UK, the most important priority for biodiversity monitoring amongst most NGOs, who are responsible for co-ordinating the majority of monitoring effort in the country, is the assessment of the conservation status of species of concern, while policy requirements (particularly at the EU level) were considered to be a low priority. This is expected to be a consequence of the establishment of the monitoring programs pre-dating the European policy requirements for the data. In addition, many of the organisations rely on memberships and donations to fund their activities, meaning the concerns of these stakeholders need to be integral to their goals.

Larger organisations, such as the Royal Society for the Protection of Birds (RSPB) also indicate that site-based monitoring of species listed under the Birds Directive is also a significant factor in its work. For the statutory agencies, EU policy needs are an important driver of their activities and they have established monitoring and research activities to specifically address the EU Nature Directives. In Poland and Greece, on the other hand, EU policy requirements appear to have been a very important driver of the establishment of monitoring activities in the majority of organisations (including NGOs), where many monitoring schemes were set up in direct response to the EU Nature Directives and the Water Framework Directive (in the case of Greece).

Who carries out monitoring

Biodiversity monitoring in the UK and Finland depends significantly on the contributions of volunteers, in contrast to Poland and Greece where it is more extensively carried out by professionals. The involvement of volunteers in both the UK and Finland is founded on a long tradition of amateur naturalism, developed over a long period of time by NGOs with support from public money, which does not exist to the same extent in the other Member States included within this study. The involvement of volunteers in these countries contributes very substantially to the overall monitoring effort and results in greater coverage than could otherwise have been achieved by professionals alone. Nevertheless, there are indications that volunteering in Greece may be growing.

Barriers of responsiveness to EU policy

A lack of funding and necessary expertise was identified as the principle barriers to responding to EU policy requirements and to increasing the scope and coverage of monitoring programmes across the Member States. A lack of funding constrains many organisations in their abilities to coordinate volunteers, address data gaps not considered national conservation priorities, and develop new programmes to cover emerging priorities. Constraints on resources limit potentially beneficial collaboration between organisations, such as for the development of methodologies, training of volunteers and in the analysis of the data. Butterfly Conservation indicated that a small amount of funding from the EU could allow Member States to capitalise cost-effectively on citizens' interest in monitoring biodiversity within their borders.

A number of smaller organisations in the UK stated that lack of capacity meant that they were only able to invest in responding to national policy requirements and did not have the time to keep up to date with those emanating from the EU. Funding is seen also as an important limitation in developing monitoring in eastern Member States. Networks of expertise were identified as significant sources of support in improving and expanding biodiversity monitoring, providing opportunities to share best practice (see also Gregory et al. 2005).

Lacking too, according to WWF Greece, is the necessary context in which citizens can be educated about and encouraged to participate in biodiversity monitoring; as a consequence, voluntary biodiversity monitoring schemes are currently not common practice.

The need to ensure comparability with long-term data-sets constrains the adaptation of existing monitoring programmes to meet policy needs. These historical constraints imposed upon monitoring schemes in the UK, which have rarely been developed with policy questions in mind, also account for the disconnect between biodiversity data and policy requirements. Also, the funding models of monitoring institutions play an important part in their ability to provide monitoring data for policy, including those funded by government, memberships or commercial operations. In some cases, institutions or NGOs have an official remit or set of objectives, which frequently refers to species or habitat conservation rather than, for example, ecosystem service provision. This means that establishing the benefit of broader ecosystem service health on species protection is required as well as clarifying the role of habitats and species in ecosystem service provision.

Responding to emerging priorities and the EU biodiversity 2020 Strategy

The EU Biodiversity 2020 strategy will require Member States to halt the loss of ecosystem services, and to reduce the pressures associated with Invasive Alien Species (IAS). There were differences in how the Member States assessed the need to monitor emerging priorities and their capacity to carry this out. For instance, in the UK, the need to monitor ecosystem services is recognised but there is a great deal of uncertainty about methodologies to use. A number of initiatives in the UK have been established at a site level to improve and monitor ecosystem service provision. Nevertheless, monitoring of these services is very site specific and therefore difficult to extrapolate from one area to the next. Trends will have to be based on generic criteria that can be adapted from existing data flows. Capacity to track invasive alien species (IAS) through existing monitoring programmes in the UK is good but improved co-ordination between institutions and a better early warning system are required. Awareness of the scheme remains low amongst the public who tend to report sightings to a separate agency.

(photo: Islay Nature History Trust)

Croasdale meadow showing a wildflower-rich sward after three years of traditional hay meadow management, funded by United Utilities (photo: Anderson and Ross 2011).

In the other Member States, there was a lower degree of effort on monitoring emerging priorities. In Finland, capacity to monitor both IAS and ecosystem services were seen as future priorities for development, and although systems for reporting IAS are now beginning to come into effect, it is too early to assess progress. Resources for addressing the new monitoring needs were judged to be scarce and therefore it was considered that EU level legislation (e.g. a Directive addressing IAS) would likely play a key role by prompting Member States to allocate further funds to establish adequate monitoring systems. Similarly in Poland and Greece, it was stated that the scope for further investment in monitoring schemes was limited and existing data and monitoring needs should be adapted to provide information on emerging priorities.

Conclusions

For the present EU policy regime, certain Member States, particularly in northern and western Europe, have established effective monitoring schemes, based to a large degree on volunteers. In the UK, for instance, the strength and longevity of the volunteer effort has been actively developed by NGOs with support from public funds. In contrast, southern and eastern Member States often have an underdeveloped monitoring regime with a reliance on professionals and a relative absence of volunteers. Therefore, a significant challenge is the sharing of knowledge concerning the development and running of volunteer-led monitoring programs in these Member States (Bell et al. 2011), as well as ensuring increased public funding to promote their growth.

With respect to emerging priorities, certain Member States, such as the UK, recognise the need to collect more data and have already begun to establish a baseline, for example of ecosystem service provision. In many other Member States, however, these are simply not yet the priority as basic biodiversity monitoring systems are yet to be established. Even in the UK, uncertainty exists about how trends will be generated. In addition, institutions currently involved in monitoring have an official remit which does not include responding to EU emerging priorities and therefore are reluctant to divert resources away from their core work.

Recommendations

The findings of the study result in a recommendation to provide greater support for the formation of collaborative expert networks across Europe. These burgeoning EU-wide networks have proved very effective at harmonising methodologies and deriving pan-European trends (see Gregory et al. 2005). A small amount of financial support was considered by interviewees to likely make a significant contribution towards developing these networks for less charismatic but functionally important taxonomic groups and accelerate capacity building in those Member States in which biodiversity monitoring is currently limited (e.g., eastern Europe).

Greater scientific understanding of the links between species/habitat quality and ecosystem service provision and the monitoring of those components of biodiversity – functional, structural and genetic – that are likely to have significant value for assessing ecosystem services, were identified as the most urgently required developments for providing data for ecosystem service provision (Henle et al. 2010, RUBICODE 2009). Research should provide guidance on how to interpret local specific conditions (e.g., soil type or topography) that are likely to have a very significant effect on ecosystem service provision.

There is an opportunity to engage the private sector beneficiaries of ecosystem services to increase funding for both ecosystem and biodiversity monitoring. Successful examples of user groups, for example hunting associations or water companies, have shown that these can provide very detailed and valuable information on the status of habitats, species and ecosystem service provision (e.g., Anderson and Ross 2011).

Opportunities exist to expand highly effective and reliable citizen-led biodiversity monitoring schemes across the EU (Schmeller et al. 2009). Technological advances offer opportunities to expand volunteer involvement and generate interest and engagement in the natural environment amongst the wider public ('citizen scientists'). The transfer of knowledge from networks of expertise, however, are likely to be needed to support the expansion of these schemes.

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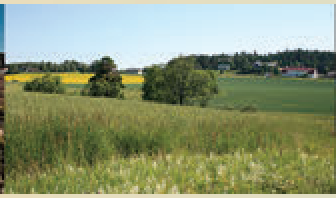
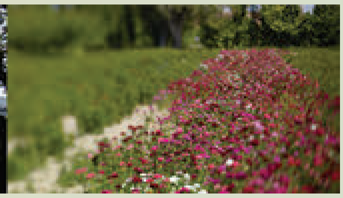
Annex: List of participating organisations

Country	Organisation	Sector
UK	Amphibian and Reptile Conservation	Third sector
UK	Bat Conservation Trust	Third sector
UK	Biological Records Centre	Third sector
UK	Botanical Society of the British Isles	Third sector
UK	British Bryological Society	Third sector
UK	British Dragonfly Society	Third sector
UK	British Lichen Society	Third sector
UK	British Trust for Ornithology	Third sector
UK	BugLife	Third sector
UK	Butterfly Conservation	Third sector
UK	Countryside Council Wales	Statutory Agency
UK	Concological Society of Great Britain and Ireland	Third sector
UK	Department of Environment and Rural Affairs	Member State
UK	Environment Agency	Statutory Agency
UK	Game and Wildlife Conservation Trust	Third sector
UK	Joint Nature Conservation Committee	Statutory Agency
UK	Natural England	Statutory Agency
UK	Plantlife UK	Third sector
UK	RSPB	Third sector
UK	Scottish Natural Heritage	Statutory Agency
UK	The Mammal Society	Third sector
UK	Wildfowls and Wetland Trust	Third sector
Greece	Ministry of the Environment	Member State
Greece	WWF Greece	Third sector
Greece	Hellenic Society	Third sector
Greece	University of Ioannina	University
Finland	SYKE	Statutory Agency
Poland	Inspectorate of Environmental Protection	Statutory Agency
Poland	Institute for Nature Conservation, Polish Academy of Science	Third sector

CHAPTER **VI**



Case studies and integration

Spatial data standardization across Europe: An exemplary tale from the SCALES project

Konstantinos Touloumis, John D. Pantis

Introduction

In recent years, growth in the availability of ecological data has been exponential and it is expected to continue at the same, if not faster, rate in the future. Thus, data sharing has become an increasingly important aspect of sound environmental management. Numerous database structures have been created to describe and store information about various environmental traits, such as topography, climate information, species traits, habitats etc. However, different organizations, countries and/or individual researchers have adopted their own unique data definitions and database structures. These differences could affect the kind of information available through these sources. In order to overcome such limitations towards accommodating data sharing across scientific communities, a data standardization process has become a necessary action every time a dataset is generated. With the term "data standardization" we refer to the establishment of an infrastructure enabling the acquisition, organisation, management, accessibility, exchange and application of data to ensure consistency and comparability across different databases. This is especially important, if not necessary, in a data warehouse environment that contains information from many sources, as well as to integrate large scientific projects, where data should be stored and presented in a way to be easily accepted and understood by a wide range of scientists from different disciplines. Without the standardization of data, no relationship can be established between the various data sources to produce results that include information from multiple datasets. The benefits of organizing and using a data standardization infrastructure are substantial and include:

- Standardized methods for acquisition, processing and management of data,

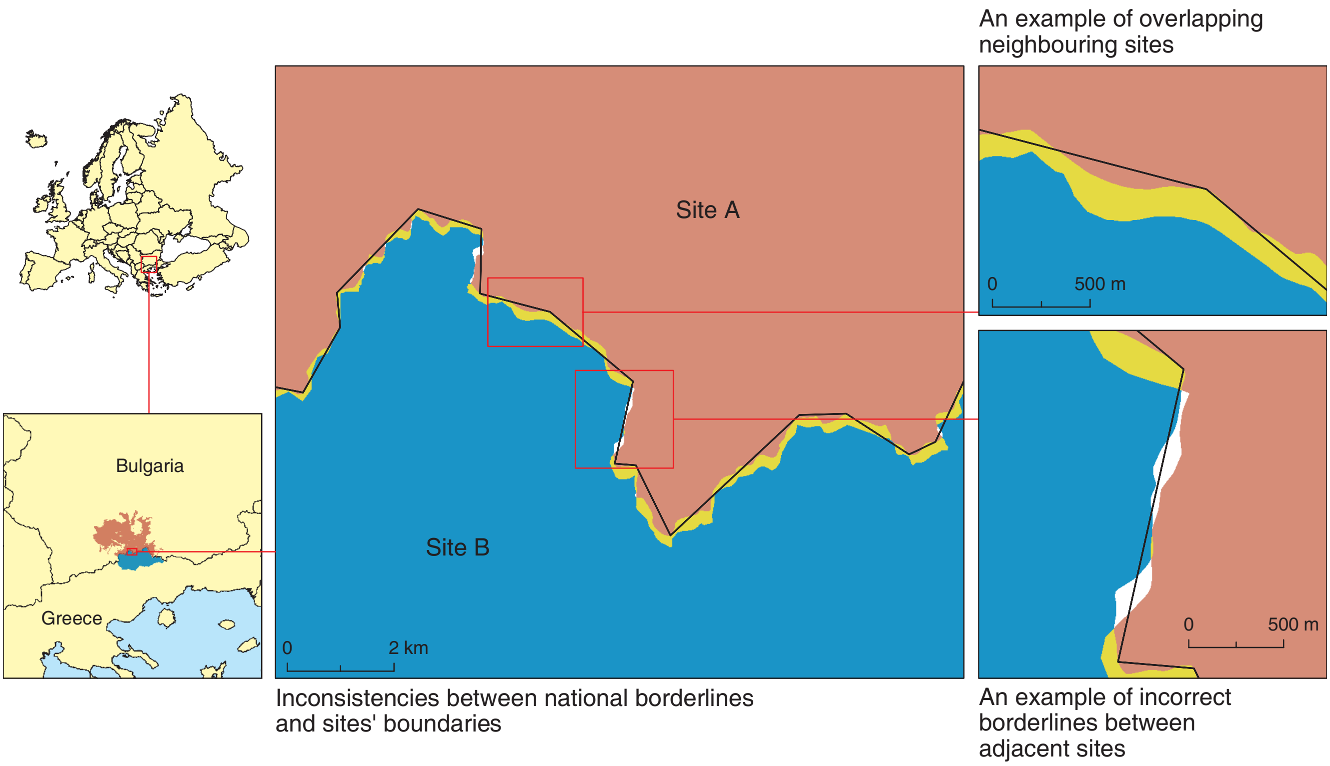


Figure 1. Typical example for technical incoherencies in GIS: digitizing problems on common borderlines. For this map we used Natura 2000 dataset (source: http://www.eea.europa.eu/data-and-maps/data/natura-4) and NUTS database (source: http://epp.eurostat.ec.europa.eu/portal/page/portal/gisco_Geographical_information_maps/popups/references/administrative_units_statistical_units_1).

- Structured and organized access to data,
- Elimination of duplication of efforts in data acquisition and management,
- Cost saving in data acquisition and management.

The matter of standardization is also important when geospatial data are concerned. The introduction of the Geographical Information System (GIS) technology in the mid 1980s as well as the broad acceptance of it in the following years, has led to the creation of numerous geospatial datasets over the past three decades. However, most of these datasets have been created independently, with various GIS and remote sensing software packages and in many cases without the use of any structured documentation system.

The lack of consistency in the development of spatial data poses numerous risks. Spatial inconsistency between data developed by different authorities and organizations is often apparent. A typical example of this is found in the digitized GIS data concerning the Natura 2000 network of protected areas. In many cases inconsistencies (e.g. incorrect borderlines for adjacent sites, double overlapping lines at neighboring sites or overlapping polygons) occur, especially at administrative borders within Member States as well as along Member State borders (Figure 1). Other possible problems related to technical issues in the development of spatial data could stem from the use of: different GIS-software, e.g., concerning data formats and their conversion, different mapping scales, projections and projection dates, different scales of topographic maps as the basis for digitisation, etc. (Ssymank 2005).

Furthermore, the absence of adequate documentation may bring about a series of difficulties. Dataset files are often misplaced and made redundant and thus cannot be revealed to the wider geospatial community, therefore limiting other researchers' access to the original dataset. It also prevents other potential users from augmenting or complementing their own datasets for various applications. The absence of information about existing datasets can lead other organizations to expend considerable time and costs in producing data that are already in existence, but at an undisclosed location (Mathys and Kamel Boulos 2011).

The Inspire Directive

The INSPIRE Directive (INfrastructure for SPatial InfoRmation in Europe, Directive 2007/2/EC) was introduced in May 2007. Its main goal was to tackle the problems of data harmonization, of the use of standards and access to spatial information in general via the establishment of an infrastructure for spatial information in Europe. This infrastructure was rated as essential in order to support Community and Member States' environmental policies as well as activities, which may have an impact on the environment.

INSPIRE is based on six key principles (Vandenbroucke 2005):

- Data should be collected once and maintained at a level where data collection can be done most effectively.
- It must be possible to combine spatial information from different sources across Europe and share it between many users and applications.
- It must be possible to easily identify the level of detail each dataset contains in order to be able to use detailed datasets for detailed investigations and rather general for strategic purposes.
- Geographic information needed for good governance at all levels should be abundant and widely available under conditions that do not inhibit its extensive use.
- It must be easy to find out which geographic information is available, fits the needs for a particular use and under what conditions it can be acquired and used.
- Geographic data must become easy to understand and interpret so it can be visualized within the appropriate context and demonstrated in a user-friendly way.

These principles should guarantee easy access to harmonized data. To ensure that the spatial data infrastructures of the Member States are compatible and usable in a Community and transboundary context, the Directive requires that common implementing rules are adopted in a number of specific areas (Metadata, Data Specifications, Network Services, Data and Service Sharing and Monitoring and Reporting, Directive 2007/2/EC).

Data availability and standardization

For the SCALES project, we compiled data from different sources on both European and national scale; the latter referred to five case study countries: Greece, Finland, France, Poland, and the United Kingdom. These data were both partly spatially explicit and partly non-spatial. The collection, preparation and standardization of existing data aimed to provide necessary information for cross-scale testing in relation to three major conservation needs: (1) Analyzing and ensuring coherence and ecological sufficiency of networks of protected areas, (2) Improving regional connectivity of habitats for various species dispersal distances & landscapes, and (3) Monitoring of conservation status and trends of biodiversity across scales. These datasets were standardized through an online dynamic system of data management (Goggle Docs, https://docs.google.com/spreadsheet/ccc?key=0AqgGOnBZMoVddFJuUVo5YmQ2eC1iOTU3U2FUTWQ5UUE&hl=en#gid=0), by using a spreadsheet with standards drawn from Inspire Directive, properly adjusted to satisfy the specific needs of the SCALES project (Table 1). Overall, 105 different datasets were gathered and standardized; 26 included data at continental level while the rest (79) were national level data. On the European level, data originated from several primary sources, among them the European Environment Agency (EEA) that has produced several maps of environmental data with continental coverage. These datasets were well standardized, since EEA follows Inspire Directives' standardization premises. However, biodiversity data at the continental lev-

Table 1. Structure of "Data availability" spreadsheet with a short description about each standard.

Standards	Short Description
LEVEL	EU, Bioregional, National, Regional.
DATA	A short title of the dataset.
LINK PERSON	Name of the most appropriate person or institution to provide information about the dataset.
E-MAIL	The contact e-mail of the most appropriate person to provide information about the dataset.
MAP/DATA SOURCE	The institution, web link, or research paper where the dataset is originated or/and located.
YEAR	The year(s) that data were collected.
EXTENT	The spatial range of data (EU, national, regional/local). For regional local, provide area in km^2.
MAP RESOLUTION	The resolution of a map.
LAND COVER TYPES	A field to describe the land cover types for analogous data.
COORDINATE SYSTEM	The coordination system each map follows.
COMMENTS	Any special information about the dataset.
COPYRIGHT ISSUES	The level of availability of a dataset.

el were usually prepared by individual researchers and often lacked proper standardization, a fact that posed difficulties in the comprehension and use of these data for research.

On the national level, it became apparent that data availability was significantly different among case study countries. Some information regarding topography, hydrology, land cover data, administrative boundaries and road and railway networks were available for all case study countries. Furthermore, due to the existence of common protocols like CORINE, the classification scheme used was identical in all case study countries. However, the dissimilarities were even more striking, especially regarding biodiversity data. On the one hand, northern and western countries (Finland, France and the United Kingdom) were data rich, with information on the spatial distribution of diversity and species of conservation interest and with several monitoring schemes providing information on the temporal changes in biodiversity. On the other hand, southern and eastern countries (Greece and Poland) were comparatively data poor, with fewer data sets on biodiversity patterns (spatial or temporal) and most importantly with data being sparsely distributed and not providing complete coverage of the countries.

Data format

All of the spatial data sets were available as GIS layers, which made their use and comparison much easier. However, the type of the data differed. Although most data layers were available in vector format (data types point, polyline and polygon), data layers in raster format are also available (e.g., Worldclim climate data). As a result, conversion between the two data types is often needed, a process that inevitably leads to a loss of accuracy (Congalton 1997). On the other hand, most of the biodiversity spatial data were attributes of more or less rectangular polygon cells created by regular dissection of longitude and latitude forming the frames of the printed maps. In most cases these cell frame borders do not correspond in their orientation with the cartesic coordinate system used within the grid cells. These datasets differed in cell's area from the European to the national level and also within the national level. In general, European data were coarser (for instance the European Bird Atlas has cell size 50 × 50 km), while the national equivalent datasets were of finer resolution (for instance the latest French Bird Atlas has cell size 10 × 10 km, and there are even finer scale datasets that do not cover the entire country like the French Bird monitoring scheme with a cell size of 2 × 2 km).

Another restriction we discovered is the copyright issue. While the European level data (especially the ones produced with European funding) are freely available, there are several national datasets that are of restricted access. Therefore, several datasets, especially regarding biodiversity at the national scale, are not available to all SCALES partners. However, for most of these datasets the case study country partners had access and could utilize them for the purposes of our project.

Throughout this process, it became apparent that standardization is a necessary step of data preparation and dissemination. The use of well known, widespread, and common protocols, like the one based on INSPIRE Directive could further facilitate the standardization of data, simplify the use of these data by the scientific community and thus speed up the development of environmental sciences.

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An optimal spatial sampling approach for modelling the distribution of species

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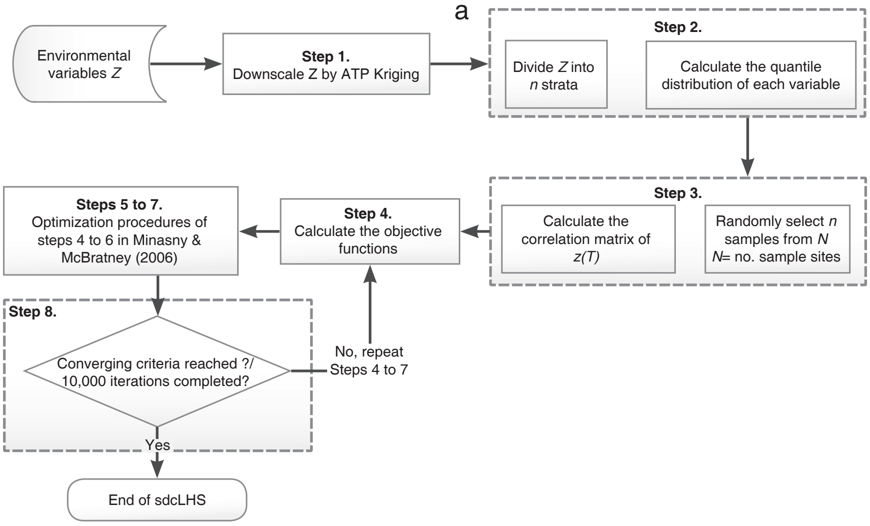
Stratified random sampling for spatial sampling

Sampling is fundamental to most ecological studies and a representative sampling design is of high importance for biodiversity monitoring. It was previously recommended that the ecological sampling design should be stratified to improve precision, accuracy, and to ensure proper spatial coverage (Gregory et al. 2004). Hence, stratified random sampling has been one of the designs frequently applied in ecological studies. Among the various options for stratified random sampling, Latin Hypercube Sampling (LHS) is promising. It efficiently samples variables from their multivariate distributions and can be conditioned in the multidimensional space defined by environmental covariates, then called conditioned Latin Hypercube Sampling (cLHS) (Minasny and McBratney 2006). The cLHS approach may be used to optimize the sampling design and improve predictions of species distributions by introducing spatial structures of explanatory variables and their cross-spatial structures into the cLHS optimization procedure. This is of special importance as overestimates in species distribution models often result from a lack of relevant explanatory variables or spatial autocorrelation (Lobo and Tognelli 2011). Environmental variables and species distribution data are frequently recorded in different cell (grain) sizes (Lauzeral et al. 2013). Therefore, spatial resolution is critical in any examination of distributions of species (Lauzeral et al. 2013). Reliable methods to downscale environmental variables or species distributions from coarse to fine grain resolutions have potential benefits for ecology and conservation studies (Keil et al. 2013). In regard to spatial resolution, species distribution models (SDMs) are impacted by the fact that environmental descriptors of samples are frequently recorded at different resolutions (Lauzeral et al. 2013) and may thus require scaling to the same resolution. A method called Area-to-Point (ATP) kriging uses spatial structures of predictors for downscaling to predict species distributions (Keil et al. 2013) by taking spatial dependence of predictors into account. We illustrate the approach using Swinhoe's blue pheasants (*Lophura swinhoii*) in Taiwan as an example.

a



b

Figure 1. (a) Flowchart of the procedure of spatial downscaling conditioned Latin Hypercube Sampling (sdcLHS); (b) Interface of the Windows-based tool of spatial conditioned Latin Hypercube Sampling (scLHS).

Combining spatial downscaling with conditioned Latin hypercube sampling (sdcLHS)

In Latin Hypercube Sampling, one must first decide how many sample points to use, and to remember for each sample point from which row and column the sample point was taken. Statistically expressed, the sdcLHS approach will address the following optimization problem: Given N sample sites with environmental variables (Z), select n sample sites ($n<<N$) such that the sampled sites form a Latin hypercube. For k continuous variables, each component of Z is divided into n equally probable strata based on their distributions and z denotes a sub-sample of Z. The steps of the sdcLHS algorithm, which are based on those in cLHS (Minasny and McBratney 2006), are as follows (Figure 1a):

Step 1. ATP kriging to downscale environmental variables Z from coarse scale to the fine scale.
Step 2. Division of the quantile distribution of Z into n strata; calculation of the quantile distribution for each variable.
Step 3. Selection of n random samples from N; calculation of the correlation matrix of $z(T)$.
Step 4. Calculation of the objective functions. The overall objective function integrates four different components (objective functions) (for details see Lin et al. 2014). For general applications, the weight assigned to each component in the overall objective function is equal.
Steps 5 to 7. Steps 5-7 are optimization procedures (for details see Minasny and McBratney 2006).
Step 8. Repetition of steps 4 to 7 until either the objective function value falls beyond a given stop criterion or 10,000 iterations are completed.

The spatial conditioned Latin Hypercube Sampling (scLHS) (Lin et al. 2014) is the sampling part (steps 2 to 8) of sdcLHS (Figure 1a), developed as a Windows-based tool to select optimal sampling sites (Figure 1b).

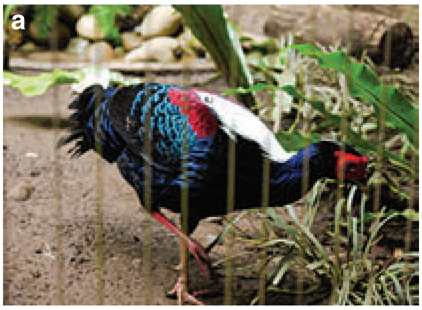


Figure 2. Photo of Swinhoe's blue pheasants (*Lophura swinhoii*) (a) male; (b) female.

Illustrative example

We applied the optimal sampling method with a downscaling approach to locate optimal sampling sites at the 2 × 2 km scale and to improve the identification of the spatial structure of the distribution of Swinhoe's blue pheasants (Figure 2) in Taiwan. The distribution of the focal species was estimated by Maximum Entropy (see Maxent; Phillips et al. 2009) based on the existing 803 2 × 2 km samples and separately based on 725 1 × 1 km samples (Figure 3). The estimated distributions were assumed to be the real distribution of the focal

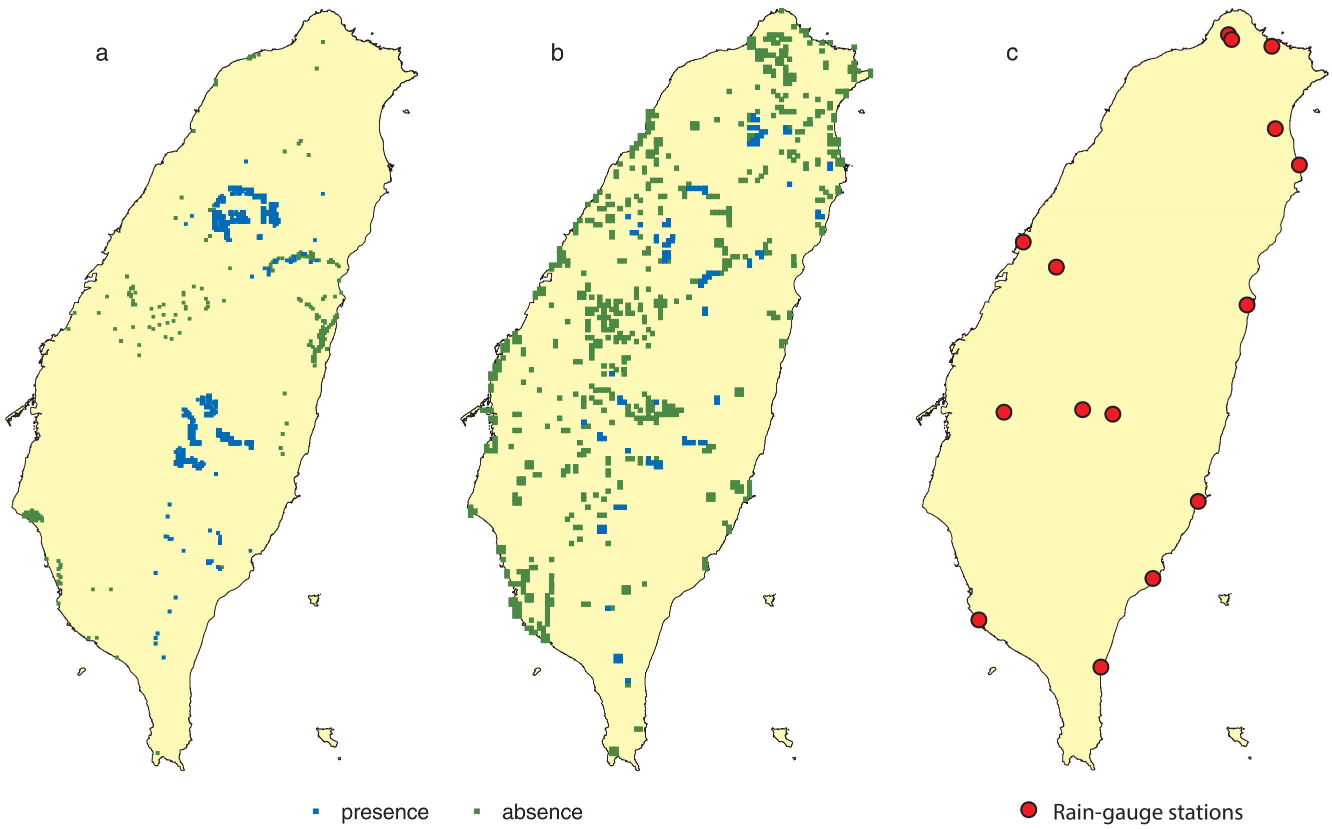


Figure 3. Observed samples with presence and absence data of Swinhoe's blue pheasant (Lee et al. 2004) in (a) 2 × 2 km sample sites; (b) 1 × 1 km sample sites; and (c) rain-gauge stations used for scaling validation. (Blue: presence; Green: absence).

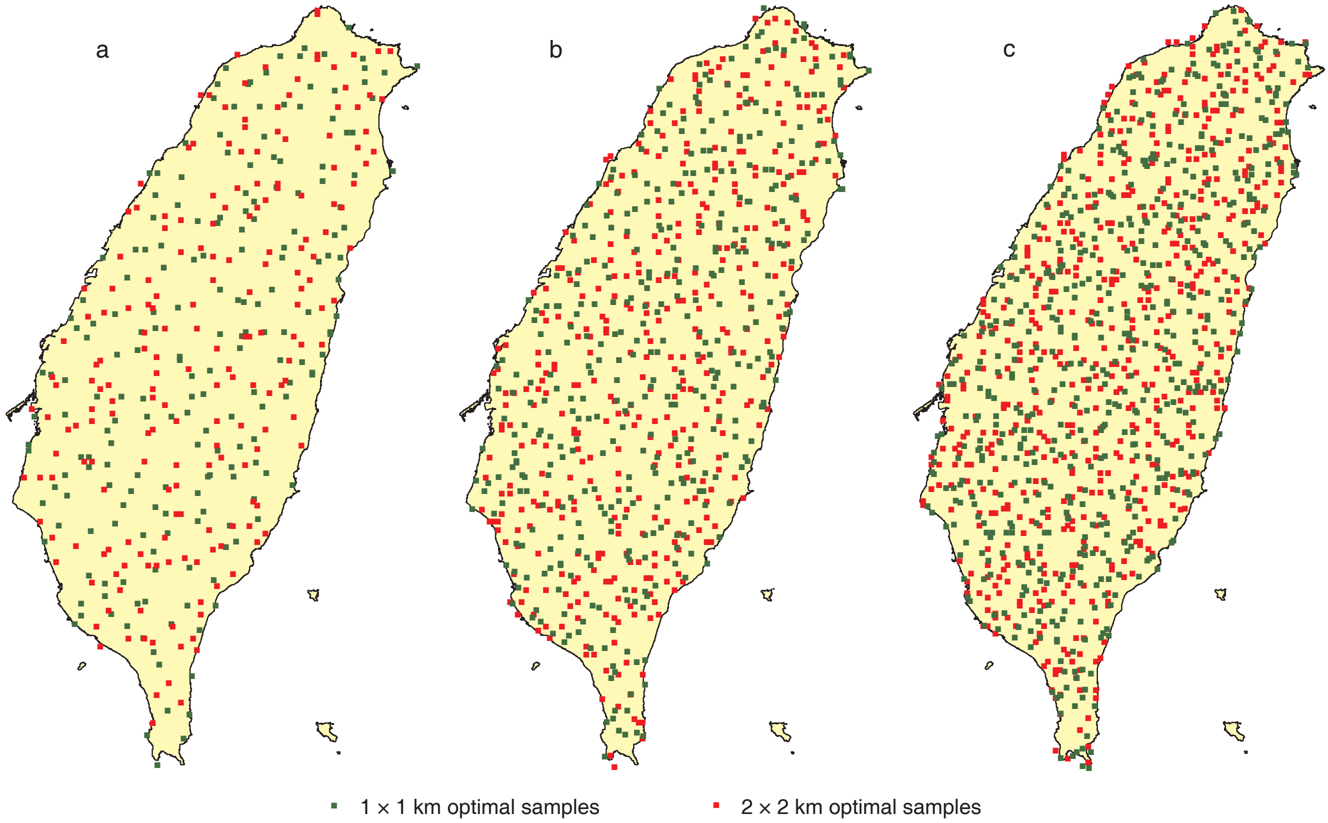


Figure 4. Locations of (a) 200, (b) 400, and (c) 600 2 × 2 km and 1 × 1 km samples derived by the optimal sdcLHS approach (sdcLHS: spatial downscaling conditional Latin Hypercube Sampling).

species for evaluating our proposed approach.

The presence data of the species at certain locations, determined from a set of samples based on presence-absence data, combined with the values of a selected set of environmental variables were used as input for the calculations. The resulting output represented the distribution of maximum entropy among all distributions satisfying the set of constraints (Phillips et al. 2009) These methodological constraints required that the expected value of each environmental variable under the estimated distribution was nearly equal to its empirical average (Phillips et al. 2009). The performances of Maximum Entropy were validated by the Kappa and AUC values.

The sample locations at *2 × 2 km* and *1 × 1 km* resolution were partially clustered due to similar spatial patterns and structures (variograms) of the variation of several environmental parameters (Figure 4). The Kappa value of the Maximum Entropy model was 0.38 and the AUC value was 0.86 in model validations using 401 samples at the *2 × 2 km* resolution. The Kappa and AUC values in the Maximum Entropy method were slightly higher when using 362 samples (Kappa= 0.58; AUC= 0.92) at the *1 × 1 km* resolution. The predictions with 200, 400 and 600 optimal samples taken from the assumed real distributions showed a consistently high performance, with AUC values of 0.99 and Kappa values of 0.97-1.00 for *1 × 1 km* cells and AUC values of 0.98 and Kappa values of 0.96-0.98 for *2 × 2 km* cells.

Concluding remarks

Incorporating spatial dependency of variables with different resolution into sampling approaches is critical to achieve efficient, unbiased spatial sampling. In the frame of the EU project SCALES, we have tested here an optimal sampling approach using the spatial downscaling sdcLHS based on selected environmental variables without pre-sampled species data, and used a Maximum Entropy approach to show the efficiency of the proposed approach in capturing the distribution of the endemic Swinhoe's blue pheasant in Taiwan. Our analysis showed that fine scale data yielded accurate presence/absence maps using a subset of presence/absence data that were optimally located. Locations of samples tended to be non-randomly spatially distributed when sample size increased at a coarser cell size. In regards to cost and resource efficiency without the loss of spatial structures (variograms) of focal species, our method with a sufficiently large sample size, 200 optimal samples in this case, performed well in capturing the spatial structure and predicting the spatial distribution of the focal species.

We conclude that the proposed sdcLHS approach considers the statistical distributions and effectively exploits the spatial structures of the selected environmental variables to capture spatial correlations in the original data recorded at various cell sizes. In addition, our approach does not require pre-sampled species data to select spatially unbiased sample locations based on information of parameters collected at various scales.

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Climate and land-use change affecting ecological network efficiency: The case of the European grasslands

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Introduction

Over the last decades, the establishment of protected areas (PAs) has served as the main tool towards biodiversity conservation. Still, although the selection of eligible sites has been based on different criteria and prioritization methods (Tsianou et al. 2013), the potential impact of global changes upon biodiversity has largely been ignored during the design and establishment of PAs networks. Climate and land-use changes are currently considered as the main threats to biodiversity, leading to changes in species distributions and ultimately species extinctions, affecting the effectiveness of established PAs (Araujo et al. 2011). The influence of global changes upon PAs could be caused directly by altering environmental conditions within each PA, or indirectly through changes in the community structure or by altering the physical and spatial properties of the landscape matrix surrounding PAs Such changes may increase the isolation of PAs and thus act as barriers to movements of individuals among sites, reducing connectivity.

Connectivity is a critical component for ensuring and evaluating the efficiency of PA networks. Maintenance of connectivity between PAs could allow the flow of individuals and, thus, genes, which in turn reduces extinction risk. Understanding connectivity for species with different habitat requirements and dispersal potential may allow identifying alternative routes for overcoming harsh environmental conditions in the landscape between protected areas. Ensuring connectivity may further allow species to escape catastrophic events or to recolonize areas, which become vacant. To this end, different approaches have been developed for assessing connectivity (see Klenke et al. 2014 this book and Mazaris et al. 2013) but assessments of PA networks at a large scale and across different scales are still rare.

The connectivity of a given set of PAs could differ considerably depending on the species under study. Different species traits, such as habitat preference, dispersal capacity and area requirements, could significantly affect the way the total landscape is perceived by the organisms, resulting in different levels of connectivity.

In this chapter, we present a methodological framework to evaluate the efficiency of a PA network under the prism of global changes by assessing connectivity at different scales. Our methodology was applied at a European scale in order to assess the connectivity of the Natura 2000 conservation network, the backbone of biodiversity conservation in the European Union (EU). We used grassland birds as the conservation target to develop and test our methodology.

Methodology

Our methodology consists of four steps. The first step involved the development of land cover maps for both current and future distribution of grassland habitats by the means of a generalized dynamic vegetation model. Briefly, we used the results of the spatially explicit land allocation model Dyna-CLUE (Dynamic Conversion of Land-use and its effects; Verburg et al. 2010) to initialize a version of the dynamic vegetation model LPJ-GUESS, which has been enhanced to incorporate land-use and land-use changes. Details on the methodology applied for modeling the structure and dynamics of terrestrial ecosystems at different scales, the processes included and the land use and climate change scenarios can be found in Lehsten and Scott (2014 this book). As a short note, habitat classification was performed according to the dominance of certain plant types (for grassland, needle-leaved, broad-leaved, Mediterranean and mixed forests) or land use (e.g. for pastures cropland and urban land uses). In our case habitats were classified based on species leaf area index, so that all the cells that have the majority of leaf area index in grass were defined as grassland. The outputs of the models, which were further used in our analyses, consist of maps presenting the land cover of grasslands in Europe for 2001 (referred to as present distribution) and 2030 (referred to as future distribution).

As a second step, we overlaid current and future projections of grassland distributions with the map of Natura 2000 protected areas to identify those patches that are protected and covered by the network. The Natura 2000 network currently consists of more than 26,000 sites distributed across the EU. In the context of our study, we considered Natura 2000 as a unified European network of protected areas and, consequently, we merged overlapping protected areas, regardless of them being characterized as Special Protection Areas on the basis of the Birds Directive (2009/147/EC) or as Sites of Community Importance defined on the basis of the Habitats Directive (92/43/EEC).

In the third step of our analysis we identified the hypothetical groups of species that were used in our subsequent analyses. The identification of groups was based on two allometric equations that consider average body mass to provide estimates of maximum dispersal distance (Sutherland et al. 2000) and minimum area requirements (MAR; Pe'er et al. 2014). With maximum dispersal distance being strongly correlated to minimum area requirements, we defined eight hypothetical groups of species representing a gradient of dispersal distance (ranging from 18 to 70 km) and area requirements (ranging from 50 to 5,000 ha) (Figure 1). Group 1 had low dispersal ability and area requirements, while group 8 had high dispersal ability and required large areas for a population to persist. For presentation purposes, and given the correlation among the three traits, we refer to groups 1-3 as small birds, groups 4-6 as intermediate, and groups 7-8 as large.

At the final step, we assessed potential connectivity among protected patches of grassland by applying a graph-theory based approach (Urban and Keitt 2001). Graph models were developed based on current and future grassland distribution inside protected areas. In graph-theory, a graph is composed of two basic elements: nodes (in our case the Natura 2000 PAs that included patches of grasslands) and edges (which represent the potential linkages between nodes). We used the centroid of each PA (node of our network) as an estimate of its geographic location. We calculated the Euclidean distances between each pair of nodes, using those centroids. We defined different networks for each species group, which included as nodes only the PAs with grassland area larger than the groups MAR; and two nodes were connected only if the Euclidean distance separating them was shorter than the dispersal ability of the group. Two network models were developed for each species group, based on the current and future distribution of its required habitat respectively.

Connectivity of the developed networks and the influence of species traits and global changes were assessed through the application of a series of network topology metrics, including number of nodes, number of links, number of components, percentage of isolated nodes, mean component size, order of the largest component (standardized with the total number of nodes in the network) and percentage of articulation points (Urban and Keitt 2001, Mazaris et al. 2013).

Apart from the network topology metrics, we applied a connectivity metric that has area units and takes into account the available area in each node representing habitat patches. The index is named Equivalent Connected Area (ECA) and is expressed in area units (Saura et al. 2011). To assess the potential impact of spatial changes on network connectivity we have to compare the relative variation (dECA) in the ECA values as calculated for two networks with the total change in the available habitat area (dA) (Saura et al. 2011). In our case, for each group of species, the ECA was initially calculated for the present distribution of grasslands and then for their future distribution; habitat area was estimated correspondingly. When dECA values are larger than dA, the spatial changes seem to improve network connectivity, while dECA values lower than dA indicate a negative impact on network connectivity.

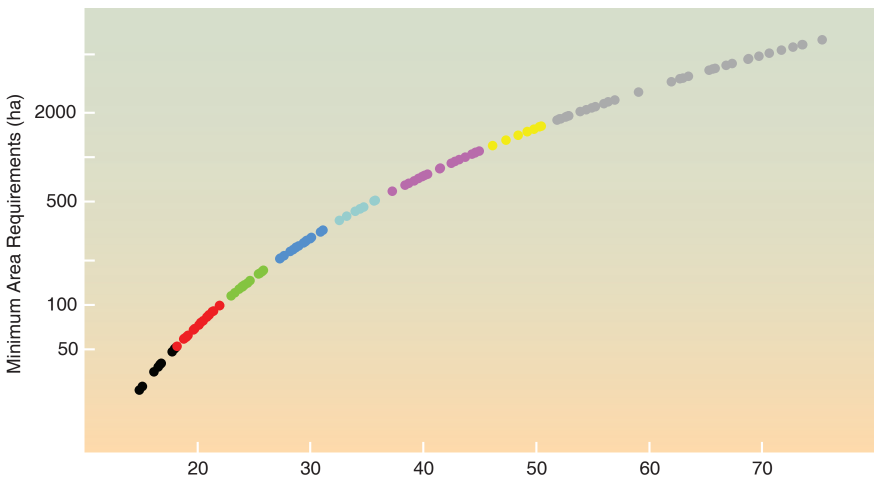

Figure 1. Minimum Area Requirements in ha versus Maximum Dispersal Distance in km for 124 bird species inhabiting grassland habitats.
Different colors are used to depict each one of the eight species groups.

Results

According to the projections of the LPJ-GUESS model, grasslands are expected to lose a significant amount of their land cover inside Natura 2000 protected areas by 2030 compared to 2001. The total area of grasslands is expected to decrease by 16.8%, while a reduction is also predicted for the total number of protected patches (-8.7%) and mean protected patch size.

The same declining trend in terms of area and number of protected patches was found for all the eight different species groups examined. The reduction is becoming more pronounced as the maximum dispersal distance and the MAR of the species increases, with larger grassland species suffering greater losses, up to more than 50% of the suitable area.

The significant loss of available area and number of habitat patches resulted in a subsequent deterioration of the network properties for all species groups. In all cases the number of nodes and links displayed a decreasing trend (Table 1). Consequently, the percentage of isolated sites and the percentage of articulation points increased in most cases,

Table 1. Basic metrics for the networks of grasslands included in the sites of Natura 2000 network for eight groups of grassland birds.

Group	MAR (ha)	Maximum Dispersal Distance (km)	Number of nodes		Percentage (%) of nodes change	Number of links		Percentage (%) of links change	Number of components		% number of isolated nodes		% number of articulation points	
			2001	2030		2001	2030		2001	2030	2001	2030	2001	2030
1	50	18	17841	17299	-3.04	231807	194903	-15.92	832	903	1.70	2.02	1.89	2.06
2	100	22	13725	12977	-5.45	229069	183077	-20.08	533	612	1.33	1.62	1.48	1.49
3	200	27	4353	3536	-18.77	44047	29469	-33.10	334	329	2.78	3.31	1.75	2.57
4	300	30	2134	1527	-28.44	15504	8989	-42.02	231	223	3.89	6.02	1.73	3.80
5	500	35	809	498	-38.44	3871	2429	-37.25	126	108	6.67	11.24	3.09	2.41
6	1000	44	183	128	-30.05	430	394	-8.37	47	30	10.93	8.59	6.01	2.34
7	1500	49	86	56	-34.88	164	140	-14.63	26	16	12.79	12.50	4.65	1.79
8	5000	70	20	13	-35.00	42	25	-40.48	6	4	15.00	15.38	0.00	0.00

indicating lower connectivity and more fragmented networks. The number of components increased only for small birds. However, taking also into account the reduction in the mean component size and in the standardized order of the largest component, we infer again that the future networks of all species groups are less well-connected compared to the current ones resulting into higher vulnerability for species.

The latter finding was strongly supported by the habitat availability index ECA. In all cases, connectivity was significantly reduced in future networks, as displayed by dECA (Table 2). In comparison to area loss, the networks of small birds seem to be the most profoundly affected. Specifically, the reduction in network connectivity for small birds was double than the reduction in area. For intermediate sized species, the change in connectivity was proportional to the change in area, while for long-distance dispersers with large MAR the reduction in connectivity was only slightly lower than expected considering the loss of area alone.

Table 2. Total change in the available habitat area (dA) and relative variation in Equivalent Connected Area (dECA) for eight groups of grassland birds.
The table refers to the changes of the grassland networks over the years 2001 to 2030. The increased intensity of red color is indicative of a higher reduction in the estimated metrics.

	MAR (ha)	Maximum Dispersal Distance (km)	dA	dECA
1	50	18	-17.07	-33.73
2	100	22	-19.51	-36.51
3	200	27	-29.89	-39.48
4	300	30	-36.56	-40.69
5	500	35	-43.33	-43.21
6	1000	44	-43.90	-43.84
7	1500	49	-48.84	-46.98
8	5000	70	-55.43	-52.35

Discussion

Here we applied a methodological framework to evaluate the connectivity of protected area networks and their performance against future changes. We assessed the impacts of global changes at a European level focusing on the connectivity of the Natura 2000 conservation network for grasslands and groups of species sharing similar traits. We examined the potential current and future geographical ranges of grasslands in Europe under scenarios of both land-use and climate change based on a generalized dynamic vegetation model. The outputs of the dynamic vegetation model were used to evaluate differences between the occurrences of broad habitat categories in Natura 2000 sites in 2001 versus their occurrence in 2030.

Our findings demonstrate that the performance of the Natura 2000 network regarding the conservation efficiency of grassland avian fauna will be severely affected by climate and land-use changes. Our dynamic vegetation model projections showed a significant reduction of grasslands within PAs, which suggests that the current spatial network configuration would be insufficient to protect grassland birds in the future. Furthermore, the future reduction in the number of protected patches would result in reduced connectivity for all species groups examined.

Grasslands are regarded as one of the most imperiled ecosystems with respect to climate and land-use changes. Consequently, there is an urgent need to establish protected areas that could sufficiently protect these ecosystems and reduce the impeding danger of range contractions and local extinctions of species inhabiting them. However, our results do not provide confidence that the current Natura 2000 network will achieve this goal in the future.

According to our study, grassland species with low area requirements (small MAR) may lose a smaller proportion of their currently available habitat patches compared to intermediate and large species; however, for all the groups, network connectivity is expected to be severely affected due to the spatial configuration of changes. As shown by our analyses, the protected grassland patches that are expected to be eliminated due to climate and land-use changes seem to be important for overall network connectivity. As a result, conservation efforts should focus on preserving sites that are essential for enhancing future network connectivity for all grassland species and for facilitating their movement to the habitat patches that will continue to be available in the future.

The approach presented here could be a valuable conservation tool in the face of future climate and land-use changes. Our methodology that combines land-use projections, dynamic vegetation models and network analysis allows making important inferences about the performance of conservation networks, even at a large scale. Additionally, the integration of different species traits facilitates the identification of vulnerable species groups and, thus, management priorities based on the needs and the risks of each species group. Still, we acknowledge that our methodological framework provides a rather conservative approximation of node inclusion and network connectivity, resulting in a robust evaluation, and most likely an underestimation, of network connectivity both in the present and in the future. In addition, our methodology was based on the most conservative assumption on the linkages between habitat patches without considering the role that suitable habitat patches that are not protected might play in facilitating connectivity.

A further development of this methodology might consist of the incorporation of detailed information on grassland habitats that occur outside Natura 2000 sites. In addition, even within Natura 2000 sites, grasslands only comprise a small proportion and may be fragmented. Under this context, a future methodological step could include the spatially explicit consideration of the structural properties of the landscape. We acknowledge that any such attempt would add complexity to the modeling regime and increase computational requirements, but it could also allow obtaining more robust and realistic estimations of grassland connectivity within the PA network. As a final note, and considering future development of the methodology presented, we should mention that graph theory considers connectivity on the basis of distances and the potential capacity of species to move between patches, but ignores the actual movement of species, the decisions taken by individuals, and potentially also their need to move – and hence the response to habitat availability and conspecifics. Thus, it would be of great importance

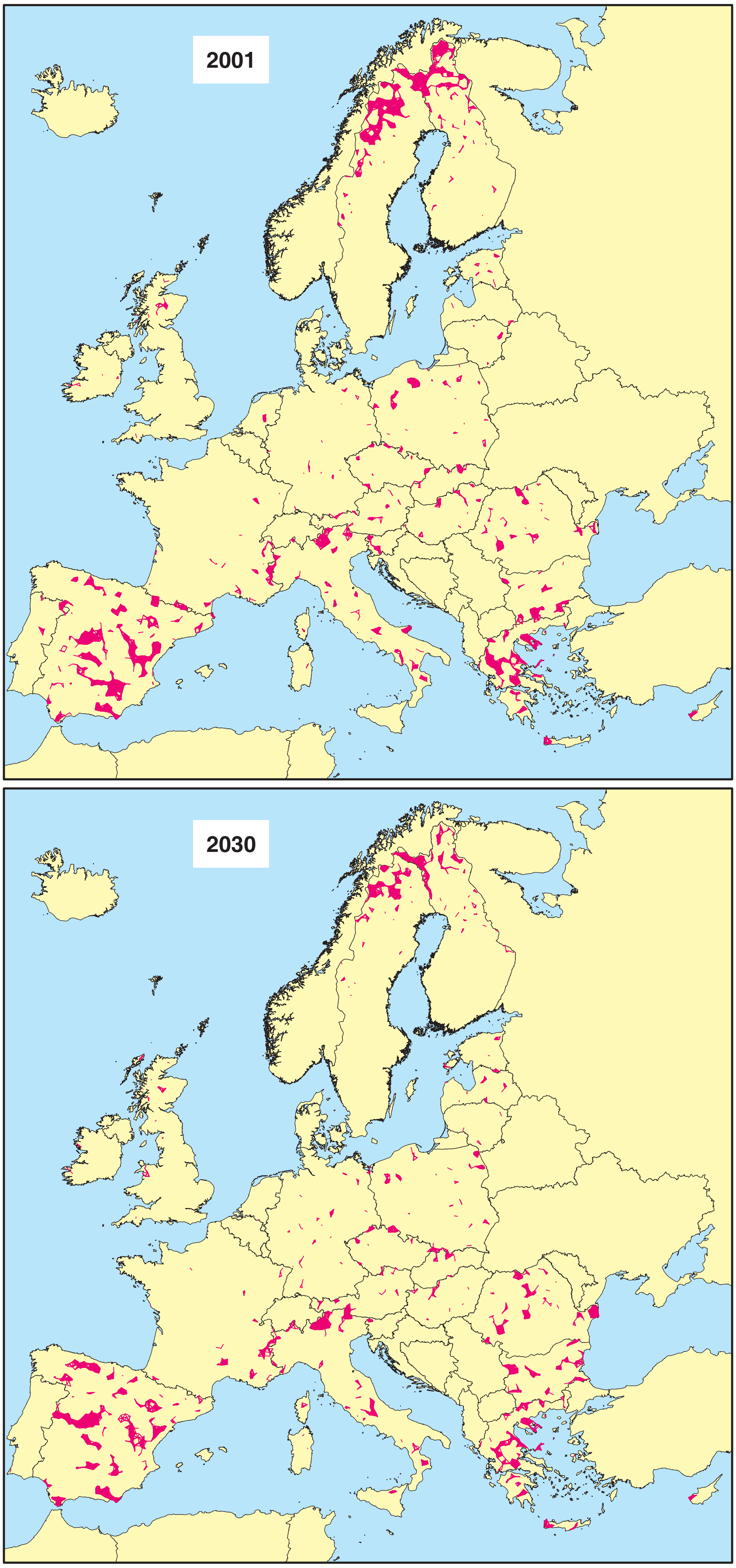

Figure 2. Network structure of the current and future modeled distributions of small birds inhabiting grasslands. The network models for species groups with maximum dispersal distance of 27 km and minimum area requirements of 200 ha are depicted on the map of Europe based on the model distribution of grasslands for 2001 and 2030.

to consider connectivity both within and among Natura 2000 sites, taking into account habitat availability and configuration over entire landscapes.

Currently, the Natura 2000 network covers more than 17% of the terrestrial territory of the European Union and represents the most extensive network of conservation sites in the globe. Under the prism of global environmental changes, it is critical that the efficiency of such networks is maintained or even enhanced. Towards this goal, an increase of connectivity between protected sites is essential (Heller and Zavaleta 2009). Conservation and restoration of natural habitats within PAs could be the first step. Plausible solutions to increase connectivity could include the establishment of green corridors, the selection and inclusion of key-sites and stepping stones located in habitats outside the Natura 2000 sites.

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The importance of connectivity for agri-environment schemes

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Introduction

Agricultural intensification and land abandonment are two major drivers of biodiversity decline throughout Europe (Kleijn et al. 2011). Together they have led to a dramatic decline of semi-natural grasslands, resulting in severe habitat loss and fragmentation across the continent. Studies on semi-natural grasslands have reported percentages of habitat loss as high as 95% to 99% since the beginning of the 20th century (e.g. Hooftman and Bullock 2012). Agri-environment schemes (AES), regulated and funded by both the EU and the national government in Finland, provide a potentially important multi-scale tool to mitigate the harmful impacts of agriculture on biodiversity. However, the effectiveness of AES depends on how they are implemented at the local scale, which is a third policy scale in addition to European and national ones. In Finland the main challenge in the conservation of farmland biodiversity is the maintenance of traditional semi-natural habitats. These habitats are presently mainly threatened by abandonment and overgrowth, which makes them dependent on continuous management, either grazing or mowing. Thus successful application of specific AES measures, including active habitat management and restoration, have a central role in the protection of farmland biodiversity. However, despite high expenditures, experiences of the effectiveness of AES have been mixed (Kleijn et al. 2011).

One possible reason for the mixed performance of the AES is that the ecological effects arising from the landscape structure on the success of conservation have been neglected. Despite abundant evidence of the importance of habitat connectivity to species persistence (Figure 1), these subsidies are allocated at the local farm scale independently of any landscape context or subsidies given to other farms in the region. This suggests that the use of subsidies could be much more effective if spatially coordinated to enhance ecological processes, such as dispersal. In other words, the local farm scale AES contracts might critically benefit from being integrated within an intermediate landscape scale AES planning which would take the spatial relationships of managed sites

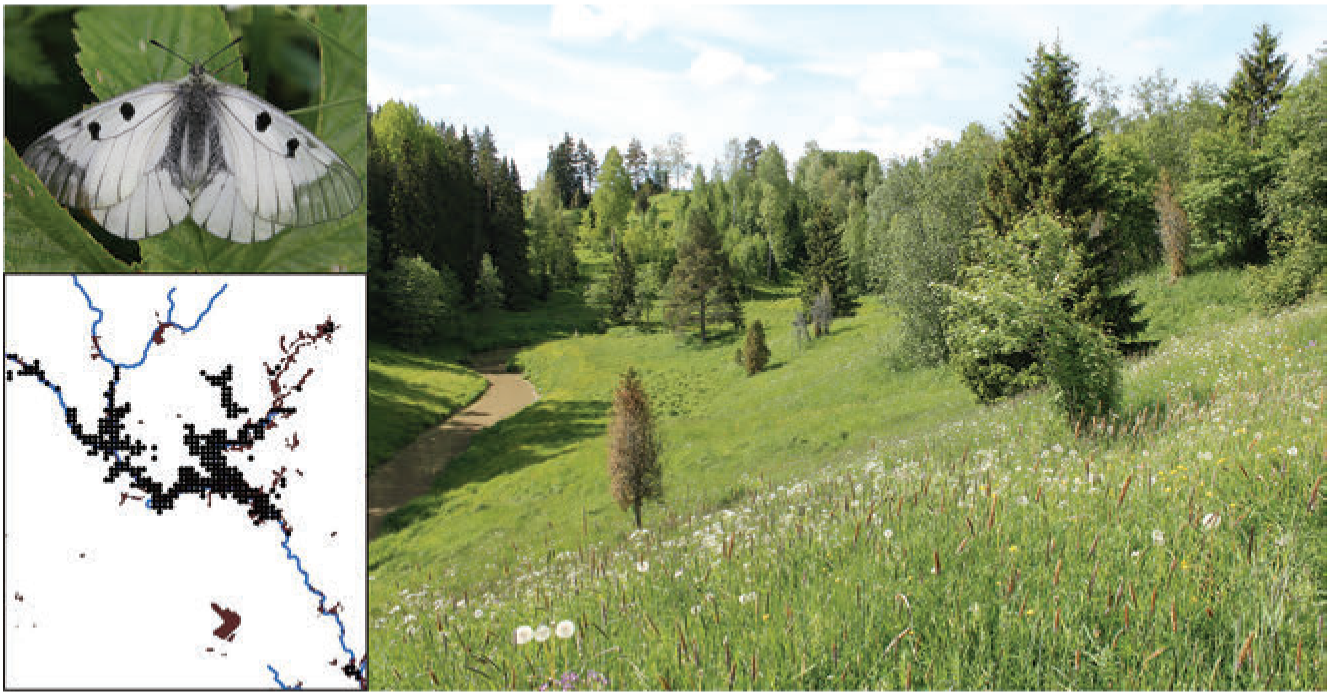

Figure 1. Many grassland specialist species benefit from high grassland connectivity. For example, the Europe-wide threatened Clouded Apollo butterfly (*Parnassius mnemosyne*) occurs in the best connected semi-natural grassland networks of south-western Finland (Heikkinen et al. 2005).

into account. Such spatially integrated planning requires certain types of numerical tools. This chapter provides an example of how the use of spatial prioritization tools (here Zonation software) may help with more effective allocation of agri-environment schemes (for a full account of the study, see Arponen et al. 2013). The study area is situated in south-western Finland, where high-resolution GIS data on the occurrence of semi-natural grasslands were available.

Spatial conservation prioritization with Zonation software

We prioritized the grassland cells for management with the Zonation software v3.1 (Moilanen et al. 2012, see Box 1 for more details on the software). The Zonation prioritisation outputs can be used to identify well-connected networks of high quality habitats. We replicated the analyses with and without connectivity. We implemented connectivity by using a "distribution smoothing" kernel in the Zonation cell value calculations: the value of each cell influences the value of its surrounding cells following the shape of the smoothing kernel. We tested kernel widths of 1, 2 and 4 km. The use of this feature means that high value cells that are located close to each other receive additional value and are highlighted in the solution as compared with isolated cells. This ensured that the connectivity among grassland patches reflected adequately the commonly used value for the mean dispersal range of grassland species.

We used three different datasets in our analyses (Figure 2): (I) Grasslands of conservation concern included in the Finnish national survey of traditional rural biotopes. These sites were classified into nationally, regionally and locally important ones primarily based on the occurrence of vegetation types and vascular plant species associated with traditional animal husbandry. (II) Other grasslands, which included open grasslands from the SLICES land cover database (National Land Survey of Finland, NLS), and semi-natural pasture areas from the register of the Ministry of Agriculture and Forestry. (III) Areas that have (or have recently had) AES management contracts of semi-natural habitats. These include two types of specific contracts: Management of traditional rural biotopes and Enhancement of biodiversity management. The former is considered more effective for the conservation of semi-natural grasslands as it is limited to species-rich traditional biotopes and typically contains actions such as mowing, grazing and prohibition of fertilizer use. The latter contain a wider spectrum of general actions for farmland biodiversity and landscape conservation, which are useful but often less effective for semi-natural grassland species. Datasets I-III were combined as the "habitat layer", whereas the management contracts (IV) were handled as a separate layer (Figure 2). The cells were given weights from 0.5 to 4 in the Zonation analyses on the basis of their conservation value (Figure 2). All data were transformed into 25 × 25 m grids.

Results

Because the solutions produced with the different connectivity scales

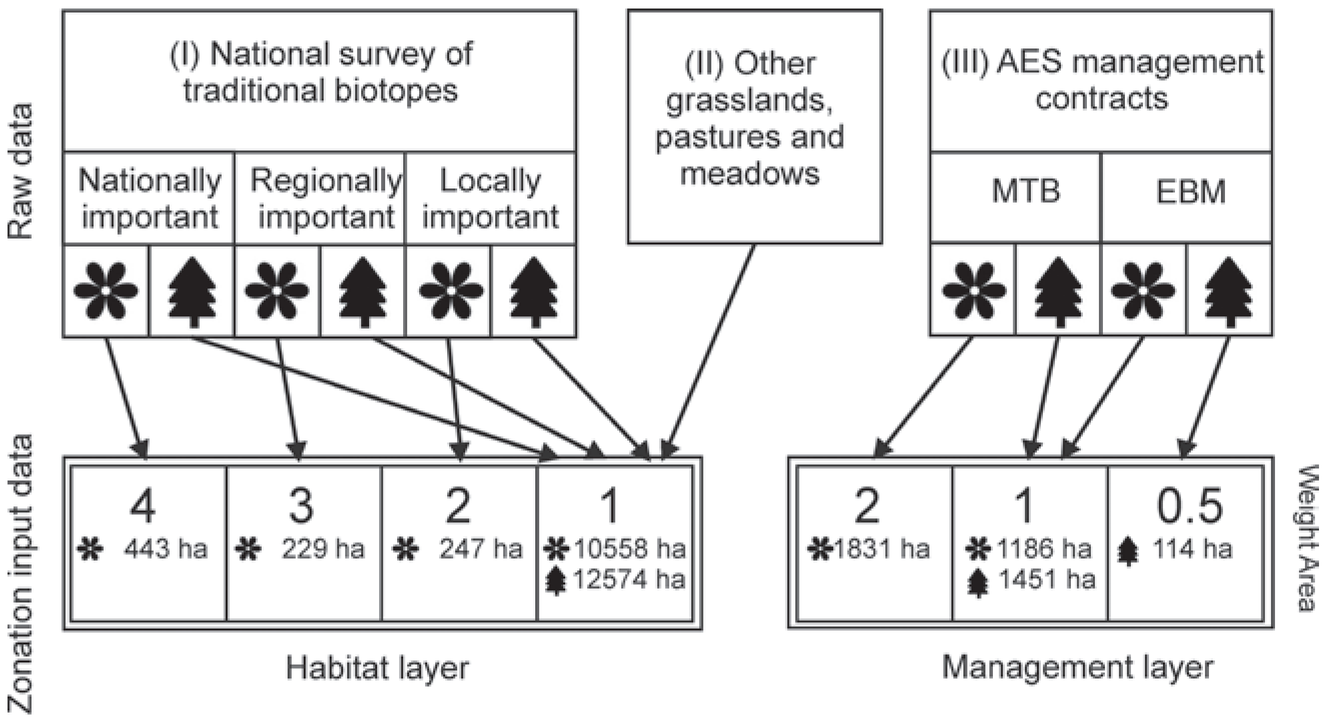

Figure 2. Descriptions of the input data. The upper part describes the raw data (I–III) used to derive two input layers to Zonation, described in the lower part. In the input layers the large numbers indicate weights given to those cells based on their assumed conservation value. The flower symbols indicate open sites, and the trees indicate wooded sites that were considered of lower conservation value currently, but were included with lower weights because they have potential for restoration and importance for improving connectivity of the network.

were similar to each other (correlations of cell rank order ranged from 0.87 to 0.97), we show the results only for the 2 km variant. When connectivity is not included, the priorities are scattered and closely follow the locations of most valuable traditional biotope sites (Figure 3b). Inclusion of connectivity helps to identify ecologically more coherent networks where species persistence is facilitated by easier dispersal between grassland patches (Figure 3c). These networks may also contain some grassland areas or partly wooded sites of lower current conservation value which could be improved by securing management through the AES. Such sites would provide valuable additional habitat near the most important localities, and facilitate population growth and dispersal of the species of conservation concern.

The highest priorities identified by Zonation partially coincided with the areas that were under agri-environment scheme management contracts (Figure 4). However, ca. 25–30% of the sites with the highest conservation values were unmanaged, and therefore are likely to eventually be suffocated by succession and lost. Some areas under management contracts fell into the lowest priority categories according to Zonation: altogether 12% of area that ranks lower than top 20% in Zonation is under management contracts. These are mostly very small and isolated sites that may have a history as a traditional biotope that is required from the AES contract sites, but which probably are not very valuable regarding the species they contain. In particular, they are unlikely to maintain viable populations of valuable species into the future.

We also ran the Zonation prioritization with settings that force the current management contracts to be included in the top rank to simulate expansion of the current network. The conservation value of the currently managed network was 85% of that of an optimal network. To achieve the same conservation benefits as with an optimal network of sites selected by Zonation, the current network of managed sites should be expanded by 50%, or by

Box 1. Spatial conservation planning framework and software ZONATION

Zonation is a conservation planning framework and software. It takes as inputs raster layers that contain the occurrence levels of biodiversity features (e.g. species) in sites (grid cells). It produces a hierarchical prioritization of the landscape by iteratively removing the least valuable remaining cell while accounting for connectivity and complementarity of species composition in the different cells. The outputs of Zonation can be imported into GIS software to create maps or for further analysis. Zonation v. 3.1 can process very large data sets containing up to ~50 million grid cells containing data. Zonation identifies areas important for retaining habitat quality and connectivity for multiple species, indirectly aiming at species' long-term persistence.

Zonation can be used to address questions such as:

- Identification of cost-effective protected areas
- Identification of protected area expansions
- Identification of areas with low conservation value that can be allocated for competing land uses

Zonation can incorporate many aspects useful for planning purposes, such as:

- Species-specific connectivity responses
- Species (or other feature) weighting
- Species interactions (e.g. predator-prey dynamics)
- Uncertainty analyses

Other Zonation-studies have been conducted in the context of SCALES

- Zonation can accommodate multiple administrative regions in a single analysis, facilitating comparisons between analyses at different administrative scales (Moilanen and Arponen 2011, Moilanen et al. 2013)
- Analysis resolution makes a great deal of difference for what is prioritized. High resolution increases cost-efficiency, but may produce small and scattered priorities unless connectivity is included. One should ensure that analysis resolution is adequate and combined with an appropriate connectivity measure to achieve cost-effective outcomes (Arponen et al. 2012)

The software package, including a Graphical User Interface, manual and tutorials, is available for download at http://cbig.it.helsinki.fi/software/zonation/

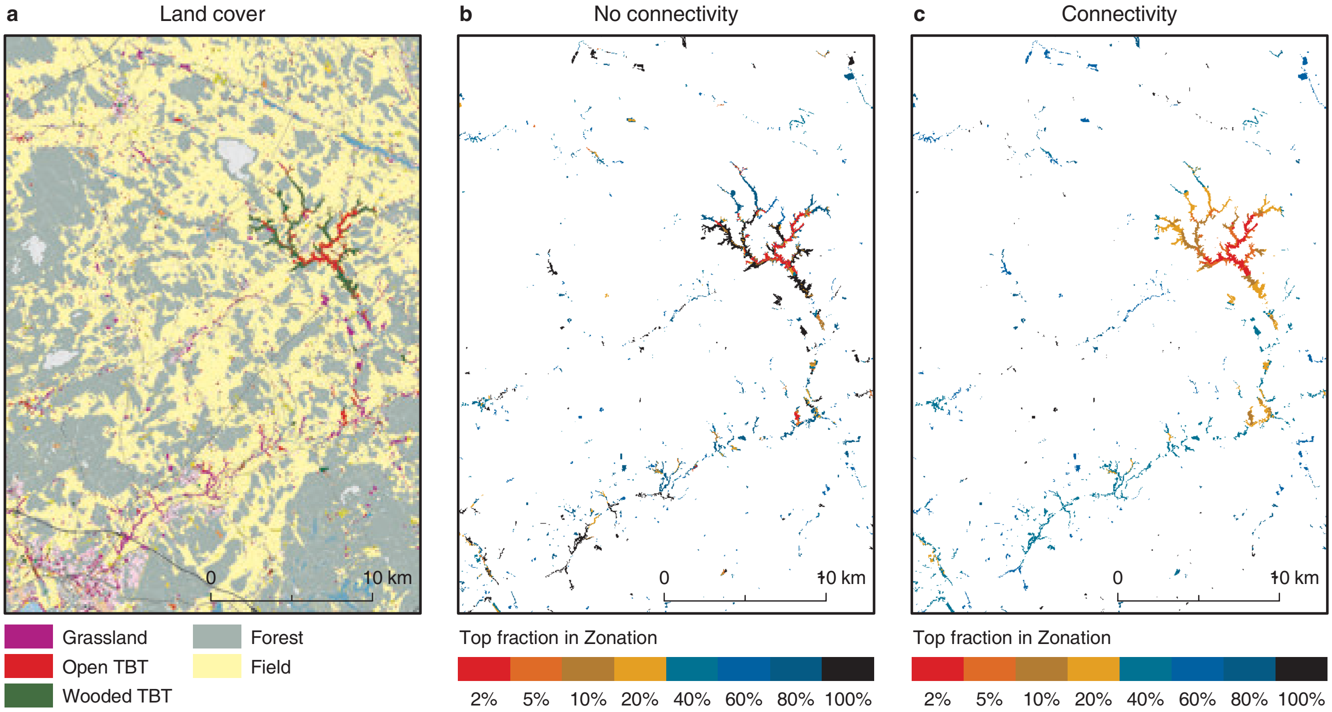

Figure 3. Land cover (a) and priority rank maps (b-c). The land cover map shows the locations of open and wooded traditional biotope (TBT) sites, and broad land cover categories for the surrounding landscape (b). The colours in (b-c) indicate high (red tones) and low (black) conservation priority: e.g. the best 2% of the landscape are in bright red. The maps are shown only for a small subsection or our study region because the grassland sites are very small and scattered, and would show poorly on a full map of the region. (b) is the solution without connectivity, and in (c) we have used the distribution smoothing feature in Zonation, where a 2 km kernel was used for calculating a smoothed conservation value of the raster cells to ensure connectivity between valuable sites.

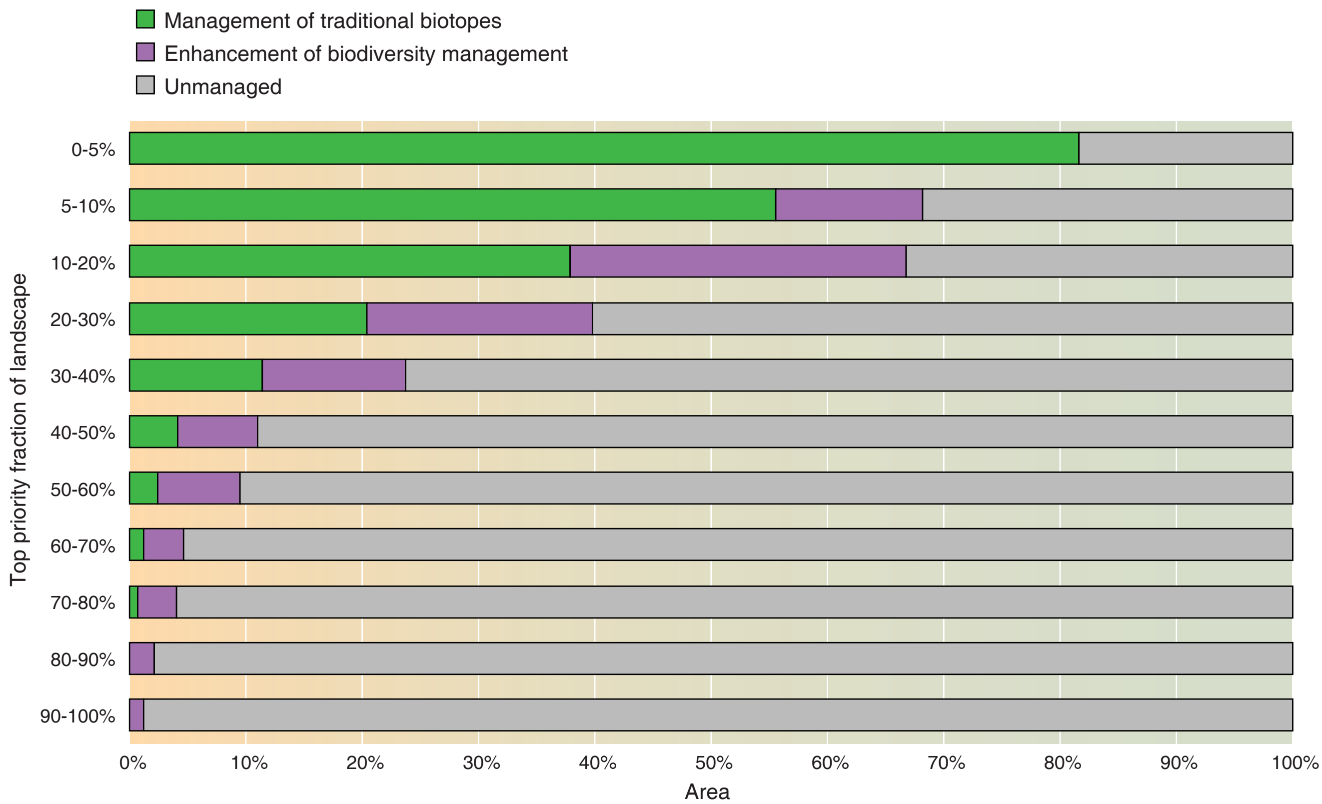

Figure 4. Management status of ranked cells. The cells are divided into different Zonation priority rank categories from best 5% (top) to worst 10% (bottom). Each bar shows the distribution of cells into the two types of management contract sites and sites outside agri-environment schemes.

1,700 ha (Arponen et al. 2013). This expansion seems realistically achievable, considering there is a national target to nearly double the amount of managed semi-natural habitats to ca. 60,000 ha. However, our results also show that reallocation of management contracts from scratch would be a more cost-effective strategy than expansion of present-day network of AES sites. This is an important finding, especially for situations where resources for AES (or other similar temporary biodiversity conservation tools) are limited.

We compared our connectivity solution to the Natura 2000 network in the same region by overlaying the grassland data with N2K sites (our data overlapped with 2.7% of the N2K data). Zonation rank values within N2K areas showed that the N2K conservation approach, where the authorities determine which areas make the best network, had captured valuable sites better than the voluntary-based agri-environment schemes (Figure 5). Even though voluntary measures have their advantages from a socio-political perspective (Paloniemi and Varho 2009), the study shows that more attention should be paid to improve their ecological effectiveness.

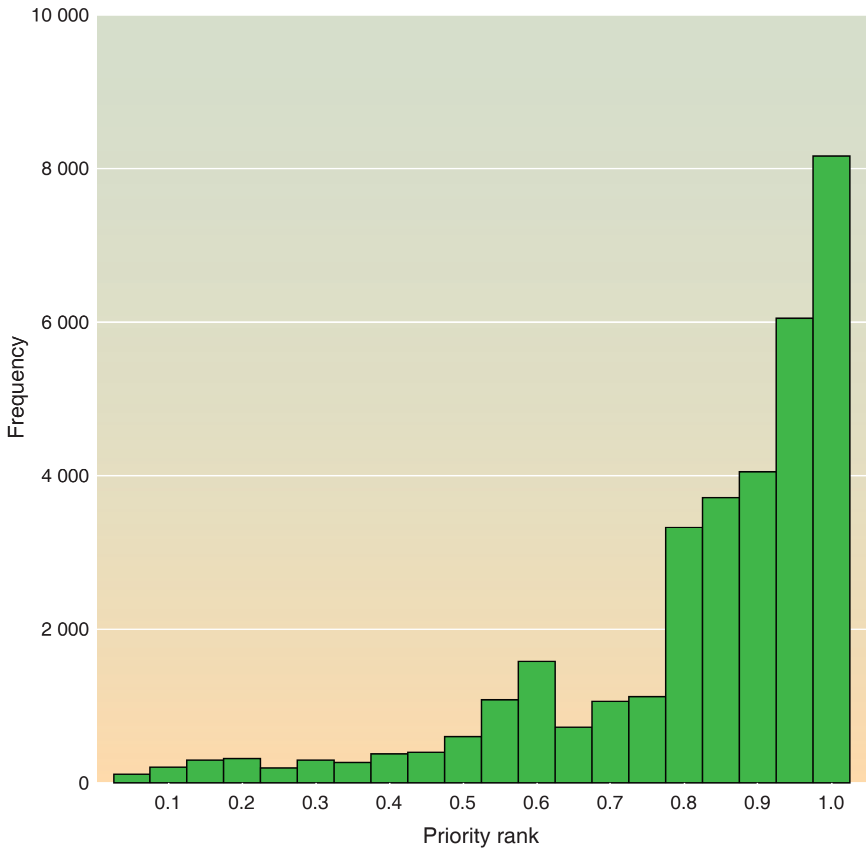

Figure 5. The distribution of cells in Natura 2000 areas that contain grassland among the Zonation priority rank values. Different priority rank categories are given on the x-axis at 5% intervals, and the y-axis indicates the frequencies of cells in each category. The low-ranking Natura cells typically occurred inside or at the edges of forest- and mire-dominated Natura areas that were not targeting semi-natural grasslands, but happened to contain some sparse grassland cells of lower value.

Policy recommendations

In order to improve the effectiveness of voluntary agri-environment schemes, decision-makers should include landscape scale criteria among the other criteria that are used in the planning process for granting subsidies. Spatial conservation prioritization tools, like Zonation, have the potential to provide useful information helping to target locations that should receive a high priority for conservation management. The relatively high ecological effectiveness of non-voluntary conservation schemes, like Natura 2000, suggests that such measures should have a role in a larger conservation strategy in the future as well. The success of the agri-environment schemes depends on their ability to motivate and involve the right people to take management action on the right sites. Potential means to achieve this include:

- Increased financial compensation. Currently only expenses are covered, which offers no true incentive for the farmers to participate.
- Differentiating payments according to the conservation value of the site. This could encourage the owners with most valuable sites to enroll.
- Agglomeration bonuses to enhance spatial connectivity. This could encourage the farmers to form collaborations, ensuring the management of large enough habitat networks.
- Improved dialogue between authorities and the land owners, to enhance landowners' awareness of the possibilities to use agri-environment schemes in safeguarding biodiversity as well as to share various knowledge about the conservation values and management demands of habitats and species that the schemes target.

The main multi-scale challenge in developing the effectiveness of the AES is the lack of flexibility in current EU scale policy instruments to reflect differences in conservation value at regional and local scales. More attention should be paid to the incentive structure and dialogue between farmers and those responsible for conservation planning on national, regional and local scales. In order to adhere to the commitments toward halting farmland biodiversity loss, the policy instruments for biodiversity conservation should be redesigned, which requires action at both EU and national level. Without EU level policy changes, the space of freedom of Member States is significantly constrained by EU regulation.

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Stay in contact: Practical assessment, maintenance, and re-establishment of regional connectivity

Reinhard A. Klenke, Yorgos Mertzanis, Alexandra D. Papanikolaou, Anni Arponen, Antonios D. Mazaris

"Connectivity remains one of the most difficult areas of landscape conservation. Measuring connectivity is not straightforward and metrics used can be highly problematic" (Lindenmayer et al. 2008).

Introduction

Indeed, dealing with connectivity is complicated because of often unclear terminology, the complicated mathematical handling of irregularly shaped and distributed geometrical objects forming landscapes, and the methodological problems we have to face when we work at multiple scales. Connectivity is very much related to terms like fragmentation and sub-dissection, but it is not simply the opposite of these terms, and it is mainly relevant in landscapes that were modified by man. In this chapter we would like to give a short guidance on how to deal with this complexity, summarise information from several parts of the SCALES project where we have tested methods for assessment of connectivity in real world examples, and give recommendations for the maintenance and improvement of connectivity among spatial objects of conservation interest.

Terminology

The literature about the effects of landscape structure on organisms is full of similar-sounding combinations of words like "ecological connectivity" or "habitat connectivity". Fischer and Lindenmayer 2007 tried to clarify the links between interconnected themes and have finally provided a clear and consistent terminology to describe landscapes modified by human activities, which should be considered generally. In combination with the results from Fischer and Lindenmayer (2006), it became more clear that species-oriented and pattern-oriented approaches are highly complementary when trying to understand the ecology of human-modified landscapes (Fischer and Lindenmayer 2006). The above-mentioned authors also differentiate between three types, or rather concepts, of connectivity, which are related in that they all focus on connections between the units of interest, but they are very different regarding their perspective and scales of interest (Figure 1). This distinction is necessary to overcome the confusion caused by the many different and often implicit definitions of connectivity found in the literature. Defining the three basic connectivity types is critical for accurately addressing questions by using the most appropriate tools, but also for the assessment and practical management in landscape planning.

Recently, there is also an increasing amount of literature mentioning functional connectivity, which emphasizes special aspects of habitat connectivity based on species' or individuals' interaction with their habitat. Based on these interactions, determined for instance by perception range, dispersal abilities or movement strategies etc., an obviously available habitat structure may not work as connecting element because of a lack of functionality. On the other hand,

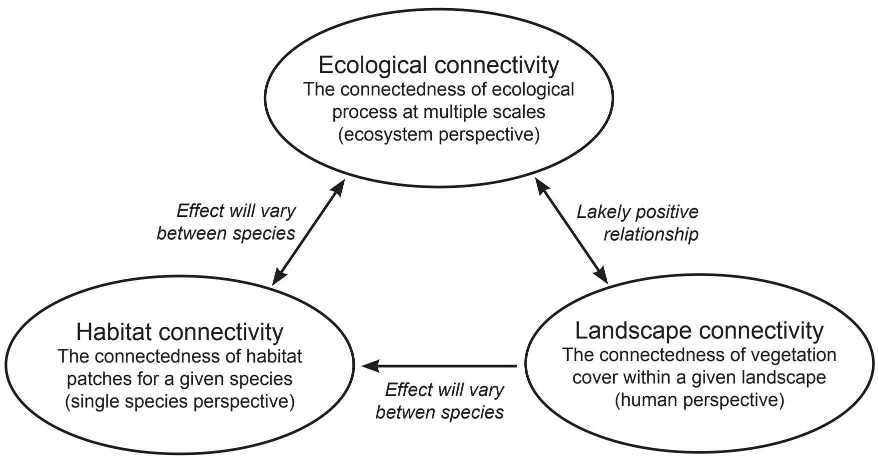

Figure 1. Relationship between the three connectivity concepts defined by Fischer and Lindenmayer (2007). The likely positive effects between ecosystem and landscape depend on the structure and size and type of vegetation and organisms; they also can be detrimental, for example, by allowing pathogens or invasive species to migrate or they may act as barriers for other organisms.

gaps between habitat patches may not work as a barrier, because the species under focus is able to fly and has no difficulty crossing even bigger distances. Functional connectivity is therefore a special aspect of structural habitat connectivity and discussed more in detail by Pe'er et al. (2014 this book).

Assessments of connectivity can be made in very different ways. Pe'er et al. (2014 this book) describe some of the methods and available tools, which are especially useful to analyse functional connectivity. However, methods that are very useful on a small scale with detailed and small-grained information about habitats and animal – landscape interactions may not be very applicable on a rather large scale where other aspects become important, such as the efficiency of networks of protected areas for a certain species or a number of species with different requirements. Therefore, it becomes apparent that each of the above mentioned concepts and scales requires its own approach. A species oriented approach using habitat suitability maps, so-called Species Distribution Models (SDM's), might be especially useful to analyse the connectivity of habitat (Habitat Connectivity) for a single species or a group of species (e.g. Almpanidou et al. 2014). On the other hand, analyses of landscape connectivity can also be performed under a rather general human perspective by using pattern oriented approaches under the basis of which we could compare different types of landscapes and/or analyse and monitor spatio-temporal changes (e.g. Arponen et al. 2014 this book).

Both approaches have their advantages as well as limitations. They also require different information and technical and mathematical solutions (e.g., representation of geographical data in raster format or vector format, matrix operations vs. graph based algorithms). For an appropriate parameterisation of the modelling procedures underpinning the interactions between landscape and species or processes, it is critical that information should not be limited to landscape components and data; it is important that good quality information is available about species' behaviour, range of perception, distributions of dispersal distances or the spatial extension of the ecological processes in focus.

Assessing connectivity on a regional scale

The scale of the analysis, defined in terms of the spatial resolution and the extension of the area, depends on various factors and is often determined simply by the availability of data and resources or the computational burden. By definition, regional connectivity implicates an analysis with a certain spatial extent and often higher-level units in the geographic hierarchy. However, the administrative level at which we can make such analyses is not always clearly defined, like for instance the Nomenclature of Units for Territorial Statistics (NUTS) geocode standard for referencing the subdivisions of countries for statistical purposes (compare Tzanopoulos et al. 2014 this book) but can vary in a very broad range. Therefore, we could estimate regional connectivity for units that are defined by administrative borders (e.g. counties, countries or even a political construct like the European Union), for units based on physical-geographical features, which allow us to define homogenous areas (e.g. Alps), or for units based on bio-geographical aspects, such as areas where a certain group of species is distributed in. All of these scales could be called regional scale (http://en.wikipedia.org/wiki/Region).

In studies of regional connectivity, the extent of the study area can range from a micro-scale (a few square kilometres) or meso-scale (several thousand square kilometres, federal state), up to a macro region (e.g. a whole continent or a socio-political unit like the European Union). It just depends on the availability of data and the size of the region on which we want to make an inference. But by definition, the extent has to be below the global scale. The grain or pixel size of the maps should follow basic principles as recommended by Hengl (2006) while the scale(s) of the analysis should be based on biological knowledge. The extent of a regional study need not necessarily define the smallest scale on which we can make an inference. Multiple scales can be addressed, for instance, with either a nested approach or with moving window statistics of different radii.

In SCALES we conducted several studies addressing aspects of connectivity at different regional scales. All of them used different methods. Two examples were described in separate articles in this book (Arponen et al. 2014 this book, Papanikolaou et al. 2014 this book). For a comparison of the methodological approaches and results we will summarise them later in this chapter. One example, published by Almpanidou et al. (2014), we will explain in more detail in the following. It is an assessment on a sub-national scale, considers landscape as a continuum and addresses habitat connectivity using a process-based approach.

Habitat connectivity: An example with brown bears in Greece

Here we present a methodological framework developed to view connectivity as a result of the interaction between habitat selection and movement (Almpanidou et al. 2014). The brown bear (*Ursus arctos*) is the most widespread bear in the world. In Europe it declined to small, fragmented populations across western and southern (i.e. Mediterranean) regions during the last decades. The brown bear distributional range in Greece consists of two separate population nuclei located in the Pindos (NW Greece) and Rodopi mountains (NE Greece). Despite the protected status and the conservation actions that have been taking place since the 1990s, brown bear conservation status in Greece remains critical and numbers are considered to be decreasing. These populations are under threat due to human-caused mortality and habitat fragmentation, loss and degradation, which raise the need for focused conservation practices driven by our knowledge on habitat use and movement patterns.

In an attempt to prioritise conservation efforts for brown bears, heightened attention has been given to habitat patches that have been recognised as highly suitable, given the fact that within them bears could fulfil their main needs for food and cover. Towards this direction, habitat suitability models have been developed for identifying those high quality patches. Still, the landscape matrix of the area where the species moves is also composed of patches of medium or lower quality. It is obvious that this matrix has an actual effect on movement strategies and patterns, and thus on habitat use through habitat selection processes. Therefore, it is likely that the study of movement patterns accompanied with information on habitat suitability is likely to display some spatial properties that could enhance our understanding of underlying drivers of connectivity and patterns of animal-habitat relationships. Any such information could guide our conservation efforts by re-directing them from a static point of view to a macroscopic perspective that would allow maintenance of overall connectivity.

By using data on eight male brown bears and landscape variables (north-eastern part of Pindos mountain range, in Greece), a three step framework was developed, which allowed the combination of habitat selection and movement as interacting processes towards revealing connectivity properties (Figure 2). The first step involved the development of habitat suitability models using satellite-tracking data of brown bears. In total we developed four habitat suitability models; for each one of these models we have used occurrence data from seven bears while keeping information from one animal at the time, which was used for the next step of the analyses, which is the development of the graph models. The final step involved the application of Markov chains for assessing how individuals modify their movements in relation to habitat quality.

Detailed network analyses provided insights on the contribution of patches of different quality to connectivity (Figure 3). They clearly demonstrated that in the movement network of the male brown bear, patches of low or medium quality are recognised as critical to facilitate movement and maintain connectivity. The results

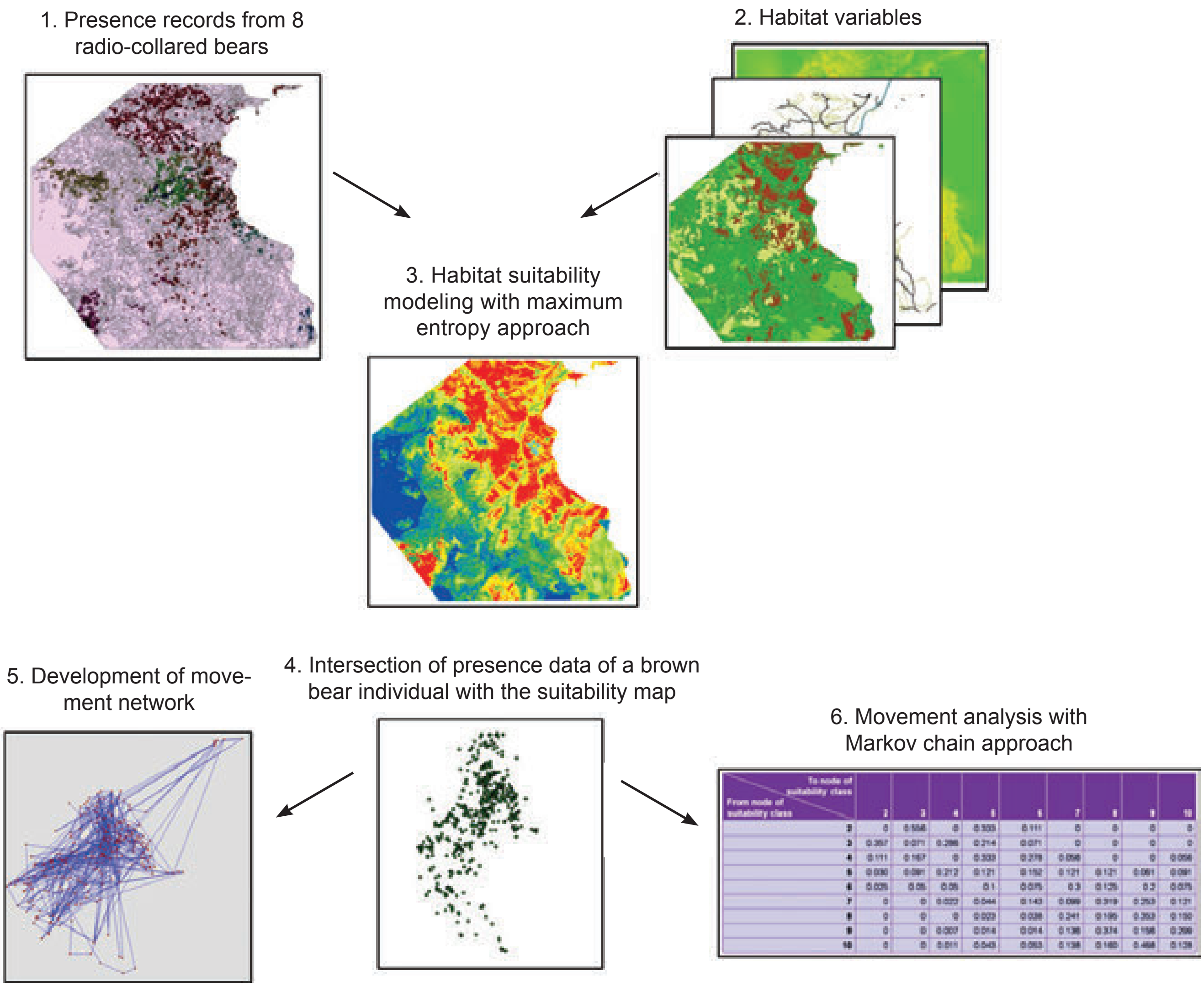

Figure 2. Methodological flow chart of the steps that were followed to study habitat selection and movement of brown bears. Steps 1-3 involve the development of a habitat suitability model. Steps 4-6 involve the contraction of movement network by using inter-connected patches as the nodes and the application of Markov chain for assessing movement among different patches.

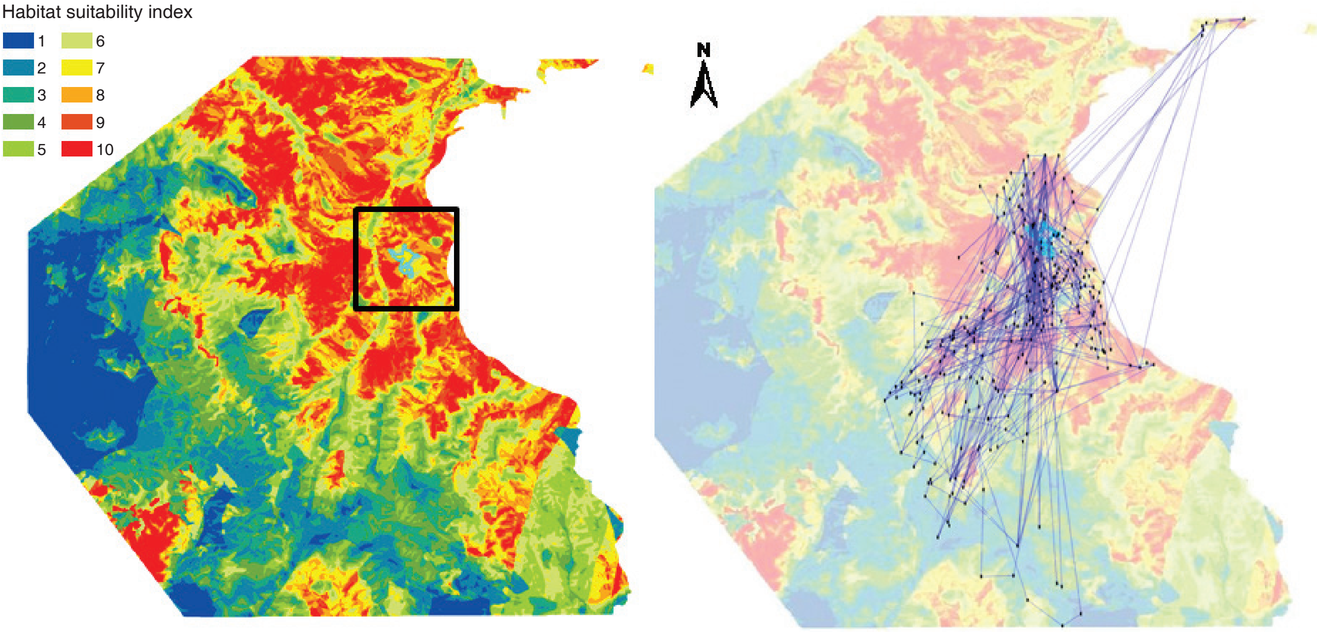

Figure 3. Habitat suitability map of a male brown bear produced by using telemetry data of 8 individuals, collected in the north-eastern part of the Pindos mountain range, Western Greece, over a twenty-month period (April 2007 – December 2008) provided by CALLISTO-Greece. Habitat suitability index is classified in ten classes. Habitat maps were produced by using data on 7 bears while withholding one bear to be used for the development of graph models. Dots and lines represent the centroids of habitat patches and the movement among patches for the studied animal, respectively.

demonstrated the importance of considering the entire habitat of brown bears in conservation planning rather than isolated patches of high quality, as might have been inferred by simple habitat suitability models. It is acknowledged that habitat suitability models are extremely useful in conservation since they rank the quality of the patches and could provide information on the underlying landscape properties; still, the incorporation of real movement data that reflect individual behaviour allow information and quality of the landscape to be judged from an animal's perception level (Almpanidou et al. 2014).

Whoever has will be given more: Aggregation of habitats gains priority

Example 2 was a study carried out on a 20,000 km² sized heterogeneous mosaic of forests, farmland and built up areas in South-western Finland. Arponen et al. (2014 this book) investigated the importance of connectivity for the prioritisation of areas to apply agri-environment schemes. They developed a spatial conservation planning framework and implemented it in the software ZONATION (Moilanen et al. 2012) that has been extended by some special functionality during the SCALES project. This example is using kernel methods (a special kind of moving window statistics) to assess connectivity. The kernel can be calculated for each grid cell but will summarise information from adjacent cells in a given distance. The kernel width and scale of this special analysis was 1, 2, and 4 km; the results of the assessment are maps indicating the connectivity for the whole region. The size of the moving window can be adapted to the requirements and dispersal distances of the species we focus on and determine whether the resulting maps look more smooth or scattered. Aggregations of cells with the habitat in focus get a considerably higher value compared to single isolated cells. This kind of connectivity assessment was not the final result in this study, but it was mainly used to integrate aspects of habitat connectivity into the spatial planning and the prioritisation of selected areas to protect the species under concern.

Ecological connectivity: The performance of protected area networks in Europe

In the last example, described in detail in the chapter from Papanikolaou et al. (2014 this book), we move to a different scale and level of available information. In their work, they present a methodological framework developed in order to evaluate the connectivity of protected area networks and their performance against future changes. The area of interest covers Europe, with protected areas of the Natura 2000 network representing the units at which connectivity is assessed. This analysis was restricted to three forest types (Boreal, Temperate and Mediterranean) and grassland recognised within protected areas. The basic goal of the study was to assess the efficiency from a network point of view, while taking into consideration both global changes and species traits. There were four objectives in the study: i) providing a framework for assessing whether the spatial distribution of an existing

Table 1. Percentage changes of area and patch number for the land use classes found in the Natura 2000 network of protected areas in the period between 2001 and 2030.

	Boreal forests	Temperate forests	Mediterranean forests	grassland
% change of area	-21.04	+14.10	+2.30	-16.80
% change of patch number	+0.87	-37.00	+12.21	-8.70

protected area network facilitates connectivity, ii) evaluating potential connectivity for different groups of birds, iii) assessing the insights provided by network analysis, based on graph theory, for conservation management decision-making, and iv) quantifying the impact of future spatial changes on the connectivity of each network.

The chosen approach uses birds as the key model species. Because they can cross large distances between habitat patches by flight, it may be possible to neglect the suitability of the matrix between the protected areas and take into account only the characteristics and distances between patches of protected areas to build a network of spatial relations, contributions in terms of area, and their functionality.

In contrast to the study on brown bears, neither detailed spatial data on species distribution, nor dispersal properties of species were available. To this end, background data as derived by LPJGUESS models (Lehsten and Scott 2014 this book) have been used, and connectivity was assessed for ten hypothetical groups of species representing different combinations of thresholds for dispersal distance and area requirements.

This generic analysis, although subjected to various limitations (e.g. use of hypothetical species, assumption that intervening sites are unsuitable, assumption that the landscape between interconnected nodes is homogeneous in terms of resistance or facilitation of dispersal) offered the opportunity to raise some basic insights on connectivity issues at a European scale.

A first interesting finding of this study actually provides critique to a long-standing notion that long-distance dispersal could favor connectivity; the authors actually found that, taking into consideration that minimum area requirement increases with body size, long-distance dispersers are facing a more serious risk with networks consisting of fewer sites and having a larger proportion of isolated sites.

Secondly, global changes are expected to cause shifts and alterations to the distribution of the selected habitat types within the Natura 2000 network between 2001 and 2030; but the different habitat types were predicted to be affected differently (Table 1). The spatial configuration of the Natura 2000 network might absorb future changes, maintaining or even enhancing macroscopic properties for some habitat types, while it might prove insufficient for others. Although we have to recognise the limitations of the study and support the need for better quality of data, it is important to note that this approach enables a user to introduce connectivity components into the assessment of the efficiency of conservation area networks. So far, any such assessment has been based primarily on the development of SDMs, largely ignoring the spatial complexity, structure and configuration of the protected areas, and the potential of species to disperse within favourable near-distant patches. Under this context, and based on our findings, the proposed framework could be advanced to provide insights for identifying key sites that may be critical to regional connectivity and inform conservation decision-making, providing insights on how to choose the most appropriate conservation plan and how to assess the coherence and ecological sufficiency of protected area networks based on the species of interest.

Maintaining and re-establishing connectivity: An outlook

The aim of the above examples is not only to analyse and show the current or future connectivity of the landscapes under concern; it is rather to reveal mechanisms, critical areas and gaps in our strategies to protect single species or species groups. It becomes clear that aspects of scale are not only relevant for the choice of the extent and spatial resolution of the study but also for the choice of the appropriate range for the analysis. A mismatch between the spatial dimensions of species perception, activity, or area demands, as well as their configuration and the focus of the methodological approach, should be avoided. This needs not only detailed biological information but also a wise selection and parameterisation of the appropriate technical methods to get meaningful results.

Such methods, and especially the resulting maps, are important instruments to support the communication between scientists, administrations, politicians and other stakeholders. They also serve as indicators and tools for an explicit spatial planning process to avoid further fragmentation and to develop efficient strategies towards improving habitat suitability as well as connectivity in the long run.

One possibility is offered by the reduction and/or concentration of recent and future technical infrastructure, large-scale habitat restoration based on improved policies for agriculture and forestry, concerted establishment or reallocation of areas with an appropriate status of protection (LIFE and Natura 2000), and the introduction and increase of Green and Blue Infrastructure. The findings of scientific studies with a focus on connectivity should be mirrored by an adequate consideration of scaling aspects in the planning processes. However, this is not enough, as we can see from the article of Kettunen et al. (2014 this book). Beside spatial and functional scales, the social scale also has to be addressed and interlinkages have to be taken into account to secure an effective implementation in the real world. Already existing policy instruments should be evaluated, as done by Paloniemi et al. (2014 this book), and improved in regard to their scale relevance.

Methodological approaches, as outlined in the examples, are still mainly limited to research projects. It still remains a challenge to integrate them with promising policies for securing connectivity in the real world. The SCALES project and the SCALES tool, as one of the main outcomes, have tried to turn this by providing data, results, experience in terms of guidelines and more easily usable tools for further analyses.

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Evaluation of policy instruments in promoting ecological connectivity

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Introduction

The need for scale-sensitive governance has been increasingly recognized in biodiversity conservation, especially during the last decade (Paloniemi et al. 2012, Young et al. 2013, Primmer et al. 2014). The current challenges for European nature conservation largely stem from the increasing habitat fragmentation (Hanski 1998, Giulio et al. 2009) which confirms the necessity of improving policies to more efficiently promote ecological connectivity. Also current European policies and practices need to be thoroughly evaluated and renewed in order to improve policy integration and synergies.

This book chapter contributes to the emerging literature on scale and governance (e.g. Cash et al. 2006, Apostolopoulou and Paloniemi 2012). Empirically, we focus on the possibilities of selected policy instruments to improve ecological connectivity by drawing on perspectives of researchers and practitioners from England, Finland, Greece and Poland. We begin by presenting the results of our multinational study, evaluating the possibilities of various policy instruments and approaches in promoting connectivity. Then we continue by focusing on the case of England, which enables the study of a policy instrument specifically dedicated to connectivity enhancement. Finally, we discuss the implications of the results for scale sensitive biodiversity governance across multiple scales.

Policy instruments and promotion of ecological connectivity

Policy instruments can be defined as "the set of techniques by which governmental authorities wield their power in attempting to ensure support and effect or prevent social change" (Vedung 1998, p. 21) or more broadly as the "myriad techniques at the disposal of governments to implement their policy objectives" (Howlett 1991, p. 2). Given the fact that not only specific arrangements provided by formal institutions, but also activity of informal institutions, as well as spatial, temporal and jurisdictional scales have an effect on the outcome of environmental policies (e.g., Paavola et al. 2009), policy instruments should be evaluated in the broader governance context. This need is reflected in the chapter as its principle goal. Moreover, it is intended to address current scale related challenges of biodiversity conservation (Paloniemi et al. 2012, Apostolopoulou and Paloniemi 2012), through covering a wide set of policy instruments.

Exploring the opinions of experts

To gather comparable results from a broad audience involved in the designation and implementation of biodiversity policy and relevant research across Europe, we implemented similar surveys in England (34 respondents), Finland (47 respondents), Greece (54 respondents), and Poland (44 respondents). The respondents were selected based on the level of their expertise, practical experience and influence on decisions regarding conservation, particularly ecological connectivity, at the national level. The respondents cover a variety of opinions and attitudes of both researchers and practitioners regarding the current performance of policy instruments in promoting ecological connectivity in the case study countries.

The aim of the survey was to explore policy instruments from various perspectives. To get an overall picture of the situation in each particular country, before asking individuals to evaluate specific policy instruments, we asked them to express their opinion on whether connectivity measures are an important aspect of the current biodiversity policy in their country. Moreover, because various instruments are in use in investigated countries, we analyzed the value of both existing and potential policy instruments in enhancing ecological connectivity and implementing biodiversity policy in each country separately (see Box 1). The respondents were asked to evaluate the importance and current performance of the policy instrument in promoting ecological connectivity *in practice*.

More precisely we focus on England, which has, at least discursively, taken connectivity into account more actively than the other case study countries, and at least one policy instrument – Nature Improvement Areas (NIAs) – does specifically aim to enhance connectivity. In England we conducted qualitative interviews to provide a more nuanced understanding of current connectivity conserva-

Box 1. List of policy instruments evaluated in Finland, Greece and Poland

- National spatial development strategy/plan(s)
- Regional spatial development strategies/plan(s)
- Local spatial development plans
- Biodiversity strategy
- Biodiversity law
- Strict nature reserves
- National parks (natural parks: national and regional parks)
- Habitat/species management areas
- Natura 2000 – Special Areas of Conservation
- Natura 2000 – Special Protection Areas
- Wildlife refuges
- Protected landscapes/seascapes -landscape parks, areas of landscape protection, nature-landscape groups
- Protected area with sustainable use of natural resources (areas of ecological use)
- Agri-environmental schemes/subsidies related to/aiming to support biodiversity conservation
- Other funding mechanism (respondents were asked to define)
- Ecological corridors
- Buffer zones around conservation areas
- Environmental impact assessments
- Networks of protected areas
- Green infrastructure (GI)
- Other policy instruments (respondents are asked to define)

tion. Based on the interviews, we present experiences about NIAs that are part of the large-scale conservation trend in England.

tive in practical promoting ecological connectivity in their countries.

In *Finland*, the policy instruments that were most appreciated in practice – i.e., Natura 2000, wilderness areas, national parks, permanent conservation contract in the current forest biodiversity program called 'METSO'

Results

Policy instruments for promoting ecological connectivity in Finland, Greece and Poland

We found that connectivity conservation has been taken into consideration in Finland, Greece and Poland only to a limited degree. In the survey, the most popular response was that connectivity measures are only a less important part of current biodiversity policies whereas the second most frequently used response in Greece and Poland was that connectivity measures are not important at all, and in Finland that they are relatively important (Figure 1).

In Figure 2 we present the five policy instruments that respondents evaluated as the most and least effec-

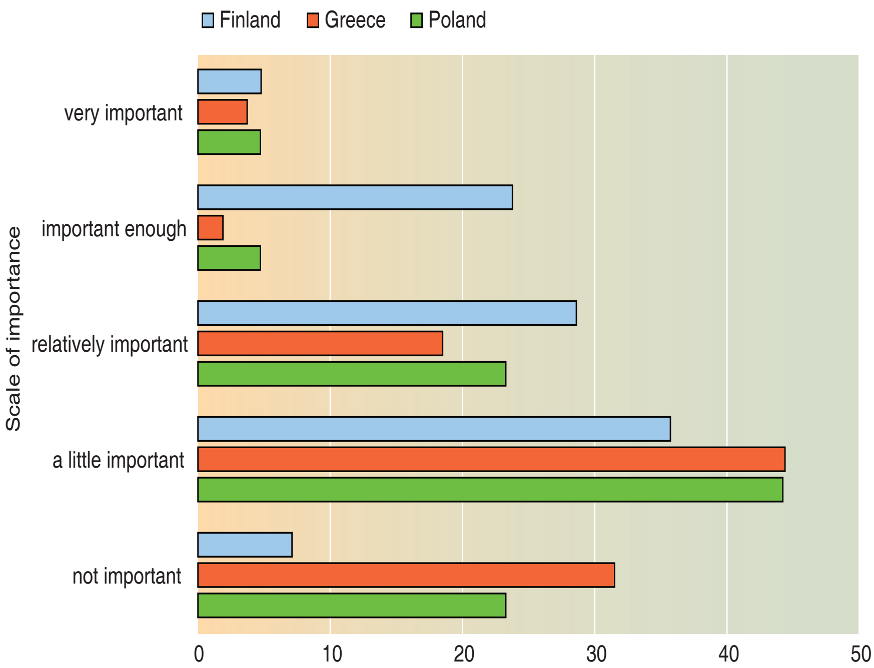

Figure 1. Importance of connectivity measures in biodiversity policy of each country. The scale used was the following: 1= not important; 2= of little importance; 3= relatively important; 4= important enough; 5= very important.

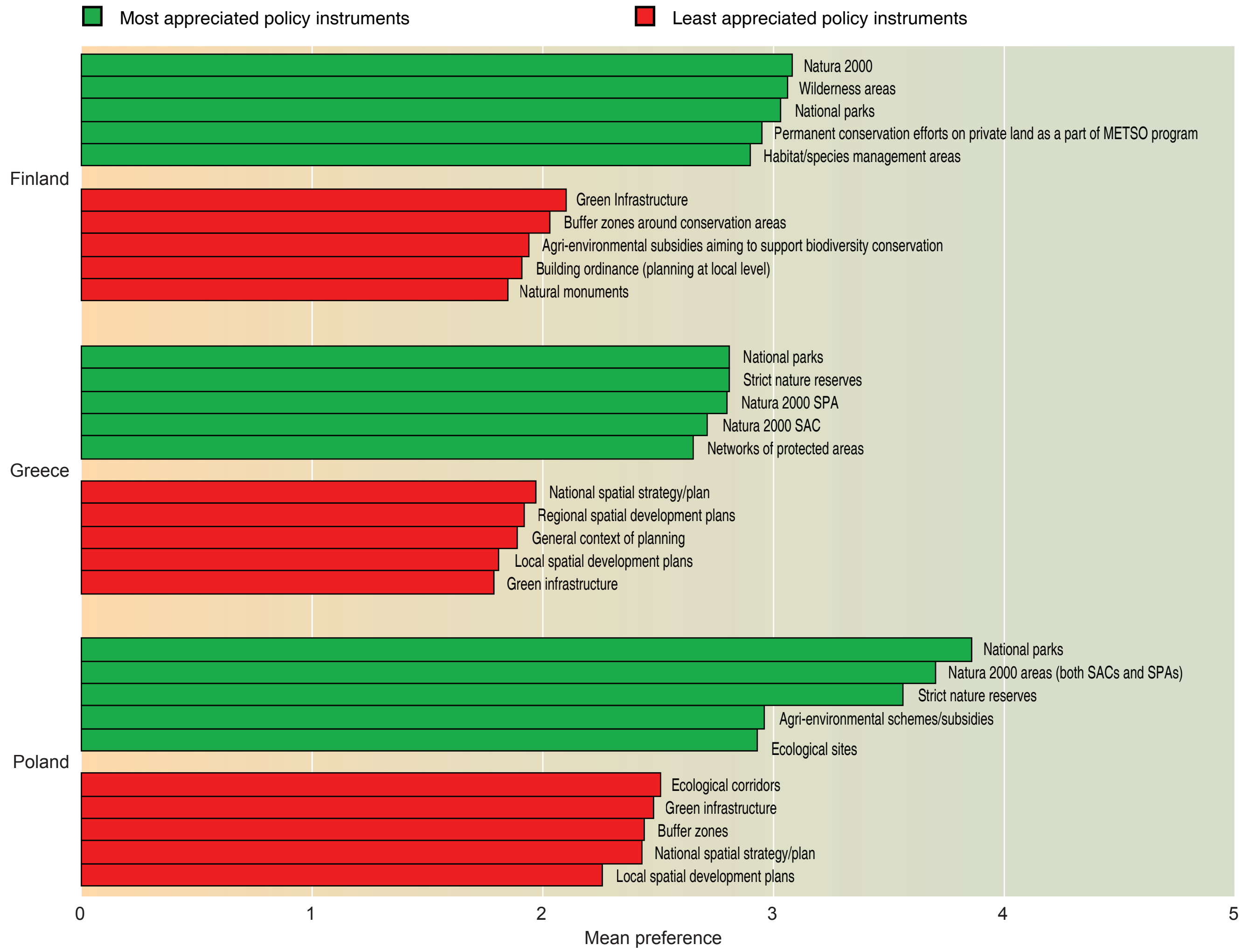

Figure 2. Current performance of selected policy instruments in promoting ecological connectivity in practice in Finland, Greece and Poland. The five *most* and five *least* appreciated policy instruments were evaluated with a scale: A= I do not know; B= instrument not in use; 1= unimportant; 2= of little importance, 3= moderately important; 4= important enough; 5= very important.

and habitat/species management areas – are all more traditional, 'command and control' type of instruments. In general, the four most appreciated instruments cover wide spatial areas and longer temporal scales than other evaluated instruments. The regulation of wilderness areas is less strict than that of other most appreciated instruments. The large spatial area included in wilderness areas and their location in northern Lapland, where biota is extremely vulnerable to the probable effects of climate change, can explain the perceived importance of the instrument. The least appreciated instruments — i.e., buffer zones, agri-environmental subsidies, building ordinance and natural monuments — are all operating on much smaller spatial scales. Green Infrastructure GI has not yet been implemented at all, buffer zones have not been used widely, and natural monuments have been established to protect only certain, small scale valuable items or spots, which explains the poor ability of these instruments in promoting ecological connectivity. It is worth noticing that the policy instruments designed primarily to protect certain small scale sites are not well adapted to promote ecological connectivity. Moreover, even though widely implemented agro-environmental subsidies could be useful in developing corridors or in encouraging conservation practices favorable to biodiversity and ecological connectivity, so far these subsidies have not been able to fulfill such promises (Arponen et al. 2013).

In *Greece*, the most appreciated policy instruments include both traditional types of nature conservation instruments, such as protected areas ranging from National Parks to strict nature reserves, as well as more recent instruments, such as the Natura 2000 network designed to enhance and restore connectivity through the establishment of a coherent network of protected areas. This is not surprising given that, due to the limited integration of biodiversity into other policy sectors, biodiversity conservation measures are being implemented mostly at species level or only within PA boundaries. The least appreciated are GI (which has not been implemented in Greece so far, except for few projects at local level) as well as spatial planning policies from local to national levels. The poor performance of planning policies is strongly related to the chronic criticism against spatial planning policies in Greece. The latter have been shaped by laws and plans focusing primarily on urban development and on the extension of statutory town plans (Sapountzaki and Karka 2001) whereas unauthorized development, especially residential, has been widespread and poorly controlled resulting in chaotic urban patterns and environmental degradation.

In *Poland,* policy instruments evaluated as the most effective for connectivity promotion are well-established (secured by law). These were: protected areas, and agri-environmental schemes (voluntary instrument with assured funding). The least important are forms of weak legal foundations, such as: ecological corridors, GI, buffer zones and national and local spatial plans. Buffer zones are, in the opinion of respondents, not very important, mainly due to their weak protection regime. National spatial development plans are not crucial due to their low compatibility with existing concepts of ecological connectivity, whereas local plans are simply not implemented in majority of municipalities. Moreover, there are still local plans which come into being not as a result of a sound planning process but because of instrumental reasons e.g. the need of realizing new commercial investment (Blicharska et al. 2011).

Nature Improvement Areas (NIAs): A case study of promoting ecological connectivity in England

In *England*, contra the other case study countries, the NIA approach shows significant potential regarding connectivity: on the conservation and ecological side, preserving larger areas has clear benefits for the viability of populations. Additionally such large-scale work is not easy to undertake by a single organization, and NIAs were correctly conceived as partnership projects. Despite the potential, however, there are also some misgivings inherent in the way NIAs were designed and implemented *as a policy*. First, from ecological and conservation perspectives, conservation should be done at multiple scales and "*there is no preferable scale*", as argued by an interviewee, even if that scale is quite large. The coherence of a network as a multi-scale attribute cannot be achieved without some co-ordination and planning at the national or county administrative and ecological scales. While the NIAs can provide significant benefits locally, there is still the question of how much they contribute to national level coherence, an attribute found lacking in the protected area network in England. The way the NIAs were selected could have exacerbated this problem, as it was through a funding competition between partnerships that did not consider network coherence as an important factor for the spatial allocation of the funds.

Conclusion

This study provides empirical evidence about the possible existence of various policy instruments to respond to the current scale-relevant challenges of biodiversity policies by focusing

Figure 3. Isokivenniemi, Natura 2000 site in Finland (photo: Terhi Asumaniemi).

Figure 4. National Park of Koroneia-Volvi, Natura 2000 site in Greece (photo: Evangelia Apostolopoulou).

on the promotion of ecological connectivity. We found that in Finland, Greece and Poland, the most appreciated policy instruments were Natura 2000 sites and national parks. These instruments have been designed with the explicit aim to protect biodiversity on the sites but they are also expected to work as core sites of a wider functional network improved by other instruments. What is relevant from the scale perspective, is that both instruments aim to cover large spatial scales: national parks constructing a base for nationwide networks and Natura 2000 doing the same at both national and European level.

The role and performance of policy instruments specifically targeted to promote ecological connectivity is still not clearly recognized in the studied countries. GI was among the five instruments whose current performance in promoting ecological connectivity was perceived as very limited in Finland, Greece and Poland. Even though GI has only been implemented in England and there are concerns over "*what it actually is*", it is already a key part of current EU strategy for biodiversity conservation. The limited acknowledgment of GI's potential reflects the need for better communication between EU and national levels for addressing the concerns regarding whether GI actually aims to promote ecological connectivity.

Moreover, the least appreciated instruments were also often, especially in Greece and Poland, related to spatial planning, reflecting the chronic problems in integrating biodiversity and planning in many EU countries (see, e.g., Apostolopoulou and Pantis 2009), highlighting the need to revisit them by promoting connectivity on larger scales and by explicitly reflecting conservation objectives at all levels. This is strongly related to the fact that in many cases the goal to integrate biodiversity to other policies or reconcile biodiversity conservation with development and growth leads to the underestimation of biodiversity objectives and to the opposite outcome: namely the integration of growth or development objectives into biodiversity policies.

However, the NIAs are a promising example of connectivity instruments used in England. They focus mainly on connectivity, but also take into consideration community engagement with nature, spiritual and cultural ecosystem services, biodiversity offsetting, payments for ecosystem services, and economic growth. NIAs should be seen in relation to the emergence of landscape or large scale conservation in England in the early 2000s. Emerging as a consensual and inclusive way of 'doing' conservation, landscape scale conservation flourished with the input of large NGO schemes such as the RSPB's Futurescapes into a national conservation imperative in the late 2000s. Multi-partner projects, across the civil society-markets-state spectrum got involved in this attempt to move

Figure 5. Agri-environmental subsidies in Beskid Żywiecki, Poland (photo: Joanna Cent).

away from habitat and species based conservation. Despite the success of several projects, like the National Character Areas, it was felt that a 'step change' was needed after 2010. NIAs embody this shift: the move from a 'third-way', 'win-win-win' and consensual way of protecting the environment *and* achieving growth, to a yet-to-be-assessed type of conservation policy that foregrounds competition among localities, heavy monitoring and evaluation, market-based conservation and economic growth. Tellingly, in the second year report, only two of the twelve NIAs even managed to assess connectivity in their area, the very issue they were designed to do.

Conservation networks are consistent with the aims of large-scale conservation, one of the central imperatives of global biodiversity conservation, to emphasize the importance of conserving entire ecosystems as opposed to patches of protected areas (Igoe and Croucher 2007). In the results of the survey some of the above-mentioned goals proved to be evaluated positively by respondents, such as the establishment of conservation networks, and some others were criticized as not very successful so far, such as the integration of connectivity measures to land use planning and development policies. This brings our attention to one of the most widely agreed aspects in the results of the common survey regarding policy integration: the opinion that any effort to improve ecological connectivity would be futile unless integrated into a wider framework that smoothly and efficiently co-ordinates land-use, balancing conservation, social and economic factors.

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Legitimacy of site selection processes across Europe: Social construction of legitimacy in three European countries

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Introduction

Legitimacy is an important element of governance processes affecting the overall institutional effectiveness (Biermann et al. 2009). In biodiversity governance and protected areas management, legitimacy is highlighted as one of the most important issues that enable proper conservation measures (Stern 2008). However, it can be either gained or compromised through the environmental decision making processes (see Brechin et al. 2002, Smith and McDonough 2001). Legitimacy relates to the whole spectrum of outcomes of decision-making processes and to their procedural aspects. Whether or not the process and its results are considered legitimate relies on numerous factors, starting from the accordance with legal basis, through the facts of who makes the decision and when and how the decision is made. Thus, legitimacy depends on the extent to which a decision is acceptable to the participants of the process (Adger et al. 2003), who is involved as a participant and how representative participants are for the actual stakeholders groups (Green 2010). It also involves the acceptance or the acknowledgement of suitable power to a certain decision-making authority (social approval of using power) (Wallington et al. 2008), which in turn requires an assumption of the desirable and proper action of this entity (Schlossberg and Shuford 2005).

In the following chapter we explore selected policy instruments and mechanisms that enable and facilitate legitimate and scientifically informed reserve site selection in three European countries: Finland, Greece and Poland. While former studies have often focused on the importance of factors improving legitimacy, or explanation of legitimacy deficits in single cases, the goal of our research is to explore the variety of ways in which legitimacy can be ensured, no matter which policy instrument is being used in a particular case. We analyse current deficits of selected policy instruments in the case study countries, by paying special attention to the possible unresolved conflicts and the limitations of systematic reserve site selection and management from the above described theoretical perspective of legitimacy. Thus, we investigate the capacities of selected policy instruments and institutional mechanisms to hinder or support adaptive application of science-based information in the selection of protected areas and their management across different administrative levels, in a broader context. More precisely we analyse how legitimacy of site selection processes is socially constructed and if there are any commonly shared visions of legitimacy and ways in which site selection process can ensure it.

Description of the study

In both Greece and Poland we analysed the Natura 2000 instrument. Additionally in Greece the process of the establishment of management agencies, institutions responsible for the management of selected Natura 2000 sites was analysed. In Finland the empirical focus was put on the policy development that has occurred after Natura 2000 implementation, namely on the forest biodiversity program METSO (to be implemented from 2008-2020) and protection of semi-natural wooded agricultural habitats that were safeguarded with the EU agri-environmental subsidies. The chosen instruments differ between the countries, but they all refer to site selection processes that are either still on-going or identified in previous qualitative research as prone to legitimacy deficits. Use and outcomes of the chosen instruments are actively present in current debates on legitimate and effective conservation in the case study countries (see also Grodzińska-Jurczak et al. 2014 this book).

In the three case study countries we concentrated on regional cases in order to refer to a specific regional and administrational context in the survey questions and select respondents that were actually involved in the same site selection process. In Finland we focused on south-west Finland, where METSO has been

Figure 1. Sheep grazing as an example of traditional, economic activity. Natura 2000 area "Beskid Żywiecki", Poland (photo: Joanna Cent).

implemented, slowly accumulating both permanent conservation areas and fixed-term conservation contracts. south-west Finland is the most intensive agricultural region in the country, and therefore agri-environmental subsidies are a relevant policy instrument supporting site selection processes. In Greece we focused on the Region of Central Macedonia where many important ecosystems protected by International, European and national laws and agreements are located. Central Macedonia contains 57 Natura 2000 sites covering about 46.5% of its total area, 3 wetlands in the Ramsar list and 4 management agencies. In Poland we focused on the Malopolska region with more than 60% of its territory covered by nature conservation areas, including 99 Natura 2000 sites (Figure 1).

The survey was conducted using a multi-mode approach – questionnaires were either sent by emails (in attachment or a link to web page with questionnaire form was provided) or conducted as face to face questionnaire interviews. The numbers of respondents and response rates varied between the countries (27 respondents in Greece with response rate 44%, 17 respondents in Finland with response rate 13%, and 42 respondents in Poland with response rate 43%). The number of respondents resulted from the particular focus of the survey on specific, regional case studies, as well as from the limited number of stakeholders and experts involved in the site selection process. The respondents were mainly practitioners (mainly employees of public administration bodies) and experts (including scientists). Respondents were carefully chosen based on previous empirical research conducted within the SCALES project, involving a desk study and focus groups.

Thirty seven aspects of legitimacy were evaluated by 86 respondents. The aspects were selected based on a literature review and a previous qualitative study conducted within the SCALES project. They represented several topics identified as the most relevant in the context of legitimacy: general aspects related to acceptance, fairness and participation; ecological aspects related to formal criteria of site selection and goals of designated protected areas, economic aspects, local perspectives on protected areas, governance factors, and other aspects related to administrative borders and property rights. Each aspect was evaluated on a 5 point scale of relevance for ensuring legitimacy of site selection processes. Principal Component Analysis (PCA) was conducted to identify the dimensions of legitimacy constructed by participants of site selection processes. The model represents different sources of legitimacy that are not determined by specific national characteristics, and the structure of the model is not equivalent to levels of importance of specific dimensions. There is a list of aspects that were excluded from the models as they did not fit the identified general legitimacy construct. Components were extracted with Varimax rotation. The model explains 62% of the total variance.

Figure 2. Participatory event in Poland (photo: Katarzyna Nieszporek).

Social construction of legitimacy: Main dimensions of legitimacy in site-selection processes

The model consists of six components, revealing a different vision of legitimacy than the ones presented in the preliminary topics derived from previous studies. The model consists of the following six dimensions that bring attention to new categories of procedural and governance aspects (Table 1):

1. Transparency and fairness,
2. Consideration of economic aspects,
3. Emphasis on science (positivism),
4. Power and decision making processes,
5. Influence on decisions at higher levels (EU-national) through formal participation,
6. Community and private ownership.

The model shows that the construction of legitimacy in the case studies is primarily explained by the aspect of **transparency and fairness** (28% of a the total variance explained by this component). This dimension, although including the overall acceptance by the general public in each of the studied countries, cannot be reduced solely to social acceptance of the site selection process *per se*. Other aspects that this component contains are: discussion of conservation goals and conflicting opinions as well as general transparency and availability of documents. This dimension also covers aspects relevant to local communities, such as: wide representation of various stakeholders groups, acknowledgement of the preservation of traditional practices and an offer of alternative livelihoods to affected groups. Participation of non-state actors, on the other hand, is required to be based on official rules, in order to ensure transparency of their influence. This result can be explained by the strong engagement of national and international NGOs in the Natura 2000 site selection and the perceived need to ensure the transparency of their role in the process.

The second dimension of the model is the **consideration of economic aspects of site selection and** refers to benefits that the site produces at local and national levels. In this dimension a need for establishing sites free from human influence is also acknowledged. Such areas do not, in fact, produce economic benefits; consideration of those aspects requires deliberation on the presence or absence of human influence at a site. In this case, if economic aspects are taken into account during the site selection process, then sites without conflicting human interests are potentially considered as more legitimate locations for Natura 2000 areas.

A positivistic approach to site selection represented by the **emphasis on science** forms another dimension of legitimacy. According to this dimension, designation of protected areas is legitimate when it is based on the presence of endangered or protected species and habitats and when it prioritizes biodiversity rich areas. It reflects the actual criteria of selecting Natura 2000 areas as prescribed in the Habitats Directive and also refers more broadly to the current debate on the role and definition of scientific assessments and evidence in site-selection processes. The positivist approach that this dimension possibly reflects is supported by the fact that other aspects included in this dimension are: the consideration of local communities as homogeneous entities and the underestimation of their internal diversity of interests and values. In combination, these aspects reveal an interpretation of the site selection process as a neutral scientific process which equally influences all social groups.

The role and influence of different stakeholders in the site selection process is described by the **power and decision making processes** dimension of legitimacy (Figure 2). It underlines the influence of the different positions in the ability to participate in the process as well as the need of ensuring the equal influence of various stakeholders and the democratic character of the process.

The possibility to **influence decisions at higher levels (EU-national) through formal participation** is another component of legitimacy. It reflects the need for formal support for local stakeholders, or stakeholders others than NGOs, in order to enable their influence on decision-making processes at national and EU levels.

The last identified legitimacy dimension relates to **community and private ownership** issues and it is constructed by the consideration of

Table 1. Legitimacy dimensions – components of PCA. Component scores smaller than 0.4 are not displayed. Kaiser-Meyer-Olkin Measure of Sampling Adequacy is 0.68, Bartlett's Test of Sphericity statistical significance < 0.005.

	Component					
	1 Transpa-rency and fairness	2 Consi-deration of economic aspects	3 Emphasis on science (positivism)	4 Power and decision making processes	5 Influence decisions at higher levels through formal participation	6 Community and private ownership
Decision-making process is transparent	0.767					
Alternative approaches regarding the goals/aims of the process are acknowledged and discussed equally	0.702					
Conflicts of opinions are acknowledged and discussed during the site selection process	0.694					
Local communities are represented by a diversity of stakeholders who can freely participate	0.684					
Maps and borders of proposed sites are publicly available	0.675					
Management plans are officially open to the public after the planning process	0.649					
Preservation of traditional practices is considered in the process	0.642					
Non-state participants are selected based on officially defined rules/criteria	0.612					
Alternative livelihoods are being offered for social groups affected by the site selection	0.514					
The site selection process is overall accepted by general public in the country	0.431					
The process considers economic benefits the site produces (before delineation as protected area) for local/municipality budgets		0.807				
The process considers economic benefits the site produces for general development prospects of the country		0.724				
The process considers economic benefits the site produces for local development prospects		0.693				
The process considers economic benefits the site produces (before delineation as protected area) for local enterprises/industries		0.687				
The process considers economic benefits the site produces for enterprises/industries other than local		0.662				
One of the main goals of site selection process is to establish some conservation areas free from human influence		0.481				
The main goal of site selection process is the establishment of protected areas of high biological diversity			0.751			
Site selection process is mainly taking into account protected species and habitats			0.720			
The process approaches local communities as homogeneous entities without internal differentiation of interests and values			0.484			
All relevant stakeholders can equally influence the decisions				0.831		
Decisions are generally made in democratic way				0.745		
Power position of each stakeholder or social group affects remarkably how this stakeholder/group can participate in the process				0.457		
Stakeholders have possibility to participate in formal way at EU level					0.915	
Stakeholders have possibility to participate in formal way at national level					0.903	
Administrative borders of municipalities and regions are considered in the site selection process and affect the borders of the new sites						0.894
Borders of land ownership are considered in the site selection process and affect the borders of the new sites						0.869

Figure 3. Recreation in Natura 2000 area “Dolina Dolnej Skawy”, Poland (photo: Joanna Cent).

the borders of land ownership and administrative borders when deciding the borders of the protected site.

The presented PCA model (Table 1) includes 26 out of 37 variables preliminarily used in the analysis. Consequently, in this case the analysis can reveal only those legitimacy constructs that are based on questions asked in the questionnaire survey. Therefore, it is also informative to consider which questions are not included in the model. In this case, these were aspects related to the following statements: (1) everything occurs, in general, fairly, (2) local communities are being considered in the site selection process, (3) the site selection process is overall accepted by the majority of local people, (4) conservation costs are allocated fairly among different social groups, (5) conservation benefits are allocated fairly among different social groups, (6) local communities are represented in the process by officials, who know the interests and values of local people, (7) non-state participants are selected based on unofficial practices/criteria of responsible institution(s), (8) non-state actors participate in the process and their participation has a significant influence on its outcomes, (9) management plans are officially open to the public for commenting during the planning process and comments affect on the plan, (10) stakeholders have the possibility to participate in formal ways at local/regional level, (11) financial compensations are being offered for social groups affected by the site selection and (12) decision making process is accountable – there is information about and justification of past or future actions and decisions, and in the case of eventual misconduct those responsible are being punished.

According to the model these aspects are not a part of a shared variety of coherent visions (social constructions) of legitimacy. The majority of respondents evaluated these aspects as ‘rather important’ and ‘important’ for ensuring legitimacy of site selection processes.

The survey of this study was directed to actors involved in the chosen site selection processes as practitioners and experts. The role of actors was not limited to the selection of a single site and thus it may reflect a broader experience about site-selection processes. In practice however, legitimacy of site selection processes is dependent on a broader range of stakeholders, including local residents, land-owners and land users. Even if the involvement of local actors is recognised in our study as relevant for ensuring legitimacy, their visions of legitimacy can potentially differ from those shared by experts and practitioners.

Policy recommendations

Several dimensions need to be considered to achieve legitimacy

Our study illustrates that legitimacy is a far more complex issue than the simple acceptance of protected areas – it considers also the way decision-making processes or local dynamic circumstances, such as land use and property rights, are handled. There is no single way to ensure legitimacy across different countries, regions and sites. As revealed in the presented PCA analysis, social dimensions of legitimacy relevant at regional and local levels are related to democracy, transparency and property rights, to power relationships between actors, to the possibility of influencing the process at higher administrative levels, as well as to factors related to the economic and scientific aspects of site selection. In order to avoid conflicts escalation and to ensure legitimacy of site selection, it is relevant to consider how governance settings and decision-making rules deal with these aspects. Ensuring one dimension of legitimacy neither necessarily leads to the overall acceptance of the process, nor to the improvement of other dimensions. A wide approach, acknowledging the variety of legitimacy facets in site selection processes would emphasise simultaneously the procedural and distributive aspects of legitimacy (e.g., Tyler 2004, 2006).

Scale issues in legitimacy

In order to increase legitimacy, more attention should be drawn in both the formal and informal aspects of participation at all administrative levels. Formal and equal participation is needed to facilitate local actors’ influence at higher levels of decision-making, as they often do not have the capacity to communicate with decision-makers at national or EU levels through informal channels. This issue also reflects problems of power aspects that potentially can be handled with formally designed, transparent and open forms of consultations and participatory programs.

Case-specific determinants need to be taken into account to develop legitimate processes

Not all aspects are *a priori* consistent or able to be maximized at the same time. It is highly questionable whether the outcomes of a site selection process which emphasizes the role of science and experts will lead to the same outcomes as a site selection process emphasizing, for example, economic aspects. This may lead us to conclude that first, each process is set within a certain institutional and policy context (Adger et al. 2003) and second, that stakeholders are affected by their previous experiences, policy beliefs or normative assumptions (Connelly et al. 2006). Perceived aspects of legitimacy are thus differentiated not only among the case study countries but also within each country (e.g. between stakeholder groups), and such diversity of perceptions should be taken into account and handled as a part of conservation processes.

A need for stronger emphasis on transparency, democracy and equity

What is also important is that our study pinpoints the fact that the social construction of legitimacy is explained first of all by transparency, democracy and equity aspects. This notion can be explained by social science theories; however, it is almost unrecognized by the current European conservation policies which focus on ecological criteria in site selection and on market-based instruments in supporting management practices.

Figure 4. Historical site in Natura 2000 area "Beskid Niski", Poland (photo: Joanna Cent).

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SCALETOOL: An online dissemination and decision support tool for scaling issues in nature conservation

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Technological innovations have revolutionised communication and dissemination of information, impacting virtually all aspects of modern life. This revolution has not halted in front of science or applied biodiversity conservation. For centuries, communication in science relied primarily on printed media. Whereas printed media still play an important role, they have undergone considerable technological innovations.

The revolutionary changes in science communication started with the availability of information without the traditional limits of time, distance, and availability of copies that depend on the exchange of physical products like letters, scientific journals or books. The availability of the Internet, and later the layout oriented PDF, have removed further limits in scientific communication and dissemination. PDF is still the most appropriate format to publish results if layout is an important issue. However, pdfs are static and additional needs emerged to present data and tools and to link the latter with databases, geographic information or multimedia. Also, with the hugely growing information, the need to extract and summarize information with searching machines became increasingly important in science. Hence, other formats emerged, for example enhanced HTML, linking to sources external to the published article, embedded multimedia, dissemination of content at sub-article level, data publishing and so on.

Articles as containers of scientific results, even in the most up-to-date semantically enriched forms, have their own limitations, especially if the users want to re-use data or implement methods to test and reproduce the results, adapt them to own purposes, or enhance their applications in "real" life, such as in decision-making and management for nature

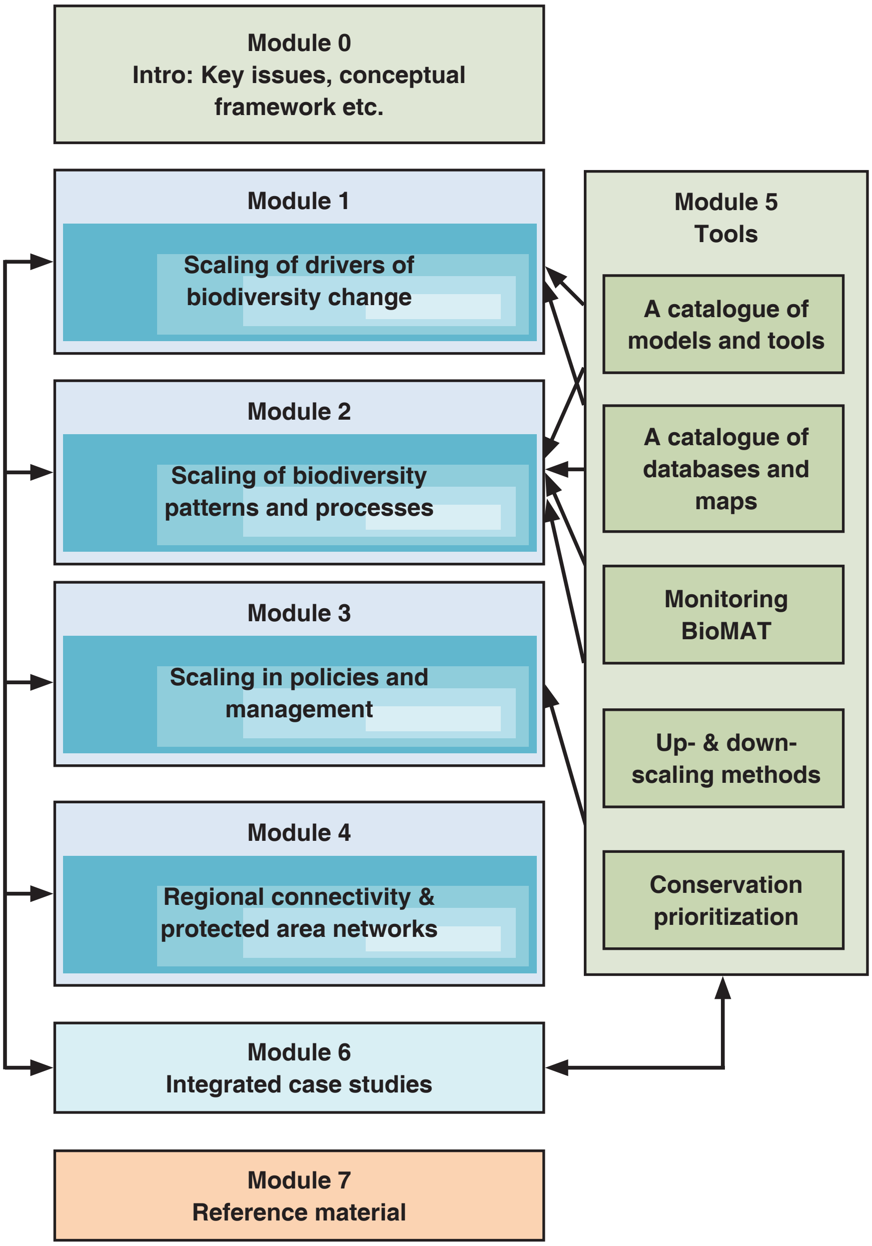

Figure 1. Structural diagram of the SCALETOOL.

conservation. All these new challenges can be solved only with web services (O'Reilly 2005). Additional obstacles for better use of science results in practice are the need to translate academic language into practical toolkits, policy recommendations, and guidelines.

In addition to innovations in scientific communication via print and online media, the Internet has taken over a dominant role in human communication at large. The Internet also plays an increasing role in science for communication and dissemination. Most large research projects use it regularly, mainly through creation of static or dynamic Internet portals that are aimed at implementation and dissemination of scientific discoveries to decision makers and practitioners. The SCALES project (Henle et al. 2010), on which this book is based, is no exception. Besides the portal, it designed an embedded web-based tool, called SCALETOOL, to help a wide range of users understand and explore scaling issues in biodiversity research and conservation. It contains recommendations for scale-conscious management practices and policy, methodological tools and advice, and web-links to relevant databases that are freely accessible. The overall aim is to improve biodiversity conservation across Europe and beyond.

SCALETOOL complements this book by taking advantage of features that web-based systems readily offer, but are difficult or even impossible to implement in printed media. It allows a broad audience to investigate specific scaling issues of biodiversity and its management by providing an interactive format. Users can explore simple and general models, as well as specific tools for conservation biology, active links to background literature and other related material. It further provides access to and enables searches of relevant databases and allows creation of a range of maps for drivers and biodiversity patterns.

Like this book, SCALETOOL is structured in a modular way, reflecting the overall structure of the SCALES project itself (Figure 1). The aim is to provide a readily understandable structure, allowing easy and intuitive navigation, while providing linkages between related topics and between scientific results and practical application. Users with particular interests should be able to identify their relevant entry points on the SCALETOOL main page (Figure 2) and follow the structure to their topic of interest.

SCALETOOL covers the scaling of **drivers** of biodiversity change, scaling of **biodiversity patterns and processes** from the genetic level through to traits, population dynamics, and selected ecosystem functions, and scaling of **policies and management** across administrative levels, from local levels up to the European Union. For instance, SCALETOOL allows exploration of the intensity and evenness of changes in a large number of drivers at different administrative levels (Figure 3). Furthermore, it provides access to **tools** (methods, software, and protocols) and **data** that allow the user to assess or visualise scaling effects. Data compilation comprises species-specific trait data, life history, and functional trait data for plants, insects, reptiles, and birds across Europe. Also, SCALETOOL presents methods that facilitate accounting for connectivity assessments, scaling issues in conservation, and simulating ecological processes. It further allows, via the link to the BioMAT tool, to create overviews on, and explore characteristics of, biodiversity monitoring activities at subnational, national, and international levels.

Figure 2. Screenshot of SCALETOOL (http://scales.ckff.si/scaletool/).

Figure 3. Sample page of SCALETOOL showing the spatial structures and trends in the distribution of drivers across administrative levels in Europe (http://scales.ckff.si/scaletool/index.php?menu=1&submenu=0&pid=5&nut=0).

The SCALES project particularly focused on applying the tools and databases to questions of **regional connectivity and protected area networks**. Based on reviews and discussions of measures of connectivity and concepts of protected area networks, guidelines are presented for researches and practitioners. **Case studies** illustrate some of the methods particularly suitable for studying scaling issues or providing insights into the relevance of scaling for biodiversity management or policy. They further integrate approaches used to improve our understanding of drivers, biodiversity patterns and processes, and policy and management responses to biodiversity change at different scales. They were performed at regional (subnational), national, and EU levels in order to provide results and recommendations on how these tools may provide inputs into the assessment and management of biodiversity at different administrative levels and on the effectiveness of policies. National case studies focused on Finland, France, Greece, Poland, the UK, and Taiwan.

SCALETOOL provides core messages from these studies, recommendations for policy and management as well as for analyses and models that are useful to perform under scaling perspectives. It includes links to external programs that are suitable for such analyses. It provides an overview of, and full access to, results from the SCALES project. SCALETOOL can be accessed via http://scales.ckff.si/scaletool/. It is planned to maintain the tool well beyond the end of the SCALES project. A full overview of publications from the SCALES project may be found on the SCALES project web-site (http://www.scales-project.net/online-library.php?P=4&SP=33&?P=4).

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CHAPTER VII

Concluding chapter

Lessons learned

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General conclusions

Biodiversity and its effective management are inextricably related to scale. The main pressures on terrestrial biodiversity (i.e. habitat loss and fragmentation and climate change, Settele et al. 2014) and the socio-economic drivers behind these pressures act differently at different scales. Effective conservation measures must thus explicitly consider the scales at which these pressures have their effect, since non-linearities may prevent the use of simple scaling rules. For example, both habitat loss and climate warming might make some species locally extinct, but species may be lost from the broad-scale landscape much more slowly than from local habitat patches because of the spatial patterning of habitats and microclimate. Interactions among such pressures, both among and within different scales, may exacerbate the challenges for biodiversity management. Consequently, effective policy interventions may need to be scale-sensitive, employing appropriate governmental levels for planning, decision-making and management. The contributions to this book illustrate these issues with selected examples from the SCALES project. They highlight the scale-sensitivity of drivers and their effects on biodiversity across scales and of processes that govern patterns of biodiversity at different scales. They show promising methodological approaches for the study of scaling issues in ecology and conservation biology. Importantly, they also derive implications for policy and biodiversity management at different scales and assess whether, and if so to what extent, current governance approaches to and policy instruments for the conservation of biodiversity consider scaling issues. Case studies demonstrate how these issues can be tackled and integrated in biodiversity conservation.

There were two main applied goals of the SCALES project (Henle et al. 2010), securing coherent networks of protected areas and ensuring effective connectivity across landscapes at different scales. A first lesson learned in relationship to connectivity was that while connectivity of habitat patches is often beneficial for nature conservation, there are also risks associated with it. These benefits and risks, and thus recommendations, depend on the level of biological organisation addressed. In particular, connectivity issues need to be considered differently for species with different dispersal potential. A dispersal database and taxon-specific trait databases help identify at which scales connectivity planning and decisions should be made in order to match the requirements of target species.

The contributions to this book also illustrate that a better understanding of the biological effects of connectivity on different types of organisms and biological levels of organisation can only be a first step towards better management practices and more effective biodiversity policies. This book impressively shows that there are still a number of challenges to the integration of existing knowledge in planning and decision-making for regional connectivity and the design and management of networks of protected areas. The contributions also highlight that the efficiency of networks of habitats depends on the spatial and temporal scales at which we assess them. While for some species a given network may be efficient and likely will retain this efficiency under future land use and climate change, for others, present and projected future efficiency is limited (Mazaris et al. 2013). For all but the most dispersive species there is often insufficient coherence of existing habitat networks. Therefore, the conservation of most species also depends strongly on sufficient quality of the habitat and quality of the matrix surrounding suitable patches.

Finally, a range of policy instruments and governance structures have been identified that may facilitate the use of such knowledge. In addition, methodological tools and databases have been developed or extended that facilitate planning for connectivity and coherence of networks of protected areas and of other sites relevant for biodiversity conservation. In particular, databases on dispersal potential and minimum area requirements have been compiled from case studies and methods were developed to allow extrapolation to unstudied species.

Connectivity and scale from an ecological perspective

Connectivity is an abstract property of networks. The natural scale for considering ecological connectivity is that of the metapopulation or metacommunity. A metapopulation is a group of local populations of a species that are spatially separated, reducing but not completely abolishing individual dispersal between them; similarly, a metacommunity is a set of multiple species' metapopulations. Connectivity here is defined as the determinant of the level of such movements. A "metapopulation scale" is therefore ecologically framed and implies a spatial extent many times larger than a typical population patch, containing patches that are separated by less than the maximum dispersal distance of the respective species. The geographical scale of this will therefore depend on the species of concern and the distribution pattern of their habitats. To facilitate the analysis of connectivity in regard to species traits and habitat patterns, the SCALES project has collated a dispersal database that contains dispersal data for a range of different species (http://scales.ckff.si/scaletool/index.php?menu=6).

Viability of metapopulations may be considerably greater than that of single populations. Moreover, it has been demonstrated within the SCALES project that metapopulation viability depends jointly on a species' demographic and dispersal traits as well as on landscape structure and composition. These findings are a major step towards more biological realism in connectivity research and planning. However, while many approaches and tools have been developed and applied to assess connectivity, the level of detail appropriate for specific questions, landscapes, or species remains less clear. Further constraints result from data deficiency. Thus, a major challenge for connectivity research is to identify the appropriate methods and metrics for a given objective or application and to increase data availability on species dispersal spanning all major species groups.

Connectivity can be promoted either by creating corridors or intervening "stepping stones", or by abolishing barriers, so as to enhance the permeability of the non-habitat matrix between local populations and enable individuals to move between habitat patches more regularly and reliably. Such movements typically constitute dispersal, but pollination and other foraging movements may also be facilitated, and may allow genetic exchange. Another means of increasing connectivity is the creation of additional patches of habitat to reduce the distance between patches.

Despite the importance of connectivity, it may have risks, such as by facilitating the spread of diseases or predators (Henle et al. 2004). Similarly, the overall species diversity of metacommunities can be enhanced if different species persist in different places, which again is favoured by some degree of spatial isolation. The SCALES project explored the benefits and risks of the spatial arrangement of conservation effort on a range of conservation objectives (Table 1).

Networks of protected areas: Efficiencies and gaps

Protected area networks have been set up worldwide to preserve biodiversity. The Natura 2000 network established under the 1992 Habitats Directive represents the cornerstone of nature conservation in Europe. Natura 2000 has now become the largest conservation network worldwide. Across the EU, more than 26,000 terrestrial sites corresponding roughly to 17.5% of EU terrestrial territory are covered by Natura 2000

Table 1. Benefits and risks of connectivity for different conservation objectives.

Conservation objective	Benefits of connectivity	Risks of connectivity	General rules
Maintenance of genetic diversity	• Reduces inbreeding depression • Increases selection opportunities for strongly beneficial alleles	• Reduces diversification through neutral evolution (drift) • Reduces likelihood of novel alleles establishing for local adaptation (outbreeding depression)	• Low rates of dispersal can still have important effects on genetic processes (Arenas et al. 2014 this book), so connectivity may be defined using lower migration thresholds than for other processes
Viability of populations	• Can create metapopulations out of isolated populations, reducing stochastic extinction risk • Enables mobile individuals to escape from local disturbances • Facilitates range shifts in response to climate change	• May reduce viability of metapopulations by collapsing them into simple populations, increasing stochastic extinction risk • May facilitate spread of endemic or epidemic diseases and predators	• Benefits of connectivity are generally assumed to outweigh risks • Number of connections should be proportional to size of patches (Figure 1) • Connections that traverse broad-scale temperature gradients are important in the context of climate change (Settele et al. 2014) • Connections themselves may constitute important habitat and so effectively increase patch sizes (Klenke et al. 2014 this book)
Community diversity	• Benefits for population viability (above), plus: • Facilitates arrival of new species, potentially increasing local (α) diversity.	• Risks for population viability (above), plus: • May facilitate invasions by competitive species, decreasing overall (γ) diversity and threatening specialist species • Long-range connectivity may decrease among-site (β) diversity	• For habitat-specialists, connections should be of comparable quality to core habitat • Edge (ecotone) habitats are crucial for many species • Solutions should consider the needs of taxa most prone to habitat fragmentation
Enhancing ecosystem functions and services	• For any keystone species, see benefits for population viability (above) • Resilience may increase with local diversity (see benefits for community diversity, above)	• For any keystone species, see risks for population viability (above) • Resilience may increase with among-site-diversity (see risk of long-range connectivity, above)	• Should consider disservices (pest species/communities) as well as services • Connections can also serve as a distribution network for service provision linked to particular habitats (e.g. pollination, biocontrol, aesthetic services) (Steffan-Dewenter et al. 2014 this book)
Overall	• Incorporating community diversity with population viability, moderate connectivity has many benefits	• Valuing neutral genetic diversity, or certain ecosystem functions, may entail more risks from connectivity	• Often intermediate levels of connectivity perform best, but the optimum level will depend on the mobility of the species, the amount of area required to maintain viable populations, and the aspect of conservation concerned (Marsh et al. 2014 this book)

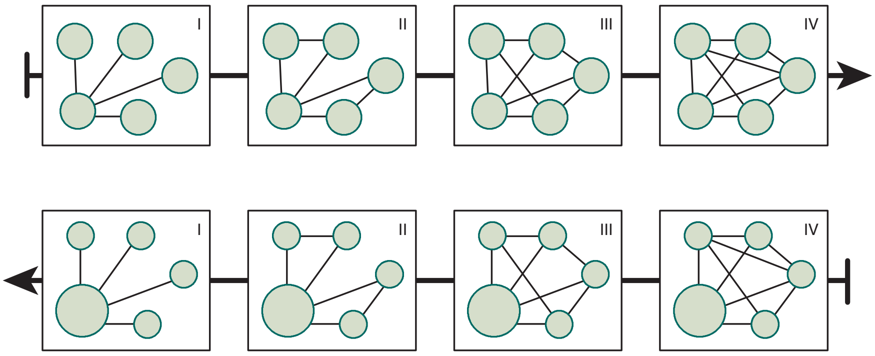

Figure 1. Relationship between areas size and optimal connectivity (after Frank 1998). Mean time to extinction increases in the direction of the arrow for different degrees of connectivity (scenarios I to IV) when sites are of similar size (upper graph) and when sites are of different size (lower graph).

Figure 2. The Natura 2000 network of Europe is the largest network of protected areas worldwide. Source: http://www.eea.europa.eu/data-and-maps/data/natura-5#tab-gis-data

(Figure 2). Despite this, Natura 2000 areas are often selected based on the restricted set of species listed in the Annexes and even many Annex species are not adequately represented (Gruber et al. 2012). Nevertheless, there may be good strategic-political reasons not to attempt to change the list (Maes et al. 2013). In any case, the network coverage is too small to serve the minimum area requirements of some larger species (Figure 3).

Despite successful establishment of the Natura 2000 network, maintenance of these sites remains challenging. One important consideration is that management authorities have been established for only a limited number of sites, and thus not all sites of the network can be equally protected. Associated with that, prioritisation is not only an issue of management, but also of money allocation. Member States have actually established sub-networks within their national Natura 2000 network that are under specific management protocols, making organisation and logistics even more complicated.

Another important issue is that monitoring tends to focus on more prominent species, while data are scarce or outdated for smaller and less "charismatic" species. Hence, the coverage of the network for those groups is difficult to determine (Trochet and Schmeller 2013). Insufficient or outdated data is an important problem that appeared during site selection as well as on-going management deliberations and needs to be solved.

Recent criticism of Europe's Natura 2000 network further relates to the fact that these sites were not established based on a true "network" plan, but rather as a set of separate, often disconnected, sites that merely represent species currently present. An important question is, therefore, to what degree these sites are effectively interlinked. Namely, can organisms move between fragmented sites? This is of particular importance for poorly-dispersing species because they are found disproportionately often on fragmented sites. Thus, the functional connectivity of a network (specific to particular species present within it) should be taken into account. It may be that networks are unsuitable for the conservation purposes they should serve because they depend on an unprotected matrix being sufficiently suitable as habitat.

The matrix between protected sites is rarely considered in analyses of protected area networks. However, natural and semi-natural habitats outside protected sites can serve as an important buffer supporting the populations of species and the movements among them and thus facilitating both connectivity and the overall viability of metapopulations and species in fragmented landscapes. Furthermore, interactions with agriculture are of high importance in times of agricultural intensification and land abandonment.

Drivers that may change species' distributions and population viability

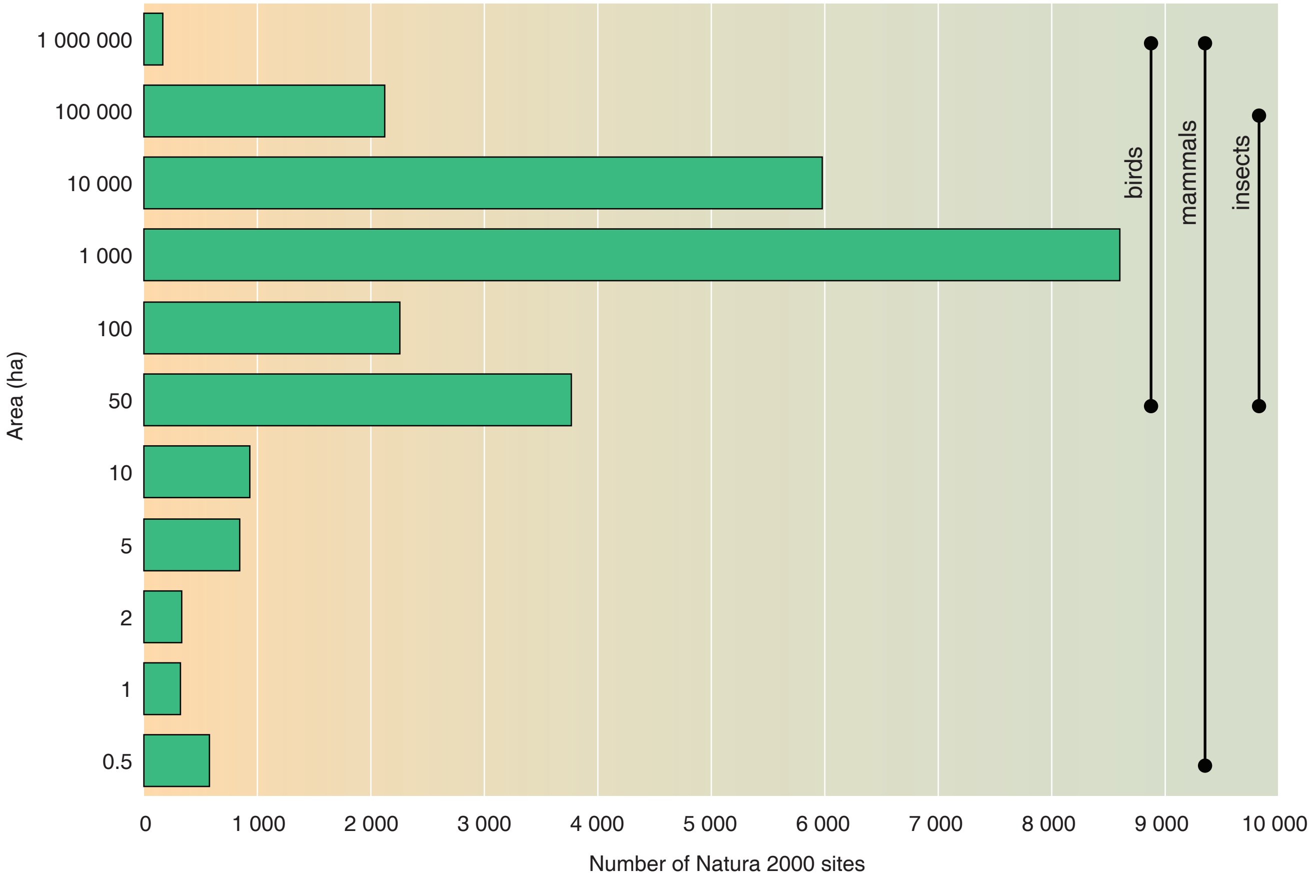

Figure 3. Area size distribution of the Natura 2000 network and minimum area requirement ranges of different taxa. Data source: Minimum Area Requirement database of the SCALES project: http://scales.ckff.si/scaletool/index.php?menu=6&submenu=1, and Natura 2000 data – the European network of protected sites (http://www.eea.europa.eu/data-and-maps/data/natura-2000).

were disregarded during the design of the Natura 2000 network. Even under the assumption that current networks of protected areas would adequately conserve natural habitats, climate change and associated land-use changes are anticipated to alter a range of habitats and landscapes in the near future. As a consequence, neighbouring sites (those falling within the dispersal range of a given species) that currently protect a sufficient area of suitable habitat to support viable populations of species might no longer contribute to overall network connectivity if climate change eliminates some habitat, or if landscape resistance between sites increase to further inhibit movements (Papanikolaou et al. 2014 this book). Likewise, new sites may facilitate connectivity in the future if land-use changes enable populations to expand into or through such favourable habitats. Previous studies have already shown that climate change and land use changes could significantly alter the effectiveness of conservation networks (Mazaris et al. 2013). These studies have attempted to quantify these sources of threat by providing evidence on the number of sites becoming unfavourable for species persistence.

Within the SCALES project we developed and tested several alternative methodologies to contribute to these primary conservation issues. Regardless of the method used, the overall outputs suggest that in order to mitigate the impacts of global changes we should focus on the application of common but flexible practices across larger regions, towards enhancing the transition from site-specific management to the consideration and assessment of how multiple factors interact across large spatial extents to affect population viability and connectivity (Klenke et al. 2014 this book).

In Table 2 we summarize key scale-related issues that need to be accounted for in management and policy to enhance the coherence and efficiency of networks of protected areas.

Driver analysis as background for policy design

Drivers provide a conceptual framework to link social, economic and environmental changes. Drivers allow us to determine how combinations of different factors affect our ecosystems either directly or indirectly. Understanding the concept of drivers, and embedding it in decision-making, is a key advance in designing polices that address environmental and ecosystem degradation.

The characterisation of drivers and their scale-sensitivity, and whether their effects change across scales, enables us to make some general conclusions. The work of the SCALES project underlined that direct drivers (e.g. afforestation, deforestation, agricultural conversion) are more scale-sensitive and that policies aimed at reducing the effects of these drivers must address multiple scales. However, indirect drivers (e.g. age structure, employment

Table 2. Scale-related issues for assessing and improving the coherence and efficiency of networks of protected areas.

Understanding network properties and efficiency
• Differences in the spatial structure of the Natura 2000 network among countries favour different taxa and different ecosystem services, and an optimal solution for a broad range of species, as well as ecosystem services, will require compromises
• Multiple species traits need to be considered to allow an assessment of species persistence and thus network efficiency (Papanikolaou et al. 2014 this book)
• The lack of specific criteria and principles for the selection of priority sites (expansion of networks and allocation of resources for management) could have a serious effect on the performance of any such network. Such criteria and methods that allow accounting for different goals at different administrative levels were developed within the SCALES project (Schmeller et al. 2014 this book, Arponen et al. 2014 this book)
• Establishment of standards for management of the Natura 2000 network, monitoring, and reporting among all Member States is critical for increasing conservation efficiency; these standards need to account for appropriate scales (Lengyel et al. 2014 this book) as well as ecological differences and priorities among Member States and the local socio-economic context (Kosztyi et al. 2014 this book)
Future steps in analysis and policy
• Promote the availability of data and tools
• Promote and communicate the need for integrating connectivity (Arponen et al. 2014 this book, Klenke et al. 2014 this book) and population viability assessments (Pe'er et al. 2014a this book) in protected areas selection and management of biodiversity priority sites
• Establish a flexible framework for site selection by considering sites serving as stepping stones or "key habitats", taking advantage of roadless-wilderness areas towards connecting or expanding the boarders of existing sites (Pe'er et al. 2014b this book, Klenke et al. 2014 this book)
Quality of the data (preferably point data) used for the designation of protected sites is critical (Touloumis and Pantis 2014a this book)
Apart from the site-specific parameters of the protected sites for network coherence, their spatial properties, structure, and configuration for species of different mobility should be considered in future planning and prioritization (Papanikolaou et al. 2014 this book)
Reducing policy and management gaps
• Considerable attention should be given to tools, personnel and personnel's expertise for understanding and addressing main scale related conservation challenges
• The lack of accurate, widely available and updated information on biodiversity components (e.g. data on less popular species, data needed for population viability analysis) represents a serious gap in existing knowledge on the efficiency of the Natura 2000 network and on future planning for a range of different taxa – but see the Minimum Area Requirement and Trait Databases collated by the SCALES project (http://scales.ckff.si/scaletool/index.php?menu=6)
• A national responsibility based approach could contribute to effectively reduce gaps in conservation networks simultaneously at Member States and EU levels and could assist in preventing biodiversity loss (Schmeller et al. 2014 this book)
• State funding for applied conservation should significantly increase, accompanied with clear conservation targets in conservation policy; innovative instruments, such as ecological fiscal transfer can account for costs and benefits that accrue at different administrative levels (Santos et al. 2014 this book)
• The role and objectives of Green Infrastructure (GI) should be communicated across the Member States and GI needs to be developed as a strategic approach to biodiversity conservation (Kettunen et al. 2014 this book)

rates) are less scale-sensitive and maybe adequately addressed at relatively coarse scales. In cases where known non-linearities across scales exist, drivers should be addressed at multiple administrative levels and scales, including temporal scales.

By understanding how scale-sensitive a driver is we can pinpoint which drivers need more data collection at finer scales and which do not, to improve our currently limited knowledge on drivers across multiple scales across Europe. Similarly, understanding the scale at which a species responds to a given driver can ensure that we collect species data at appropriate scales: e.g. coarse-scale temperature data will not be a good predictor for species which exploit localised microclimates (e.g., Settele and Kühn 2009). Furthermore, there is a challenge to understand how drivers, particularly social, economic and climatic drivers, will change in the future, and how this will shape our environment. More research is needed on projected land use and the potential effects on ecosystems.

For policies to be effective they must not only consider the status and trends of biodiversity at the correct spatial and temporal scales, but also consider how drivers operate across these scales. If the conservation and sustainable use of biodiversity is to be achieved then robust matching of scales in science, land management, and biodiversity policy is a fundamental requirement.

The governance of biodiversity

Previous research has established that the scales of governance mechanisms and policy instruments often do not match the ecological processes that they seek to address. Clearly, this is partly due to the complexity of both ecological and political systems. While it is possible to outline general rules for achieving certain conservation objectives (e.g. Table 1), their chance of success is strongly context specific. This is partly because the relevant spatial and temporal scales vary from case to case; for example, biodiversity drivers operate at various spatial and temporal scales and some of them are highly scale-sensitive so that mitigating policies fail if applied at the wrong scale.

These complexities alone would make designing a single universally applicable policy approach extremely challenging. In addition, governance itself is a complex issue. Biodiversity conservation is designed and implemented at legally defined administrative levels, which differ from country to country. Levels are divided into sectors in variable ways, and rights as well as responsibilities of actors vary from case to case. Furthermore, cross-scale and cross-level interactions are a genuine challenge for governance (Kettunen et al. 2014 this book), sometimes of a very political nature (Apostolopoulou and Paloniemi 2012). As a result, multiple actors make conservation decisions under multiple conditions (Primmer et al. 2014). These reflect the historical, institutional and political context in which policy design and implementation takes place (Cent et al. 2014 this book, Paloniemi et al. 2014 this book).

Governance of site selection

The Habitats Directive, for example, was celebrated at the time of its adoption in 1992 as a major policy turn in European nature conservation, and with good reason. However, many years later the actual site-selection at local levels caused long and serious conflicts where many land-owners and natural resource users (farmers, foresters, hunters, fishermen) challenged the legitimacy of nature conservation and of any new biodiversity conservation policy in most European countries (Mathevet et al 2014 this book). The implementation of site selection procedures of Natura 2000 was challenging in various historical, institutional and political contexts. As a consequence of this, governance mechanisms of site-selection beyond Natura 2000 have been based on other logic. Voluntary participation of owners and experimental governance mechanisms are stressed, meaning that the governance mechanisms are continuously monitored, evaluated and redirected. From a scale perspective, this has changed the temporal scale of governance, the role of actors at various levels, and the spatial scale of the outputs of the policy-making process. Protected areas arising from voluntary site-selection processes tend to be smaller than those from top-down driven processes. However, adoption of a voluntary approach might provide an opportunity to protect more sites than a top-down approach. Interestingly, new science-based tools for site selection, like ZONATION, provide opportunities to support this kind of decision-making (Arponen et al. 2014 this book).

The legitimacy criticisms arising from top-down site-selection mechanisms do not necessarily point in the same direction as the assessment of the effectiveness of policy instruments. Furthermore, the perceptions of legitimacy may change over time and people may become more positive towards Natura 2000 after the challenging site-selection phase has passed. Within the SCALES project, we found that, of current policy instruments, the instruments regarded as most effective in promoting ecological connectivity at larger scales are Natura 2000 and national parks (Paloniemi et al. 2014 this book).

Governance for connectivity and management

Policies to enhance regional connectivity are still in their infancy; their mechanisms tend to be sporadic and of insufficient magnitude in relation to the pressures, as our driver analysis indicates (Scott et al. 2014 this book, Touloumis and Pantis 2014b this book, Tzanopoulos et al. 2014 this book). However, one key policy idea, that may change this, is Green Infrastructure (Kettunnen et al. 2014 this book). Some countries or state unions, like the United Kingdom and France as well as the European Union as a whole, have already taken action in this direction. This idea combines the protection of biodiversity and that of ecosystem services and, because it concerns connectivity (see section "Connectivity and scale from an ecological perspective" above), may provide a systematic and comprehensive way of addressing conservation needs across multiple scales. It may lead to policy tools that are able to coordinate scientific information, planning, policy instruments and participatory processes in new ways (Mathevet et al. 2014 this book).

The importance of strengthening the experimental dimension of governance became evident also in our analysis of protected area management. New institutional arrangements for governing protected area management should be supported, evaluated and redirected. The establishment of official institutions consisting of a variety of actors from different governance levels with the responsibility for protected area management could be beneficial for dealing with scale challenges, if such governance arrangements are based on clearly defined goals that promote social-ecological resilience (Grodzinska-Jurczak et al. 2014 this book).

Equitable allocation across scales

Another scale problem in biodiversity governance arises because protected areas may be spatially clustered, whereas benefits are shared at national or even global scales. This may result in a situation where local economies bear a disproportionate share of the costs of biodiversity conservation. Intergovernmental ecological fiscal transfer is one policy response to this problem. It redistributes public revenue from national and regional state governments to local governments to share the costs of biodiversity conservation more equitably. Within the SCALES project we found promising avenues for future ecological fiscal transfer design and implementation in transfers based on qualitative indicators, alongside the quantitative protected-area-based indicators currently in use in Portugal and France (Santos et al. 2014 this book).

A general trend within governance in Europe is towards mechanisms that involve both public and private actors. While public actors have the dominant role and responsibility for biodiversity conservation, there are some signs of increasing involvement of private actors. A good example is citizen-based monitoring (McConville et al. 2014 this book), which is relevant also for managing scale related problems. Currently, the monitoring of biodiversity across the EU is a combination of state- and NGO-funded schemes and carried out by varying proportions of volunteers and professionals. The most pressing challenge is the sharing of knowledge for developing and running volunteer-based monitoring programs in different countries. With respect to emerging priorities from the EU 2020 Biodiversity Strategy, with six new dedicated targets, only a few Member States officially recognise the need to collect more data (e.g. the UK has begun to establish a baseline for monitoring ecosystem service provision). The typical remit of institutions currently involved in monitoring does not include responding to emerging priorities and so such institutions are reluctant to divert resources away from their core work. Thus, there is a need for greater guidance and sharing of knowledge and research into the development of new monitoring systems.

Policy outlook

The initiation of Natura 2000 as a broad-scale politico-ecological network was a big step forward for the conservation of European biodiversity, but there is great potential for improving its design and management. Much more emphasis should be given to the spatial arrangement of reserve sites and especially to the non-protected areas in between, which should be managed in a way that helps ensuring functional connectivity for organisms with all kinds of area requirements and dispersal capacities. Consideration of these different requirements is also a prerequisite for adapting networks of protected areas to future climate and land-use changes. Data and tools are increasingly becoming available (e.g. those compiled in the SCALE-TOOL: Henle et al. 2014 this book) to facilitate future improvements in regional connectivity and the coherence of protected areas networks.

Further potential steps to improve the effectiveness of the Natura 2000 network also could involve the harmonization of policies between Member States regarding biodiversity monitoring. Research and policy should account for the differences in scaling properties of different types of drivers of biodiversity change and explore governance structures that are better adapted to the scaling of conservation problems. New and improved policy instruments are available that may facilitate designing biodiversity conservation policies and management in more scale-sensitive ways. Nevertheless, there remain major challenges for the integration of these innovations and of scientific advances and tools into policies, decision-making and on-the-ground management. Reducing these obstacles could make a major contribution to securing the conservation of biodiversity across all kinds of spatial, temporal, ecological and administrative scales.

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This book is the first of its kind to describe the challenges that arise in studying and conserving biodiversity across different scales. Taking a scale-conscious view of the drivers of change, biodiversity patterns and processes themselves, and policy actions aimed at management and protection, it describes a wide range of practical methods and recommendations to improve conservation at continental and global scales.

Drivers of change are considered at different spatial scales, including the likely effects on biodiversity under land use and climate change. Ecological patterns and processes are examined and modelled at different levels of biological organization, from genetics, through individual dispersal and population viability, to community structure and selected ecosystem services. Trade-offs and tensions between different conservation goals are explored, and promising new methods for the study of scaling effects are digested from the scientific literature. Different governance and policy tools are evaluated and recommendations given. Finally, case studies from both Europe and Taiwan illustrate many of the scaling issues with a focus on networks of protected areas and their connectivity.

The book is addressed to a wide range of readers. Scientists will find readable summaries of analyses, methods and case studies. Conservationists and policy makers will find recommendations and ideas for management, biodiversity governance, and decision-making. Lecturers will find good examples to illustrate the challenges that arise from considering multiple scales in ecology and biodiversity conservation. Moreover, everyone concerned with conservation will find ideas in this book to help in the urgent task of protecting biological diversity through study, insight and action at all kinds of scales: spatial, temporal, administrative and ecological.

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