Matthias Schröter · Aletta Bonn · Stefan Klotz
Ralf Seppelt · Cornelia Baessler *Editors*

Atlas of Ecosystem Services

Drivers, Risks, and Societal Responses

Springer

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Foreword

Ecosystems constitute our natural capital. They provide us with the essential ecosystem services that underpin our economy, from fertile soils and multifunctional forests to productive land and seas, from good quality fresh water and clean air to pollination and climate regulation and protection against natural disasters.

One way of protecting our natural capital is the conservation of its biodiversity through establishing protected areas which safeguard vulnerable habitats and species, such as the European Union's Natura 2000 network. However, the services that ecosystems provide do not stop at the borders of protected areas. Many of these ecosystem services are produced outside nature reserves, thus in man-made landscapes; for instance in urban parks and gardens, farmlands, estuaries, or coastal zones.

This is why the European Union proposed a new biodiversity strategy (The EU Biodiversity Strategy to 2020) in 2011. This strategy includes an ecosystem service–based approach which complements nature conservation as a strategy to protect biodiversity. Drawing heavily on the outcomes of the Millennium Ecosystem Assessment and the TEEB study (The Economics of Ecosystems and Biodiversity), ecosystem service–based approaches aim to keep ecosystems healthy by demonstrating how they deliver value to the society and the economy.

The inclusion of ecosystem services in global and EU biodiversity policies has spurred a lot of research. In particular Action 5 of the EU Biodiversity Strategy has been an important incentive for scientists to develop tools and methods for analysing ecosystem services. Action 5 of the strategy urges the EU member states to map and assess the state of ecosystems and their services in their national territory, to assess the economic value of such services, and to promote the integration of these values into accounting and reporting systems. Furthermore, the EU's Horizon 2020 framework programme for research and innovation has funded large-scale actions. These projects have studied the complex relationships between biodiversity and ecosystem services, and they have tested the utility of the ecosystem service concept in real-life test cases and policies.

Controversy is never far away. In particular the monetary valuation of ecosystem services has sparked a sometimes harsh debate between conservationists who argue that nature should be protected for its own sake and proponents of instrumental values who say we must save nature to help ourselves.

However, it must be clear that ecosystem services scientists do more than just monetary valuation. Ecologists and geographers have developed maps, models, and assessment tools to quantify the physical flows of ecosystem services that ecosystems provide to society. Economists are building accounting systems to capture the different values of biodiversity and ecosystem services. Social scientists have developed new governance models which engage different stakeholders in biodiversity policy and to understand the plurality of values which people associate with nature.

This book is a perfect reflection of the integrated approach to understand how ecosystem services increase human well-being. It demonstrates that joint projects and integrated assessments have turned the research on ecosystem services into a truly interdisciplinary science. It is not merely a complement to conservation; it is a paradigm for sustainability.

Whereas the knowledge base on ecosystem services has substantially expanded during the last decade, policy uptake still needs to be improved. This is indeed one of the specific actions of a recently adopted EU Action Plan for nature, people, and the economy. This action plan has been developed to strengthen the implementation of nature legislation. A better integration of ecosystem services into decision making is also a key objective of IPBES, the Intergovernmental Platform on Biodiversity and Ecosystem Services, as it provides policy makers with recommendations based on thorough scientific assessments.

Policy applications of ecosystem services need more visibility and require a more consistent framework which delivers numbers when you need them. This is why the EU has started the development of natural capital accounts. In 2015, a Knowledge Innovation Project on an Integrated System for Natural Capital and Ecosystem Services Accounting (KIP INCA) was launched jointly by the European Commission and the European Environment Agency. These accounts can be used to monitor the extent and condition of ecosystems and to measure their contributions to the economy. At national scale such accounts can be linked to economic data. Remote sensing is expected to become a major contributor but this requires customising products that are obtained from satellite data. At local and regional scales, the EU invests in projects which operationalize ecosystem services through nature-based solutions or by developing green infrastructure, for instance in and around cities.

Within this book key driving forces and pressures that put sustained ecosystem service provision at risk are identified, presented, and discussed across different ecosystems and across scales.

This book shows why research and innovation remain crucially important to develop, test, and evaluate ecosystem services–based approaches and to successfully integrate them in planning and policy. It provides excellent examples of best practices for a better management of our natural capital.

Ispra, Italy Joachim Maes

Preface

The aim of this *Atlas of Ecosystem Services* is to collect current knowledge on drivers, trade-offs, and synergies of ecosystem services and biodiversity, as well as on societal responses. It presents case studies from various fields to demonstrate concepts of sustainable land management and governance. While research on ecosystem services has advanced in the past decades, there is yet little knowledge on how drivers and trade-offs, especially in their complexity, translate into risks of sustained ecosystem service provision. The *Atlas* aims to contribute to closing this knowledge gap.

Structured in five parts and in total 60 chapters, the *Atlas* starts with a collection of conceptual background chapters, in which also the framework of the *Atlas* is presented. Part II presents drivers and their risks for ecosystems, their functions, and services. Part III contains chapters on trade-offs and synergies among ecosystem services. Part IV focuses on societal responses to drivers and trade-offs among ecosystem services. Part V presents our conclusions. Each section starts with an introductory chapter to guide the reader through the *Atlas*.

The idea of this *Atlas* arose from the Integrated Project on Land Use Conflicts of the Helmholtz Centre for Environmental Research–UFZ. It quickly developed into an integrative synthesis project that many colleagues from outside the UFZ joined. We hope that this *Atlas* can contribute to an interdisciplinary understanding of risks faced by ecosystems and the services they provide and that it provides easily accessible insights from different disciplines to support environmental policy.

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Part I

Conceptual Background

1 The Risk to Ecosystems and Ecosystem Services: A Framework for the Atlas of Ecosystem Services

Matthias Schröter, Christian Kuhlicke, Johannes Förster, Cornelia Baessler, and Aletta Bonn

1.1 Description of the Framework

Our framework (Fig. 1.1) depicts different components of a dynamic social-ecological system in which ecosystem services are provided and managed [1]. The impact of drivers on ecosystems can lead to a loss of ecological processes and properties, which are the basis for the provision of ecosystem services to society. A loss in ecosystem service provision can lead to a reduction in human well-being, with respective costs to individuals and societies. The loss of a particular ecosystem service can also be the result of trade-offs with other ecosystem services [2], which might be different among societal groups [3]. For this reason, the effect of interactions among ecosystem services is recognized as a specific source of risk in our framework. Society expresses different forms of demand for ecosystem services [4], which can result in dependencies on particular sets of ecosystem services. Socio-ecological responses to potential losses in ecosystem services address mitigation (Response 1), adaptation (Response 2) and transformation options (Response 3). In this opening chapter of the Atlas we will introduce and explain these components in detail, with a conceptual focus on risks, and provide a glossary of terms (Table 1.1). The chapter complements other conceptual contributions in Part I of the Atlas.

Which ecosystem services are addressed? Presentation of the general framework of the Atlas, applicable to all ecosystem services.

What is the research question addressed? We conceptualise how drivers relate to risks, which affect ecosystems and ecosystem services, and how society responds to these risks.

Which method has been applied? Conceptual thinking.

What is the main result? We distinguish the risk to ecosystems (first order risk) and the risk to ecosystem services (second order risk), the latter widening the scope to social-ecological perspectives. We identify drivers and trade-offs between ecosystem services as sources of risks, and distinguish three societal responses: avoidance, adaptation, and transformation.

What is concluded, recommended? We provide an outlook to the structure of the Atlas, consisting of parts on drivers, on trade-offs, and on societal responses.

1.2 From Ecosystem Risk to Ecosystem Service Risk

Risk can be defined in multiple ways [5]. Risk is often conceptualized as an undesirable consequence of either natural or human-induced events. Examples of risk are floods or droughts that lead to loss of property or valuable resources. Risks are composed of the interaction of hazard and vulnerability. Hazards are phenomena that threaten, while vulnerability is a system property that defines the capacity to cope with these threats (e.g. the level of flood protection of a town or the availability of alternative sources of water supply). Risk is embedded in social contexts, which define what is valuable, how risk is perceived, who is affected, and what management options are available or acceptable.

We distinguish between the risk to ecosystems (first order risk) and the risk to ecosystem services (second order risk) (Fig. 1.1). An ecosystem risk is defined as the interaction of a hazard potentially causing harm to the state and condition of an ecosystem and the vulnerability of an ecosystem. A hazard is a situation or condition potentially leading to harm, such as the drainage of a wetland or deforestation. Vulnerability of an ecosystem is constituted by its exposure to harm, its susceptibility, and its resilience. Ecosystem state and condition have dimensions of quantity (areal extent or biophysical mass and volume), quality (specific characteristics that influence the

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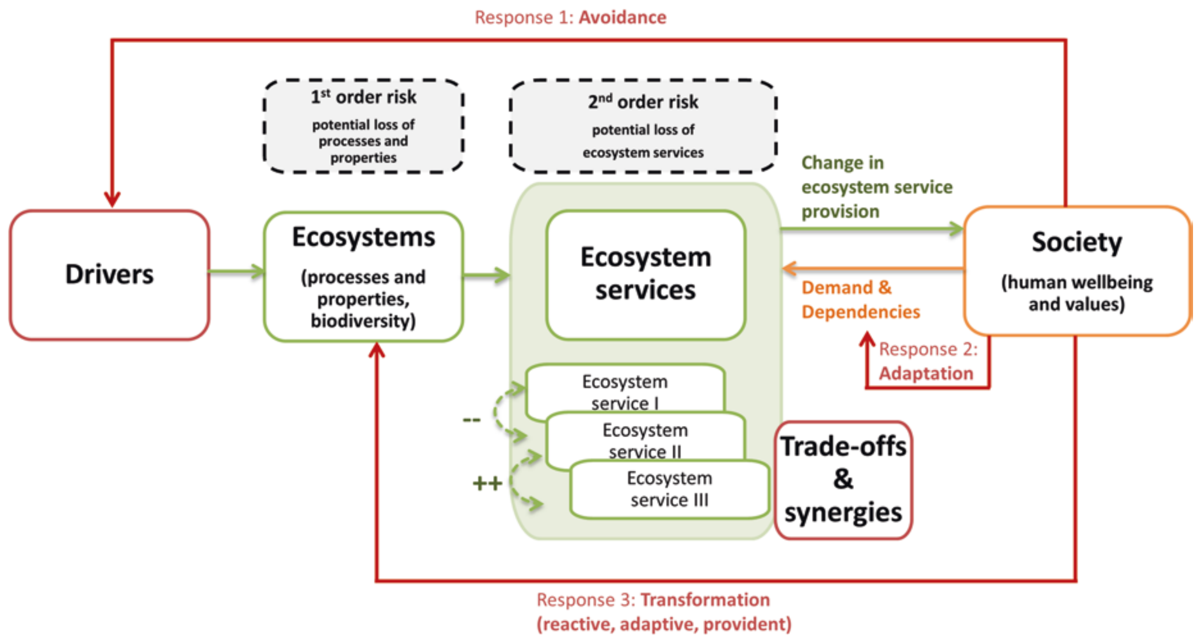

Fig. 1.1 Framework of the Atlas of Ecosystem Services depicting elements that are addressed in contributions of the Atlas. Drivers as well as trade-offs lead to risks to ecosystems and the services they provide. Different societal responses can address these potential losses of ecosystem service provision (responses 1–3)

Table 1.1 Glossary

Ecosystem services: Ecosystem services are "the contributions that ecosystems make to human well-being" [8].
Ecosystem risk (first order risk): An ecosystem risk is defined as the interaction of a hazard potentially causing harm to the state and condition of the ecosystem and the vulnerability of the ecosystem. Vulnerability is constituted by the exposure of an ecosystem to harm, its susceptibility, and its resilience. The state and condition of an ecosystem are characterized by its diversity, functions, and flow of energy, matter, and information.
Ecosystem service risk (second order risk): An ecosystem service risk is defined by a hazard potentially causing harm to the services an ecosystem provides to society (individual, groups, communities, sectors, regions etc.) as well as the vulnerability of society, which is constituted by its exposure, susceptibility, and resilience (in the sense of ability to cope with and adapt to the disruption/loss of service). Hence, the risk to an ecosystem service can be on both the ecological side of ecosystem services, which relates to the capacity of service provision, and the socio-economic side of ecosystem services, which relates to actual use of services and societal preferences.
Risk to ecosystem capacity to provide ecosystem services: The capacity of an ecosystem to provide ecosystem services can be at risk if ecosystems and their biophysical processes are changing, leading to a change in the quality and quantity of potential ecosystem service provision.
Risk to societal use of ecosystem services: Risk to actual ecosystem service use or appropriation results from the capacity of society to cope with hazards or scarcity. The capacity of society to use and appropriate ecosystem services is determined by available skills, knowledge, and management strategies. These can include management practices and policies that consider ecosystem services (e.g., spatial planning). The availability of skills, knowledge, management practices, and policies determines the ability of society to deal with scarcity of ecosystem services, the ability of management to ensure renewability of ecosystem services, and the dependence of society on a particular service, i.e., the degree of its substitutability.
Hazard: Any event or process that can cause harm.
Exposure: Physical precondition to be affected.
Susceptibility: Precondition to be harmed, defined by degree of fragility or degraded state and conditions of an ecosystem.
Resilience: "Resilience is the capacity of a system to experience shocks while retaining essentially the same function, structure, feedbacks, and therefore identity" [9]

functioning), and spatial configuration (distribution of ecosystems) [6]. Examples of ecosystem risks are the introduction of new species that can change food webs, environmental pollution that can change species abundance, or overuse of an ecosystem that changes species diversity and ecosystem processes. Such risks could lead to changes in ecosystem properties, that is, characteristics such as the composition of species and their functions, population sizes, or growth rates. The focus of ecosystem risk is, in principle, on biocentric values, i.e., on the continued existence of ecosystems and biodiversity, while ecosystem service risk, as we shall see, is anthropocentric with regards to its role for human well-being.

Ecosystem services widen the scope of ecosystem risk to a broader socio-ecological systems perspective [1]. Risk to ecosystems and risk to ecosystem services differ. While the risk to ecosystems relates to the consequences for a specific ecosystem, the risk to ecosystem services relates to the change of ecosystem service provision to society. An ecosystem service risk is defined by the interaction of a hazard potentially causing harm to the services an ecosystem provides to society (individuals, groups, communities, sectors, regions etc.) and the vulnerability of a society, which is constituted by its exposure, susceptibility, and resilience. Hence, the risk to an ecosystem service can occur both on the ecological side of ecosystem services, which relates to the capacity for service provision, and on the socio-economic side of ecosystem services, which relates to actual use or disruption of services and societal preferences.

For example, degradation of wetlands and deforestation changes water retention in soils and species composition. This can have a negative impact on the ability of these ecosystems to mitigate flood events or to provide continued water supply during droughts. The distinction between ecosystem risks and ecosystem service risks acknowledges that while all risk assessments are based on social values and norms, ecosystem service risk highlights associated risk to beneficial characteristics of ecosystems (e.g., flood mitigation or water provision). This perspective underlines, for instance, that not every ecosystem risk turns into an ecosystem service risk as beneficiaries for a service might be absent, society might change preferences for a respective service or find anthropogenic substitutes. For example, a village can reduce its susceptibility to floods by mitigation and adaptation strategies such as building a dyke, moving houses out of floodplains, and/or maintaining natural floodplains in order to reduce the impact of flood events. In addition to risk perception and acceptability, risk assessment and management are also influenced by social norms and values (e.g., how scarce a service is in relation to its ability to satisfy needs), and human dependence on a service. While conceptually the risks to ecosystems and to ecosystem services can be distinguished, it is challenging, in practice, to assess these separately, in particular for cultural ecosystem services, where social norms and values and biophysical attributes affecting a service are intricately interlinked.

1.3 Origins of Ecosystem Service Risk

We distinguish two types of origins of ecosystem service risk: drivers and trade-offs (see boxes "Drivers" and "Trade-offs and synergies" in Fig. 1.1). The first are direct natural or anthropogenic drivers that impact the ecosystem services provided by an ecosystem, such as land use or climatic changes. Drivers turn into a risk when they affect a system that is vulnerable to that particular driver. The impact is the magnitude of change in ecosystem processes and properties that lead to a change in service provision.

The second origin of ecosystem service risk is ecosystem management that leads to a trade-off between services, i.e., a situation where the provision of one service decreases while the provision of another one increases (Fig. 1.1) [7]. While such ecosystem management is also an anthropogenic driver, the focus here is on the complex interactions between multiple services. An example is land use change, which, as a driver, affects ecosystems and usually induces a trade-off between different potential services they provide. Within the context of the Atlas, however, we have distinguished contributions that specifically focus on the trade-offs and synergies between different ecosystem services (Part III).

1.4 Societal Responses to Ecosystem Service Risks

Target variables of ecosystem risk assessment and management are ecosystem processes and properties, while target variables of ecosystem service risk assessment and management are ecosystem properties as well as social norms and behavior. Risk perception and recognition for the need to govern and manage ecosystem service risk can lead to four types of societal responses. These three responses are illustrated in Fig. 1.1 by the red arrows (Responses 1–3 in Fig. 1.1). Response 1: *Avoidance* refers to governance and management instruments that target options for mitigating drivers, such as land use or climatic changes. Response 2: Societal responses can foster adaptation to the change of ecosystem services through *finding substitutes or adjusting preferences*. Response 3: Active anticipated transformation through *reactive, adaptive and provident strategies and measures* focused on ecosystem management.

1.5 Outlook to the Atlas of Ecosystem Services

Each contribution to the Atlas addresses particular aspects of the framework. Contributions in Part I explain concepts and ideas upon which this Atlas builds. Some contributions study a particular driver and its consequences for the provision of an ecosystem service (Part II). Others study the interactions between multiple ecosystem services (Part III). Contributions also address the socio-economic side of ecosystem service risk and focus on specific aspects of societal dependencies or societal responses (Part IV).

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2 The Ecosystem Service Concept: Linking Ecosystems and Human Wellbeing

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2.1 The Development of the Ecosystem Service Concept in Science and Policy

Ecosystem services are the contributions of ecosystems to human well-being. Well-being is understood as freedom of choice and actions, security, health, social relations, and basic materials for a good life [1]. Interest in the ecosystem service concept has been increasing in science and policy during the last two decades. However, the simple idea that ecosystems provide benefits to people and that conservation of ecosystems may prevent the risk of loss of ecosystem services is rather old. Plato and Aristotle, for instance, observed that degraded forests were linked to higher soil erosion in ancient Greece. In the 1980s, the concept was strongly promoted by conservation biologists who pointed out that biodiversity should be conserved not only for its own sake but also because of the services it provides to people [2]. Ideas similar to the notion of ecosystem services have been developed in spatial planning [3]. In the 1990s the concept became increasingly mainstreamed in the literature, with prominent contributions by de Groot, classifying functions of nature [4]; Daily, setting an agenda for studying multiple facets of ecosystem services [5]; and Costanza et al., assessing the global economic values of ecosystem services [6]. The initial focus of the concept on economic valuation has been strong, and has also been perceived as dominant, but the concept can be operationalised for landscape planning and nature conservation based on mainly biophysical information on services. The Millennium Ecosystem Assessment [1], was the main push for the concept to be taken up by policy initiatives and – following this – also in broader science agendas. At the policy level, The Economics of Ecosystems and Biodiversity (TEEB) [7] reports were initiated to draw attention to the global economic benefits of biodiversity and to crucially highlight the growing costs of biodiversity loss and ecosystem degradation. TEEB has synthesized existing scientific knowledge and practical experiences, drawing on ideas and concepts from many sources and scientific fields (including the concept of ecosystem services). TEEB emphasized the interdependency of the social and biophysical world and the need to manage natural resources with a balanced perspective of its social, economic, and ecological dimensions. Ecosystem services are now included in the global *Strategic Plan for Biodiversity of the Convention on Biological Diversity* [8] in a set of goals (Aichi targets). Goal D, for instance, calls to "enhance the benefits to all from biodiversity and ecosystem services." The use of the concept at the science-policy interface is furthermore enhanced through the establishment of the *Intergovernmental Platform on Biodiversity and Ecosystem Services* (IPBES), set up in 2012 by over 100 governments with the goal to critically assess available knowledge on the state and trends of biodiversity and ecosystem services.

Which ecosystem services are addressed? General introduction to the ecosystem service concept.

What is the research question addressed? Introduction to the development of the ecosystem concept, distinction of components of ecosystem services, and the role the concept can play in the analysis of human-nature interactions.

Which method has been applied? Conceptual thinking.

What is the main result? The ecosystem service concept has a long history. Different components play a role in the emergence of a service. We distinguish properties and functions, capacity, flow, benefits, and values. Ecosystem services can be an anthropocentric argument for nature conservation, an analytical tool or a catalyst for inter- and transdisciplinary research.

What is concluded, recommended? We introduce the aim of the Atlas, in which we explore synergies and trade-offs existing between ecosystem services, options to inform threat assessment and risk management, and the development of management and policy tools to sustain the capacity of ecosystems to provide services into the future.

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At the European level, the *EU Biodiversity Strategy* has formulated specific targets linked to ecosystem services. Among other targets and actions, it requires member states to "map and assess the state of ecosystems and their services in their national territory" and to integrate "these values into accounting and reporting systems at EU and national level by 2020" [9]. Several national ecosystem assessments have been launched to gather knowledge on the state and trends of ecosystems and the services they provide [10].

2.2 Components of Ecosystem Services: Linking Ecosystems and Society

Ecosystems can provide multiple services. These services are beneficial flows of energy, matter, and information from ecosystems to society. They include provisioning, regulating, and cultural services. Provisioning services comprise provision of food and natural resources extracted from ecosystems, such as timber or biomass for energy. Regulating services modify environmental conditions and include the climate regulation through carbon sequestration or storage, the remediation of pollutants, the reduction of the risk of damages from, e.g., landslides, storms, or floods, as well as the maintenance of physical, chemical, and biological conditions of the environment, such as water quality regulation. Cultural services represent non-material benefits of ecosystems, such as the opportunity for physical experiences, like hiking, cycling, or other outdoor activities in a pleasant setting. Cultural services also provide the biophysical basis for aesthetic, scientific, and educational values people hold for the continued existence of species and ecosystems.

Ecosystem services are typically conceptualised as a series of components that play a role in the relationship between nature and human well-being (Fig. 2.1). Ecosystem properties and functions are a web of ecological interactions that influence the structure of ecosystems and flows of energy, matter, and information. Ecosystem service capacity is the potential of an ecosystem to provide ecosystem services. It is derived from a set of ecological properties and functions. Ecosystem service flow is the actual use, i.e., appreciation or appropriation, of a service. Benefits – positive changes in human well-being – are derived from the appropriation of ecosystem services. The value of these benefits can vary among different groups of society, depending on societal norms and traditions as well as individual needs, principles, and preferences. Values are measures of importance. If a service has a value, interests are articulated and resources for its appropriation and management are mobilised. This can trigger private and public decisions that can have an impact on land use directly or indirectly through policy instruments such as regulations (e.g., conservation designation), economic instruments (e.g., payments or taxes) or informational instruments. These instruments have an impact on different types or intensities of land use and land use change. Land management can entail (a) different forms and intensities of

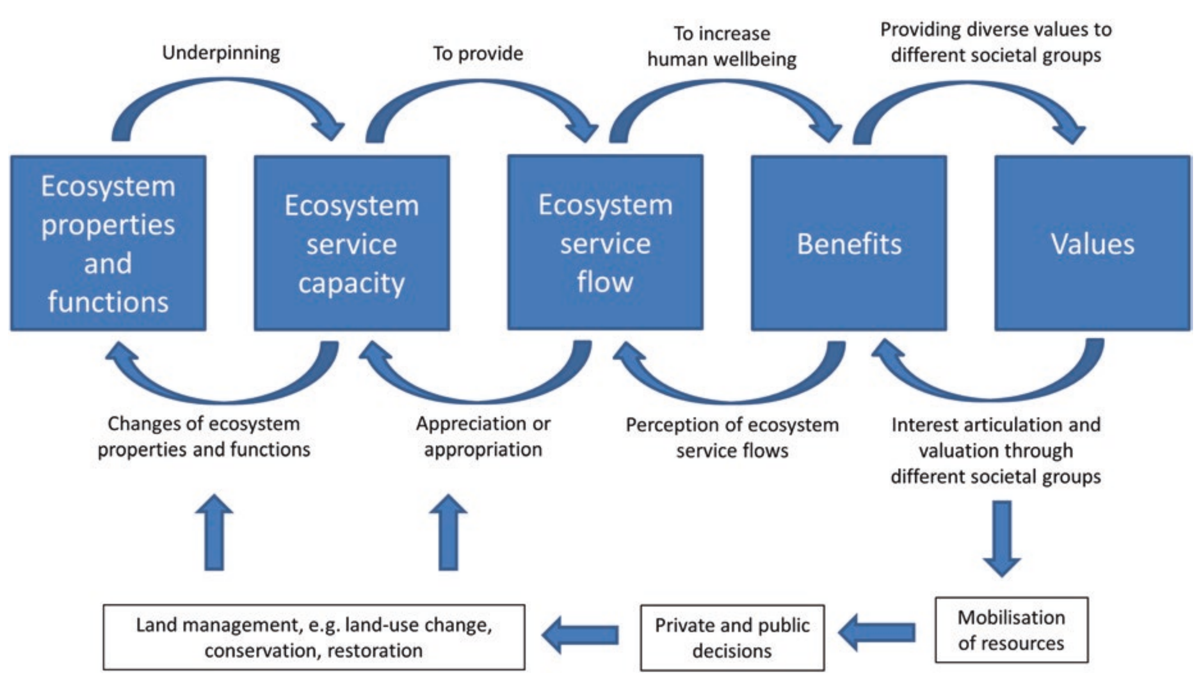

Fig. 2.1 Components of the ecosystem service framework [21]

management for, e.g., provisioning services; (b) transformation of land use, e.g., from forests to agriculture; (c) protection of land, e.g., through conservation designation to facilitate the use of regulating and cultural ecosystem services; or (d) habitat restoration in order to re-establish ecosystem properties and functions that foster the provision of a set of ecosystem services.

It is important, however, to acknowledge that provision of ecosystem services is often determined by different forms of capital, not only ecosystem functions or natural capital [11]. An ecosystem service flow is the result of a multitude of biophysical, social, and institutional factors influencing the mobilisation of ecosystem service capacities, e.g., via land management measures. For the use of many services, additional anthropogenic contributions are needed, which can include the use of technology or knowledge, e.g., the use of machines and fertilizer to manage timber or agricultural production, or access routes to accommodate recreational visits.

In addition, benefits, and thus benefit-generating ecosystem services, are perceived differently depending on the type of value-articulating institution (e.g., markets, social norms, traditional habits). For example, if the importance of regulating services (benefits), such as climate regulation or maintaining soil fertility, does not resonate in market-based considerations of agricultural production (values), these services may be disregarded in the mobilisation of ecosystem service capacities. Hence to sustainably manage ecosystem service provision, it is of central importance to understand societies' preferences for the importance of ecosystem services and to possibly enhance awareness, where services are currently taken for granted and synergies and trade-offs with other service provision is not visible.

2.3 What Roles Can the Ecosystem Service Concept Play?

The ecosystem service concept can have different roles. On a political level, it has been proposed as an additional anthropocentric argument for conservation: biodiversity and ecosystems should be sustainably managed, restored, or protected because of the benefits they provide to humans.

At a conceptual level, the ecosystem service concept operates as an analytical tool to understand particular aspects of the human-nature-relationship. Combining the political and conceptual roles, it can be used to link different disciplines and sectors – ecology, geography, economics, and other social sciences in particular – to analyse the role of ecological complexity in providing benefits to humans and the linkage to socio-economic values. The concept emphasizes the importance of the socio-cultural setting for ecosystem service identification, and thus conceptualises the human relation and view on ecosystems and their characteristics.

The concept can serve as a catalyst for inter- and transdisciplinary research, i.e., a joint work of different disciplines, or a joint work between science and society on a research problem. The concept can also facilitate joint understanding and development of ecosystem management by resource managers of different sectors and scientists by bringing added value through facilitating and broadening partnerships, opening up new sources of funding [12], developing understanding and focusing research, and informing policy [7]. Ecosystem services can therefore be understood as a boundary object [13], i.e., an idea that binds different views of different actors from different disciplines and decision makers, who work together at the interface of ecological and socio-economic systems.

The original conservation focus of the ecosystem service concept has increasingly shifted from mere protection to management of ecosystems that can provide different sets of services. In this way, the ecosystem service concept can serve as a platform for different groups of stakeholders to articulate their interest in how ecosystems should be managed to provide specific services. In this role, the concept serves as a communication tool in policy and management arenas (Fig. 2.2).

As the ecosystem service concept has a facilitating role to span disciplines and sectors, it has also been criticised as a "complexity blinder" [14], as it supposedly cannot cover the ecological complexity while focusing on outputs of ecosystems. Focusing on the economic angle, there is considerable critique of the perceived focus on monetary valuation, including the characterisation as an "economic production metaphor" [15] that might be at odds with other conceptualisations of the human-nature relationship. The ecosystem service concept therefore needs debate, and gains from continuous development.

2.4 What's Next? Research on Ecosystem Services at the Interface Between Science, Policy and Practice

Ecosystem services science relates to important societal questions and hence decision-making processes on how to manage, conserve, restore, and use ecosystems to sustain the provision of ecosystem services. This provides much-needed evidence to feed into science-policy interfaces, illustrated by the prominent role of the ecosystem service concept in the international IPBES, TEEB, or the EU MAES (Mapping and Assessment of Ecosystems and their Services) initiatives [7, 16, 17].

Fig. 2.2 Ecosystem services as a platform to articulate different values people hold for ecosystems, and how these should be managed and used. (*Courtesy of* Javier Sáez for the POLICYMIX project [22])

Important research areas within the field include the following subjects. A large proportion of research on ecosystem services has focused on setting up methodologies and standards for mapping and assessing services, which also forms a considerable part of this Atlas. Identifying patterns of spatial heterogeneity of potential service provision and actual use of services are among the main research interests. Considerable progress has been achieved in further developing methodologies and recommendations regarding valuation of ecosystem services, and recent developments focus on integrative valuation [18]. Of crucial interest is the relationship between biodiversity and ecosystem services [19]. This includes studying the dynamics of trade-offs and synergies of ecosystem services.

The information on ecosystem services informs different fields of decision-making, including landscape and urban planning, systematic conservation planning, environmental impact assessments, and strategic environmental assessments. This can be aided by practical applications, including inclusive cost-benefit or multi-criteria analyses, as well as scenario analyses, that assess the multiple effects of land use decisions on ecosystem service values. Ultimately, this provides crucial evidence to inform the design of policy instruments building on the diverse values ecosystem services provide to people. For this Atlas, we want to explore where synergies and trade-offs exist between ecosystem services and how this can inform, on the one hand, threat assessment [20] and risk management and, on the other hand, the development of management and policy tools to sustain the capacity of ecosystems to provide services into the future.

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3 The Link Between Diversity, Ecosystem Functions, and Ecosystem Services

Sonja Knapp

3.1 Ecosystem Functions and Their Relationship to Biodiversity

Which ecosystem services are addressed? Provisioning services: Biomass production, fruit production

Regulating services: Climate regulation, pest control, phosphorous retention, pollination, soil formation, water purification

What is the research question addressed? Will the loss of biodiversity result in a loss of ecosystem functioning and ultimately in the ecosystem's breakdown or its transformation to another ecosystem?

Which method has been applied? Literature review

What is the main result? The loss of biodiversity reduces the efficiency by which ecosystems provide ecosystem functions and related ecosystem services. The amount of biodiversity loss that will cause a significant loss of ecosystem functions and services, however, is largely unknown.

What is concluded, recommended? Our knowledge that a loss of biodiversity will ultimately cause a loss of functions and related services illustrates the need to protect biodiversity as completely as possible – as a precautionary principle.

Will the loss of biodiversity result in a loss of ecosystem functioning, and ultimately in the ecosystem's breakdown or its transformation to another ecosystem? This is a central question in biodiversity-ecosystem function (BEF) research. Although it remains unclear how much biodiversity is needed to sustain different ecosystem functions, biodiversity-ecosystem function research clearly shows that the extreme loss of biodiversity can reduce both ecosystem stability and a range of ecosystem functions. The more species are lost from an ecological community, the higher the risk that relevant functions provided by these species will be lost, i.e., this community will be less able, for example, to efficiently produce biomass or decompose and recycle organic matter [1].

Several interrelated concepts have been discussed in order to explain biodiversity-ecosystem function relationships: stability, complementarity, and redundancy (Fig. 3.1). The diversity-stability hypothesis assumes that because species differ in their traits (i.e., their functional characteristics, such as the way they are being pollinated or their rooting depths), "diverse ecosystems are more likely to contain some species that can thrive during a given environmental perturbation and thus compensate for competitors that are reduced by that disturbance" [2]. This assumption is closely related with complementarity, which "occurs when species exhibit various forms of niche partitioning that allow them to capture resources in ways that are complementary in space or time, or when interspecific interactions enhance the capture of resources by species when they are together" [3]. Here again, the differences in traits among species improve ecosystem functioning and its temporal stability. On the contrary, the species-redundancy hypothesis assumes that many species are so similar in their traits "that ecosystem functioning is independent of diversity if major functional groups are present" [2]. Thus, if species are functionally redundant to each other, a loss of species will not necessarily cause a loss of ecosystem functions; rather, the relationship between the number of species and the ecosystem functions being promoted will follow a saturating curve (Fig. 3.2) [4].

In summary, two decades of biodiversity-ecosystem function research have led to the consensus that "biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients" [1]. Consequently, "biodiversity increases the stability of ecosystem functions through time" [1], with biodiversity often impacting ecosystem functions in non-linear and saturating ways [1]. This means that the loss of functions with a loss of species will at first

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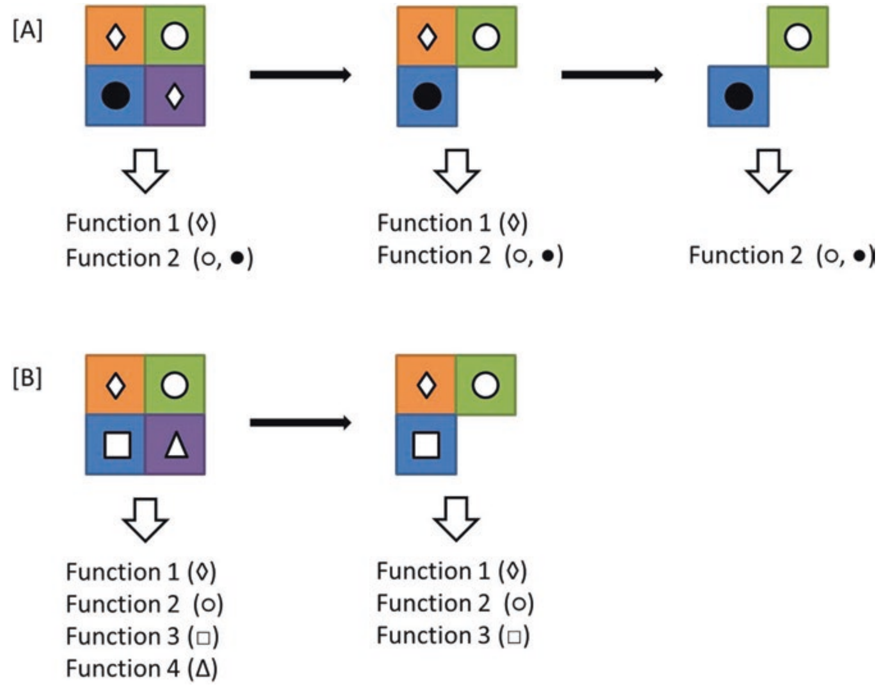

Fig. 3.1 If different species (colours) are either functionally redundant (same symbol) or complementary in their traits (white and black circle symbol), the loss of a species will only reduce ecosystem functioning, if the whole functional group is lost (**A**). If there is no redundancy and no complementarity, each species loss will reduce ecosystem functioning (**B**)

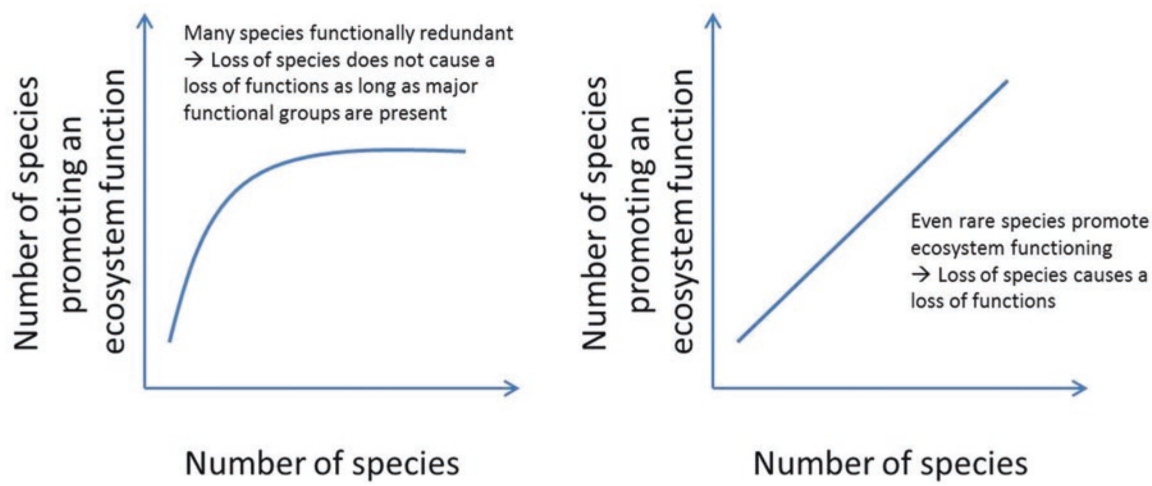

Fig. 3.2 Hypothesized relationships among biodiversity and ecosystem functioning [4]

be moderate, but will accelerate the more species are lost. However, linear biodiversity-ecosystem function relationships have also been observed, with every lost species causing a reduction in ecosystem functioning [4].

3.2 Ecosystem Functions, Biodiversity and Ecosystem Services in a Changing World

As a consequence of the relationships outlined above, environmental changes that affect biodiversity will affect not only ecosystem functioning, but also the provision of related ecosystem services. Agricultural intensification, for example, can trigger losses in belowground biodiversity. This in turn can reduce the complexity of food webs, soil ecosystem functions, and related ecosystem services such as water purification [5]. Similarly, agricultural intensification threatens large pollinators more than small pollinators, but the former more efficiently pollinate crops [6]. What follows is a loss of pollination potential that can reduce the yield of pollinator-dependent crops such as apples and almonds. Evidence suggests that many more ecosystem functions and services depend on biodiversity (Table 3.1). However, research on the relationships of biodiversity and ecosystem services (BES research) by now yielded less clear results

Table 3.1 Examples showing which groups of species support or counteract an ecosystem service and the underlying ecosystem function[a]

Species group	Support/counteract	Ecosystem services	Ecosystem functions	References
Aboveground predators Belowground herbivores Primary producers	Support	Biomass production	Primary productivity	[1, 3, 4, 9]
Primary producers	Support	Climate regulation	Evapotranspiration	[7, 10]
Aboveground predators Belowground predators Belowground herbivores Detritivores	Support	Pest control	Predation Consumption	[7, 9, 10]
Primary producers	Support	Phosphorous retention	Nutrient cycling	[9]
Belowground invertebrates Primary producers	Support	Soil fertility	Nutrient cycling	[10]
Aboveground herbivores Aboveground predators	Counteract	Biomass production	Primary productivity	[9]
Aboveground predators	Counteract	Pest control	Predation	[1, 9]
Aboveground predators Bacterivores	Counteract	Phosphorous retention	Nutrient cycling	[9]

[a]All examples stem from meta-analyses that drew their results from large numbers of studies (e.g. 44 experiments [3] or 150 natural grassland sites [9]). Note that this is a selection, not a complete overview, meant to illustrate that evidence for BES-relationships is mixed and many different species groups can affect different services

than biodiversity-ecosystem function research. Often, services cannot be measured directly, and data is insufficient [1]. Biodiversity was shown to support a number of ecosystem services [7], but for many services, evidence is still mixed (Table 3.1) [1, 8].

3.3 How Much Biodiversity Is Needed to Guarantee Ecosystem Functioning and a Sufficient Provision of Ecosystem Services?

The amount of biodiversity loss that will cause a significant loss of ecosystem functions and services is largely unknown. This amount cannot be quantified by studying single trophic groups, functions, and services. Rather, diversity across different trophic levels is required for multifunctionality and multiple ecosystem services [9]. Moreover, the amount of biodiversity that is necessary to maintain ecosystem functions and services cannot solely be quantified from diversity, but both the identity and diversity of organisms need to be known to do so [1]. Different species and their traits promote different functions and different services; how they do so might differ depending on environmental context [4]. Notwithstanding these difficulties, our knowledge that a loss of biodiversity will ultimately cause a loss of functions and related services illustrates the need to protect biodiversity as completely as possible – as a precautionary principle.

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4 Embracing Community Resilience in Ecosystem Management and Research

Christian Kuhlicke

4.1 Introduction

This chapter sketches out a framework that provides a heuristic perspective for better understanding and analysing community resilience in biodiversity management and research. The framework itself was developed in an iterative process within the European research project known as emBRACE (Building Resilience Amongst Communities in Europe), funded by the European Commission (contract number: 283201). (For more information, visit embrace-eu.org.) It builds upon existing scholarly debates, on case study work in five European countries, and on participatory consultation procsses with community stakeholders. In addition, the framework was applied and ground-tested in different contexts and for different environmental risks. In this chapter, the argument remains on a more general level, while more specific details on the theoretical underpinning of the framework, the emprical case studies, and operational indicators can be found in papers by Jülich et al. and Kruse et al. [1–3].

Which ecosystem services are addressed? Ecosystem services in general, without further specifications.

What is the research question addressed? How do we conceptualise and analyse community resilience in the context of ecosystem and biodiversity research and policy?

Which method has been applied? Conceptual overview and outline of an assessment framework

What is the major result? Relevant for community resilience actions in the field of ecosystem management as well as the wider area of social protection, capacity, resources, and social learning processes; all processes are embedded in interaction with wider governance settings.

What is concluded, recommended? In order to analyse and enhance community resilience, all aspects of the framework presented need to be considered.

4.2 Resilience: A Dazzling Term

Although the theoretical and empirical engagements with the concept of resilience are multi-disciplinary and stem from such different fields as ecology, psychological, and organisational studies, to name just the most common ones, they rarely engage with each other [4]. This is quite surprising, as there are common themes in most arguments: Resilience is usually utilized to refer to some kind of crisis or disturbance that needs to be acted upon and is usually understood as a system's or as an actor's capacity to adapt to and/or respond to disturbing events while simultaneously maintaining some degree of functioning [5]. The overall orientation on resilience concentrates, therefore, on how the concept might be useful for managing organisations or individuals to enhance, build, or develop their capacities to come to terms with new and unexpected events.

Holling's work on resilience, originating from the field of ecology, is probably the most prominent. Holling wrote that resilience is the "persistence of relationships within a system and is a measure of the ability of these systems to absorb changes of state variables, driving variables, and parameters, and still persist" [6]. This understanding of resilience implies that ecosystems are defined by multiple states of stability and that resilience describes the ability of ecosystems to absorb disturbance while maintaining essential structures and functions.

While the concept, as used in ecology, was initially meant as a descriptive aid to better understanding of the nonlinear dynamics of ecosystems, during the 1990s scholars started to also include human systems by expanding their analysis to so-called socio-ecological systems [7]. According to Folke, the reason for expanding the analysis was to acknowledge that "natural and social system behave in non-linear ways, exhibit marked thresholds in their dynamics, and that social-ecological systems act as strongly coupled, complex and

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evolving integrated systems" [8]. As an implication, resilience-based management concepts such as adaptive management and adaptive co-management of ecosystem [9] emphasize that knowledge about the future is always partial and defined by inherent uncertainties [10]. Management concepts should therefore provide ways of trying to learn and adapt while also managing and maintaining the basic functions and structures of a system. It is hence the ability to manage change and surprise while simultaneously maintaining the ability of a system to meet societal expectations and demands without eroding future needs.

Various disciplines beyond the immediate field of ecology have engaged more with the idea of resilience. Studies in socio-psychology have sought to identify individual competences and traits that make a person able to cope with traumatic experiences or to develop well in the face of adverse living conditions [11]. In the field of organisational studies, the concept was introduced as an alternative way of managing risks [12]. Particularly with regard to research on high reliability organisations (HRO), resilience became a relevant concept in order to better understand how organisations that are operating in a highly complex and tightly coupled working environment can maintain their operational functionality under all circumstances, since their failure would have catastrophic consequences [13]. Resilient factors identified include, among others, structure flexibility, redundancy, sense-making, culture of reliability, and mistake-orientation and improvisation [14, 15].

4.3 The emBRACE Community Framework

The emBRACE community resilience framework presented here takes these varied, briefly skechted out, strands of discussions on resilience into account, and focuses on the community level. It also incorporates the wider societal governance context within which decisions in ecosystem and biodiversity management are made.

Following the approach of Mulligan et al. [16], it is proposed to apply a dynamic and multi-layered understanding of community by understanding community as a place-based concept (e.g., inhabitants of a specific neighbourhood); as a virtual and communicative community within a spatially extended network (e.g., members of eocsystem management in a region); and/or as an imagined community of individuals who may never have contact with each other, but who share an identity.

Furthermore, the emBRACE framework conceptualizes community resilience as a set of intertwined components in a three-layer framework (see Fig. 4.1).

At the core of the framework are three interrelated domains that shape resilience within a community: (1) resources and capacities; (2), actions, and (3) social learning (see Table 4.1).

1. The domain of "capacities and resources" of the community and its members is informed through the Sustainable Livelihoods Approach (SLA) and its iterations [17, 18] as well as the through the concept of social capacity building [19]. Table 4.1 provides an overview of further specifiations.
2. Actions: Within the emBRACE framework, community resilience comprises two types of actions: (a) actions taking place within the more narrow setting of ecosystem management; and (b) actions in the wider context of ensuring social protection. Action within an ecosystem management framework aim predominantly at reducing the risk to a specific ecosystem and can include more mechanistic measures that aim at increasing the diversity, redundancy, or heterogeneity of an ecosystem or structural features that improve the recovery and resistance of an ecosystem. In addition, social protection includes actions that aim to reduce the risk to the provision of ecosystem services to society. Generally they include diverse types of actions intended to provide community members with the resources necessary to improve their living standards and their well-being.
3. Social learning is the third domain for community resilience in ecosysem and biodiversity management and research. Generally, social learning can take place in a formalised, curriculum-based setting, as well as in informal contexts that are shaped by conversations and mutual interest. It can result in different social outcomes, acquired skills, and stocks of knowledge [20]. More specifically, we understand social learning as consisting of different elements such as: (a) the perception of risks or losses; (b) the problematisation of such losses; and (c) critical reflections and testing/experimentation in order to evolve new knowledge that can be disseminated throughout and beyond the community, enabling resilience to be embeded at a range of societal levels. Table 4.2 provides an overview of and further specifications. About social learning.

These three domains are embedded in two layers of extra-community processes and structures. The first layer is the wider governance context, including regulatorory laws, policies, and responsibilities of different actors on multiple governance levels. These governance regimes usually aim at enabling and supporting regional, national, and international practices in ecosystem and biodiversity management. The second layer of extra-community processes and structures is the broader social, economic, political, and environmental context. This layer is influenced by all factors associated with the context, including rapid or incremental socio-economic changes in these factors over time and caused by disturbance.

Together, the three domains constitute the heuristic framework of community resilience, which through application can assist in defining the key drivers and barriers of resilience that affect any particular community within a context where ecosystems and the services they provide are put under pressure.

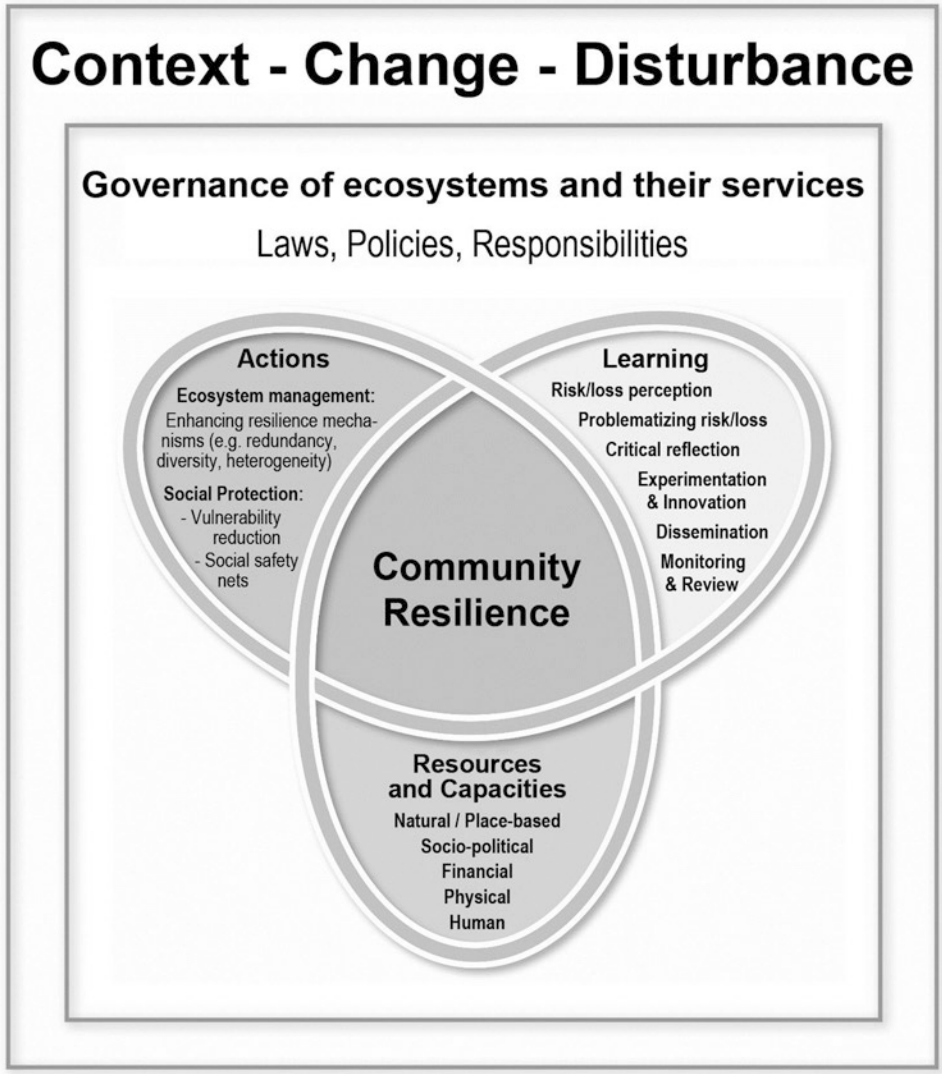

Fig. 4.1 Framework of Community Resilience identifying core elements. (*Adapted from* Kruse et al. [3]; Creative Commons Attribution 3.0 License)

Table 4.1 Overview on different capacities and resources

Natural and place-based capacities and resources	• Protecting and developing ecosystem services • Land, water, forests, and fisheries • Cultural/heritage resources; local public services, amenities, and access to jobs and markets • In-situ (legacy) housing, roads, water and sanitation systems, transport, communications, and other infrastructure
Socio-political capacities and resources	• Family, friends, and informal networks • More formal group memberships • Trust relationships that assist in collective action and knowledge-sharing • Power and capacity to influence political decision-making through formal and informal participation in and/or access to political processes
Financial capacities and resources	• Earned income, pensions, savings, credit facilities, benefits, access to insurance
Physical capacities and resources	• Adequate housing, roads, water, and sanitation systems • Effective transport, communications, and other infrastructure systems • Availability of and access to premises and equipment for employment
Human capacities and resources	• Health (physical and mental), work, knowledge, skills, education, self-esteem, and well-being • Fundamental resources for anybody, i.e., those resources without which it is difficult to make use of the other resource sets

(*Adapted from* Kruse et al. [3]; Creative Commons Attribution 3.0 License.)

Table 4.2 Overview on different dimensions of social learning

Risk and loss perception	The ability of any actor, organisation, or institution (the market) of interest to have awareness of current or future ecosystem or ecosystem services risks. Awareness can be derived from scientific analysis or other stocks of knowledge
Problematizing risk and loss	Arises once a threshold of risk tolerance is passed. Perception that potential or actual risk, or the costs of risk management actions, are inappropriate
Critical reflection	The act of questioning the appropriateness of measures and the underlying values and governance frames that are attributed to the management of ecosystem and ecosystem services risks
Innovation	The processes that transfrom new ideas in original management actions, technologies, and products. This can include the incorporation of knowledge from other places or policy areas, as well as advances based on new information and knowledge generation
Experimentation	The testing of multiple approaches to solving a risk management problem by understanding the outcome of such processes as unknown. This shifts risk management to a new efficiency mode where experimentation is part of the short-term cost of resilience and long-term risk reduction
Dissemination	The spreading of social and policy communities of ideas, practices, tools, techniques, and values that have proven to meet risk management objectives
Monitoring and review	The existence of processes and capacity that can monitor the appropriateness of existing risk management regimes in anticipation of changing social and technological, environmental, policy, hazard, and risk perception contexts

(*Adapted from* Kruse et al. [3];)

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5 Risk and Uncertainty as Sources of Economic Value of Biodiversity and Ecosystem Services

Bartosz Bartkowski and Bernd Hansjürgens

5.1 Risk and Uncertainty in Ecosystem Service Contexts

The future is uncertain. Given our seriously limited clairvoyant abilities, this is a trivial fact, which applies to all domains of human life – including our interactions with ecosystems. In the context of ecosystems and the goods and services they provide to us, this uncertainty about the future has multiple facets. In fact, it is this uncertainty that makes particular properties of ecosystems valuable. But before we come to questions of value and valuation, we need clarity about the different facets of uncertainty (in the sense of this Atlas' conceptual framework, the focus here is on "ecosystem service risk," i.e., uncertainty regarding the capacity of ecosystems to provide ecosystem services).

In its most general sense, uncertainty can be defined as prevalent when "[a] person…lacks confidence about his knowledge relating to a specific question" [1]. In economics, it is common to differentiate, building upon Knight [2], between *risk* and *uncertainty* (in what follows, standalone *uncertainty* is used as a general concept, while *Knightian uncertainty* refers to the specific definition provided by Knight); later contributions added the notion of *ignorance*. Risk prevails in the "world of classical statistics," where there is a known set of possible future events and every event can be attributed a probability (distribution) on the basis of observations about relative frequencies with which these events took place in the past. Uncertainty is more severe – it prevails where possible future events are known but not all can be assigned objective probability (distributions). This is, in a sense, the "world of Bayesian statistics," as in the latter probability distributions need not be based on previously observed frequencies of events, but can be derived, at least in theory, from subjective "gut feeling" (*a priori* probabilities). Ignorance is the most problematic issue: not only are probability distributions unknown, but so are possible future events. This is, so to speak, the world of "anything can happen."

Most people are risk-averse [3], which means that they prefer a safe pay-out over a lottery whose expected value

Which ecosystem services are addressed? Biodiversity (and implicitly all ecosystem services).

What is the research question addressed? What is the link between biodiversity's economic value and the risk and uncertainty surrounding ecosystem services provision?

Which method has been applied? Conceptual reasoning, literature study.

What is the main result? Biodiversity ensures future provision of ecosystem services (insurance value) and can be considered a pool of "potential ecosystem services" that may be demanded in the future (option value).

What is concluded, recommended? By conserving and enhancing biodiversity, uncertainty about future supply of and demand for ecosystem services can be hedged. Biodiversity's insurance and option value offer information about society's willingness to tolerate risk and uncertainty. It can help to make the right decisions about how to manage ecosystems in the face of risk and uncertainty.

equals the safe pay-out (called *safety equivalent*). Consequently, they welcome and are willing to pay for activities which help to lower risk.

Risk-aversion is crucial here, because it means that objective risks are relevant to the decisions and evaluations of states of affairs by people. In the context of ecosystems and the services they provide, we can differentiate between two main types of uncertainty about the future [4], from which ecosystem service risk of 2nd order results (Chap. 1):

1. Ecological uncertainty leading to **supply uncertainty**: Ecosystems are highly complex, dynamic systems embedded in an even more complex network of interactions with other systems. It is therefore impossible to predict their

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future behaviour with precision. The future supply of ecosystem services (be it provisioning, regulating, or cultural services) is uncertain. Depending on the ecosystem, how well studied it is, we have to do here with either risk or, mostly, uncertainty (both in Knightian sense).

2. Socio-economic uncertainty leading to **demand uncertainty**: Human preferences and needs are constantly changing, depending on both subjective and objective factors. Therefore, ecosystem functions that are not considered services today may be demanded in the future, by ourselves or our descendants. The problem is that we can hardly anticipate future preferences [5]; we can speak of ignorance or radical uncertainty in this context.

Of course, supply and demand uncertainty interact, and it is not always possible to delineate them clearly. Yet it is useful for our present discussion to keep this differentiation in mind.

5.2 From Risk and Uncertainty to Value: Insurance and Options

The two types of uncertainty about the future, supply uncertainty and demand uncertainty, correspond neatly with two economic value concepts: insurance value and option value. Both can be attributed to the same property of ecosystems, namely their biodiversity.

The intuition that biodiversity serves as *ecological insurance* is one of the central results of the research on biodiversity-ecosystem functioning relationships [6, 7]. Biodiversity positively influences the stability and resilience of ecosystems, although its influence on *provisioning* ecosystem services is not necessarily positive [8]. An important mechanism in this context is functional redundancy, i.e., the existence of species within ecosystems which serve as a "backup" for others in terms of fulfilling particular ecosystem functions [9].

If we couple this notion with the idea of supply uncertainty and human risk-aversion, we arrive at the concept of economic **insurance value** of biodiversity [10–12]. By stabilising the capacity of an ecosystem to reliably provide goods and services, biodiversity serves as an economic insurance. It fulfils a function similar to that of a financial portfolio: In a biodiverse ecosystem, risk is spread among many different items (species, genes, functional traits, etc.), so that its capacity to sustain exogenous shocks is increased [11].

Two specific interpretations of insurance value are possible: First, biodiversity can be argued to promote *acute stability* in the sense of resistance of the biodiverse ecosystem to exogenous shocks, e.g., storms or climatic events such as droughts. Second, when biodiversity positively influences the temporal stability of an ecosystem, this can be valuable if coupled with intergenerational equity concerns. People may appreciate the fact that a relatively biodiverse ecosystem is more likely to be available to future generations, on top of its availability to themselves. However, this temporal stability is not to be understood in the sense of a non-changing state, but rather as the more general capacity to provide goods and services.

Biodiversity is economically valuable because it serves as insurance against the uncertainty surrounding the capacity of ecosystems to reliably provide goods and services in the future. Thus, it responds to supply uncertainty.

However, it also hedges against demand uncertainty and is thus carrier of **option value**. Since we do not know the future needs and preferences of ourselves and our descendants, it may be wise to "invest" in the preservation of options, i.e., things (potential ecosystem goods and services) we might want to use in the future [13, 14]. A biodiverse ecosystem, which contains many different species and genomes, can best accommodate unanticipated desires (preferences) of both current people in the future and future people. As in the case of insurance value, this can be coupled with considerations of intergenerational equity: high levels of biodiversity now mean many different options for our grandchildren, who may want to extract from ecosystems technological blueprints, substances, and genes for which we currently have no use [15]. In this context, genetic biodiversity is of particular importance, as exemplified by the phenomenon of bioprospecting [16]. Also, preservation of biodiversity of wild forms may have direct implications for agriculture. There are many cases of non-commercial wild varieties of agricultural crops such as rice or coffee whose genetic material could be used to counter diseases or pests [17]. This holds particularly in times of increasingly widespread use of biotechnology: Genetic engineering enables the use of genetic information from completely unrelated organisms, which increases the significance of (the diversity of) non-agricultural wild species for agriculture. While in traditional agriculture only genetic material found in varieties of the same species or, sometimes, some related species, could be used to create new, better crop varieties, today such limits no longer apply, as DNA snippets can be potentially transferred between very different organisms. Thus, the range of options available to respond to future preference changes has become much larger than before the advent of modern biotechnology.

Various elements of an ecosystem provide goods and services to humans, thus influencing their well-being. Furthermore, ecosystems (and some of their elements) can have existence value, at least for some people. Both the supply of these goods and services and the demand for specific elements of the ecosystem are uncertain. However, biodiversity, i.e., the multiplicity of dissimilar items in various

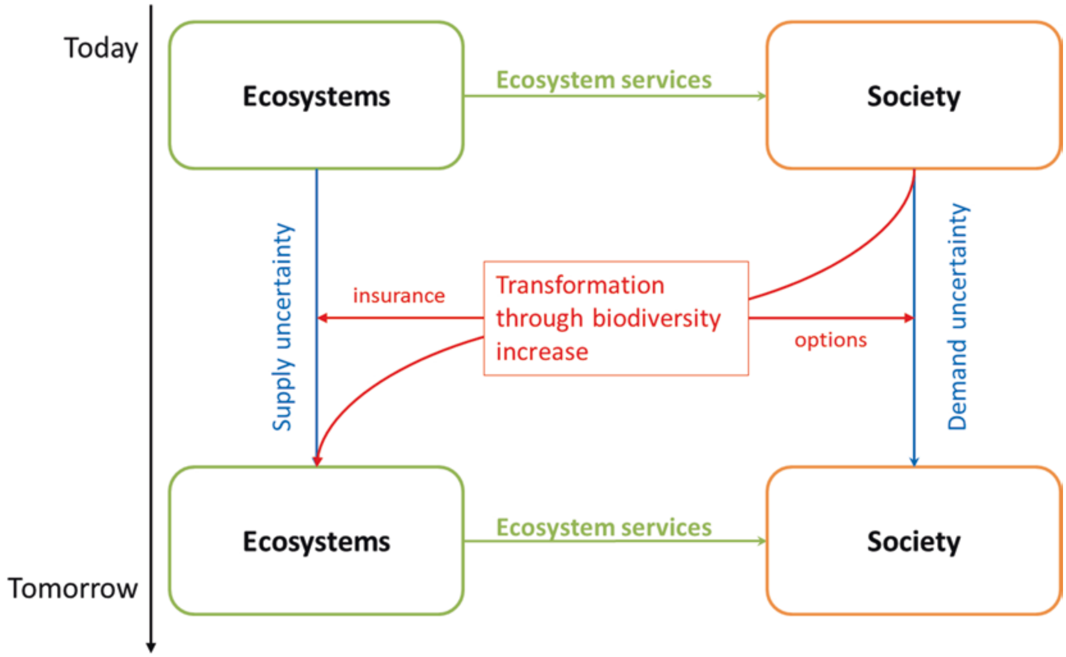

Fig. 5.1 Biodiversity value as result of uncertainty-lowering influence on ecosystem services. The two blue arrows symbolise increasing uncertainty with increasing time-horizons

biotic categories [4], works against both sources of uncertainty. The two relevant effects are depicted as red arrows in Fig. 5.1. First, biodiversity has a stabilising and resilience-increasing effect on ecosystem functioning, which alleviates supply uncertainty. Second, it is a pool of options, which can be drawn upon to accommodate future needs and preferences.

5.3 Estimating the Uncertainty-Related Value of Biodiversity

How can these uncertainty-related values of biodiversity be estimated? First, it is necessary to adapt the common total economic value (TEV) framework to better reflect the fact that we live in an uncertain world. A possible adaptation can be found in Fig. 5.2, which has been inspired by a similar work by Pascual et al. [12].

Compared to the usual TEV framework, this version makes a distinction between economic values prevailing in a world of certainty (use and non-use values), i.e., independent of risk/uncertainty considerations, and those that are dependent on uncertainty about the future, i.e., option and insurance values.

Second, it is essential that suitable valuation methods are used to identify these values. This issue cannot be analysed here in much detail, so a few hints must suffice. It is especially important that the methods used to estimate the economic value of biodiversity should be able to handle the idiosyncrasies of this valuation object. These are, specifically: the non-market nature of many ecosystem services that are "insured" by biodiversity; the inherently subjective nature of option value (i.e., little sources of objective information about the probability of finding useful entities in a given ecosystem); the complexity and abstractness of the concept of biodiversity and the resulting unfamiliarity of stakeholders (i.e., potential valuers) with it. These considerations, together with more general ones [18], help to identify suitable methods for the valuation of biodiversity. The thus identified values can then be fed into political decision-making processes so as to trade-off the benefits of increasing biodiversity levels (in the language of this Atlas, a "provident transformation" response to ecosystem service risk) against the (opportunity) costs of the necessary land-use changes.

5.4 Summary

It has been argued here that the uncertainty surrounding the provision of ecosystem services has two major components: supply uncertainty, i.e., our limited knowledge about the future behaviour of ecosystems; and demand uncertainty, i.e., our even more limited knowledge about future needs and preferences of ourselves and our descendants alike. In other words, we face uncertainty about the provision of ecosystem services, and uncertainty about their identity. Both sources of uncertainty are effectively hedged by conserving and enhancing biodiversity, which has a positive influence on ecosystem

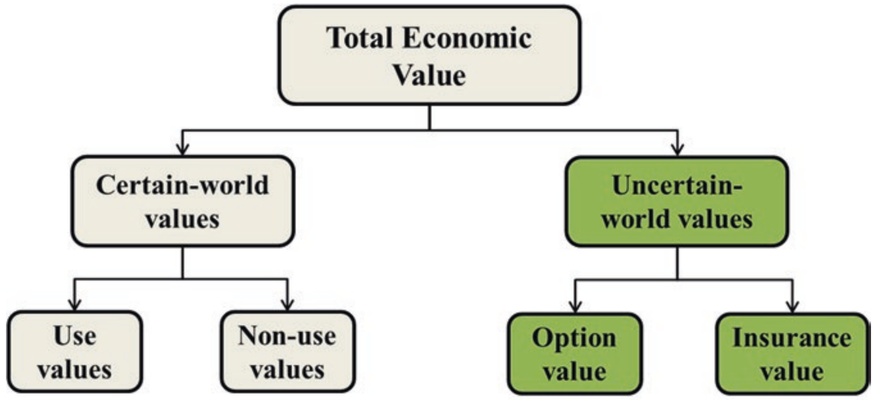

Fig. 5.2 Total economic value (TEV) framework in a world of uncertainty including insurance value. (*From* Bartkowski [19]; *with permission*)

stability and resilience, and which is a pool of options for the future. The identification of these biodiversity values offers information about society's willingness to tolerate risk and uncertainty or rather to hedge against them, and it can help us make the right decisions about how to manage ecosystems in the face of risk and uncertainty.

Acknowledgment *This contribution draws upon results of Bartkowski* [4].

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6 Taking Social Responsibility in Using Ecosystem Services Concepts: Ethical Issues of Linking Ecosystems and Human Well-Being

Kurt Jax

6.1 Introduction

Talking about ecosystem services is to talk about human relationships with nature, and about aspects of nature that humans value. The concept is thus at the interfaces of nature and society as well as of facts and values. From its beginnings, "ecosystem services" was not a merely academic term, but it aimed at human actions. Given this, ethical questions come into play, the foremost being "How should we deal with nature?" and "How does our interaction with nature affect human well-being?" The latter also includes questions of justice, as particular uses of nature may influence not only one's own well-being but also those of other humans – and of non-human beings as well.

While ecosystem services are by definition beneficial to humans, the continued exploitation of ecosystem services may pose risks to humans and to ecosystems. For example, if the flow exceeds the capacity of an ecosystem to produce services, it may become unsustainable; or, specific uses of an ecosystem may be beneficial to some groups of people while putting others, such as those who produce them, at risk.

Ethics, as a theory of morality, deals with the ways humans interact with each other and – in environmental ethics – how humans interact with non-human nature [1, 2]. The ethical question asked is: Which kinds of relations with humans and non-human nature can be justified as morally right or wrong, and on what grounds?

Ethical issues appear in several forms within an ecosystem services context (Fig. 6.1). To understand and tackle these issues, it is necessary to acknowledge that ecosystem services are not simply "out there" in nature; they are dependent on individual and, even more, societal choices. What constitutes an ecosystem service hinges on what we consider to be benefits to humans. A mass production of plants in a eutrophic lake may be considered a disservice because it impedes many uses of the lake, but when the same plant is used for bioenergy, it can be characterised as a service [3]. Moreover, what constitutes a benefit to humans is dependent on what we consider as human well-being. Human well-being is thus not only the end point of a chain leading from nature to society, it must also be considered as the major starting point for conceptualising ecosystem services [4].

Starting from this, we must ask several questions of ethical relevance with respect to conceptualising and using the ecosystem services concept.

Which ecosystem services are addressed? All kinds of services.

What is the research question addressed? What are critical ethical issues in conceptualising and using the ecosystem services concept, and how can we deal with them?

Which method has been applied? Text analysis, thinking.

What is the main result? Major ethical issues questions related to the ecosystem services concept are: Who decides? Which values are included? Who benefits and who carries the burdens from ecosystem services use?

What is concluded, recommended? To account for possible ethical problems, it is important to first be aware of them and to clarify underlying values. The various values and valuation languages should be respected, and possible conflicts and trade-offs be made explicit and mediated. An elaborated idea of what constitutes human well-being is helpful.

6.2 Who Decides?

"Who decides?" This is the foremost and overarching question. Phrased with more detail, it asks: Who has a voice in deciding what constitutes human well-being and what constitutes an ecosystem service? It is a question about power and justice.

If values and aspects of human well-being that are important to some people are not included in the discussion, features

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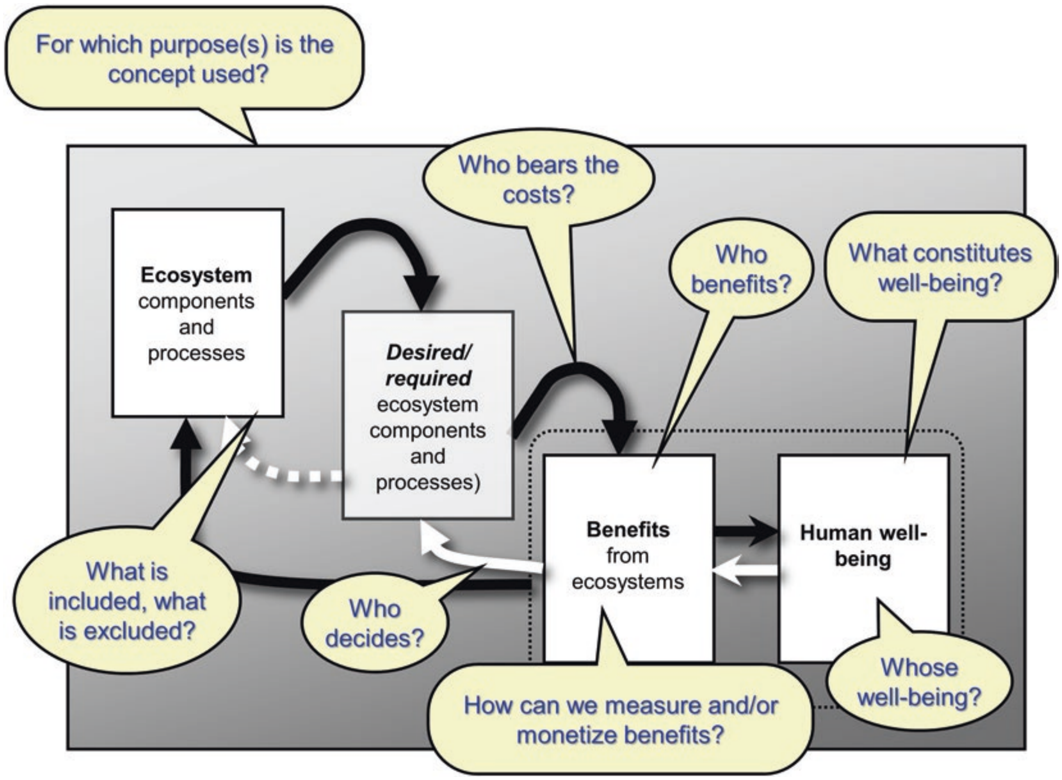

Fig. 6.1 The basic idea (or generic definition) of "ecosystem services" and ethically relevant questions related to the ecosystem services concept. While different definitions exist for the term "ecosystem services," the common idea is the causal relation (black arrows) that some components and processes of ecosystem provide benefits for human well-being. The use of ecosystems by humans in turn affects the systems. What counts as service, however, is subject to societal choices and decisions (white arrows) about what benefits are and which ecosystem processes and components are considered as desirable to promote these. Due to the hybrid nature of the ecosystem services concept, which includes descriptive and normative dimensions (related to values and choices), ethically relevant questions can and should be posed in regard to the different components of the concept and its application. Most of these are dealt with in the text. Note that in some definitions the "desired/required ecosystem components and processes" are called "ecosystem services" in the narrow (measurable) sense, in others they are called the "benefits derived from ecosystems." (*Modified from* Jax et al. [13]; with permission of Elsevier)

of nature that contribute to these values can be neglected, with the further consequence of overlooking entire modes of living and disparate understandings of what constitutes a good life. An example of this is the divergent views of the official Australian conceptualisation of well-being and those of Australia's Aboriginal people. Kinship to natural and mystic objects and access to sacred places, for instance, are crucial to the Aborigines' way of life, but are not covered by the government's definition, which focuses on classical western ideas of well-being [5].

Likewise, as Chan and colleagues have demonstrated, an overly simple and strict classification of ecosystem services can miss the values of some natural phenomena that are the most relevant to people [6]. They found that for First Nations people on the coast of British Columbia, fishing for wild salmon is not just a "provisioning service," as most scientist and economists would classify it. Instead it provides at the same time an array of crucial cultural services. While salmon perceived as a provisioning service alone could be replaced by salmon farming, the cultural values associated to fishing (wild) salmon could not be replaced by this, even if salmon farming would result in equal food and income. This example again shows the major role that understanding (and considering) different ideas of human well-being have for adequately conceptualising ecosystem services.

Detailed and systematic elaborations on human well-being in connection with ecosystem services are still scarce [7]. There is, however, a rich literature on conceptualising and assessing human well-being in other fields, especially development research, psychology, and economy [8].

While some ideas on the components of human well-being will be the same for all humans, many others, which are culturally and environmentally determined, are not. There is a danger of (unwittingly) imposing a simplified and restricted idea of well-being on all humans – based on mainly "western" or "northern" ideas guided by contemporary science and the economic mainstream. A good example of this challenge is the recent conceptual framework of Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Fig. 6.2) [9]. After long negotiations [10], the framework that resulted did not use a single terminology for all participants, but instead developed a framework that put important concepts in parallel (blue and green colours). Bolivia and Ecuador, for instance, rejected

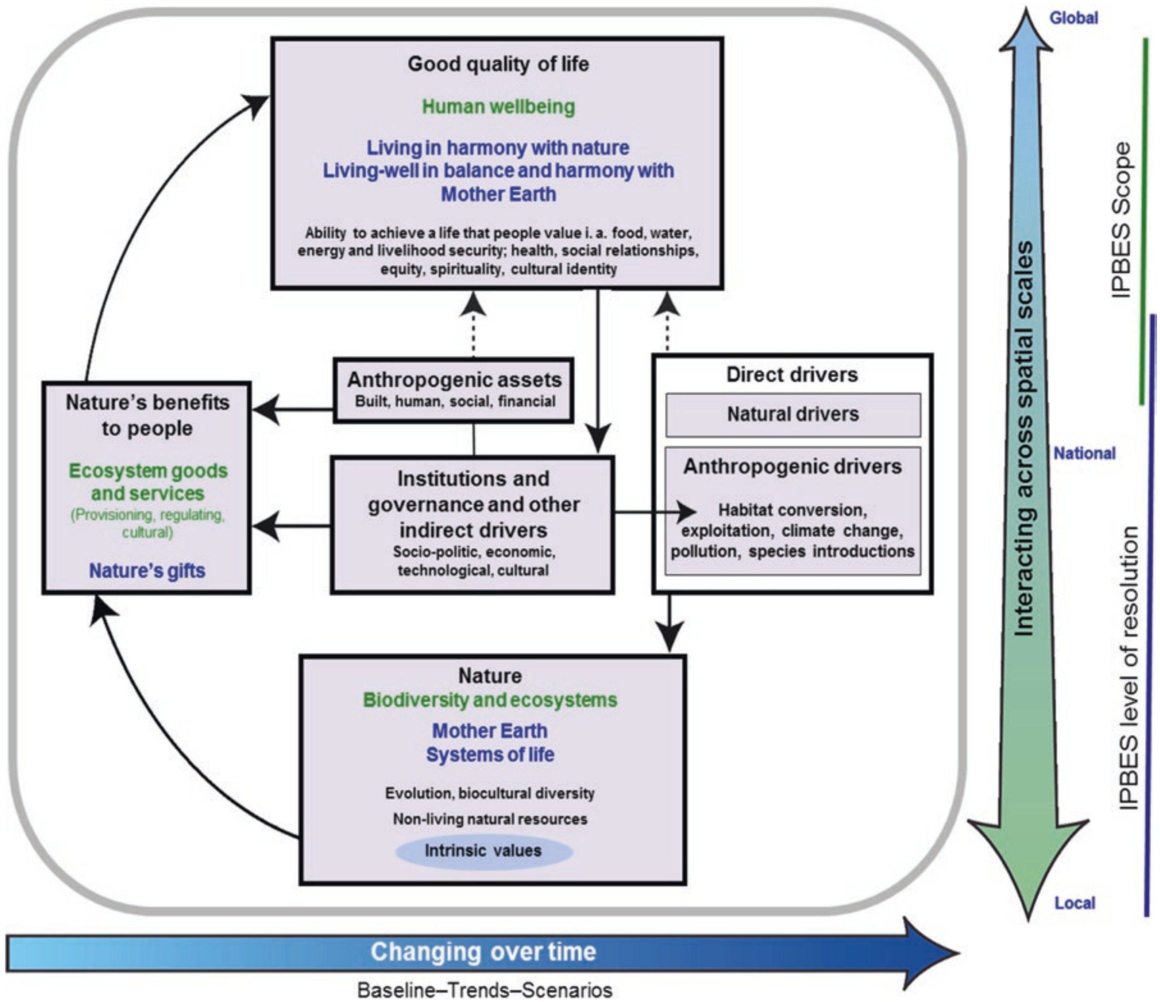

Fig. 6.2 The IPBES conceptual framework depicts, in a simplified form, a model of the major relations between humans and the natural world, focusing on those elements which are crucial for the goals of IPBES and which "should therefore be the focus for assessments and knowledge generation to inform policy and the required capacity-building." (*Reprinted from* UNEP [9]; with permission of the IPBES secretariat)

concepts like "services" and "ecosystems and biodiversity" and instead promoted a non-western idea of human-nature-relationships building on the notion of "mother earth" and "living in harmony with nature." Although some researchers [11] have called the framework a "Rosetta Stone" for biodiversity (and ecosystem services) concepts, it is not, because this implies that the terms in different colours are "translations" with identical meanings. Instead there is a persisting tension between the different ways of conceptualising human–nature relationships. There is a major difference between, for example, a *service* (with strong economic connotations) and a *gift*. Keeping up the tensions here is productive, as it allows for different voices to be heard. It may also prevent the pitfall of articulating an application of the ecosystem services concept that has not been sufficiently reflected upon, such as assuming that all people share the same the same idea of human–nature relationships. The technical and economic connotations implied in words like *system*, *service*, and *capital* may further the impression that scientists and policy-makers try to impose a value set and worldview that is not shared by, for example, many indigenous people and conservationists.

Finding adequate societal responses to solving local and global problems depends not only on gaining acceptance for specific measures, but on involving an array of stakeholder groups in formulating the specific problem (e.g., What are the important services? What are options for responses that are in agreement with values also of minority groups?) instead of imposing a dominant scientific expert perspective.

6.3 Which Values and Which Objects of Nature Are Included–and Which Are Not?

An ethical issue of major concern to conservationists is determining the kind of values and objects that are covered by the ecosystem services concept, and those that are possibly neglected and consequently impaired by using it. This is an issue that poses risks both to non-human beings as well as to the ideas of a good relationship with nature embraced by people. Excluding some aspects of nature from an ecosystem services framework will put these aspects at risk when decisions are taken on how to deal with nature are based on the ecosystem services framework. What we do not perceive as an ecosystem service may not be considered as being of value, and thus neglected.

The ecosystem services concept initially deals with nature within a utilitarian framework, i.e. one that explicitly aims at the use(fulness) of nature for humans in terms of fostering human well-being [12]. This does not mean, however, that "value" as understood in an ecosystem services context is restricted to an economic or even purely monetary value. In principle, the values of human nature to which the ecosystem services concept can relate range from purely instrumental values (nature as a pure means) through eudaimonistic values. The latter describe a non-instrumental approach to nature, where the *relation* to nature in itself (and not just as a means) is valuable for a good, flourishing human life [13, 14]. While most values of nature embraced by people may be within the scope of an ecosystem services approach, there is always a danger that values may be perceived (especially by politicians and the broader public) as much more restricted, namely in purely economic and even monetary terms. This leads to the commodification of nature, i.e., the transformation of ecosystem components or processes into products or services that can be privately appropriated, assigned exchange values, and traded in markets [15]. This is at odds with valuing the uniqueness of certain features of nature, as is common in biodiversity conservation and in everyday relations of humans with nature. Some items (e.g., some species) that are valued may thus become invisible or excluded from consideration because they are beyond an instrumental perspective on nature. The values embraced in ecosystem services protection and in biodiversity conservation overlap, but do not simply coincide [16]. What is clearly not covered by ecosystem services, however, are intrinsic values, understood as values for nature, independent of any human well-being and interests [17].

6.4 Who Benefits from the Use of Ecosystem Services and Who Carries the Costs?

When we look at where ecosystem services are produced (and at who's cost) and where they are consumed (and to who's benefit), questions about justice and equity come to the fore. The costs of and benefits derived from the provision and use of ecosystem services can be, and often are, distributed very unevenly among different regions and/or social groups, e.g., when the production of crops for food, feed, or bioenergy imported to highly industrialised countries threaten the livelihood of people elsewhere (Box 6.1). This distributional issue is important across all scales, both spatial (e.g., global gain, local loss, or vice versa) and temporal, (i.e., use by present generation vs. options for future generations).

The degree to which the ethical issues described above become crucially relevant depends on the ways in which the ecosystem services concept is defined and used. If the concept is used more in a didactic purpose, i.e., to generally raise awareness of the value of nature, few of these problems will be pertinent. If ecosystem services, however, become the basis of planning and management, the potential problems necessarily have to be considered and accounted for. To avoid or at least attenuate problems, it is important to be aware of them, to clarify the application purposes as well as the underlying values and social contexts – including power relations – in the specific situation. The various values and valuation languages should be respected and possible conflicts and trade-offs be made explicit and mediated [13].

Box 6.1: Benefits and Drawbacks of Soy Production

Especially after the occurrence of mad cow disease in the 1990s and the resulting strong restrictions on feeding meat and bone meal to cattle, Europe's import of soybeans from South America for feed significantly increased. According to World Wide Fund for Nature (WWF), demand for soy within the European Union requires an area of almost 15 million ha, 13 million ha of which is in South America (Fig. 6.3). This area would amount to ca. 90% of the whole agricultural area of Germany. Around 75% of soy production worldwide is used for feeding domestic animals, i.e. mostly for meat and dairy production [18]. In South America this has further increased deforestation and the transformation of other natural ecosystems (Fig. 6.4). Industrial soy production also involves the use of high loads of pesticides, which contribute to diseases among the local human population. While people in Europe thus largely profit from the flow of this provisioning service between the continents (better meat, less pressure on European land use), many people in South America carry the burden of this production. Although many gain income and profit there, many people are also affected negatively, e.g., through health problems, poor working conditions, loss of previous natural and semi-natural ecosystem, loss of biodiversity. We also see a trade-off between different services here, i.e., between the production of soy as a provisioning service on the one hand and – as a result of land transformation – a decrease of regulating services such as (global) climate regulation, regulation of (local) water quality, regulation of erosion, and others. Such impacts will also influence the livelihoods of future generations.

Fig. 6.3 Soybean field in the Province of Buenos Aires, Argentina. Source: Image by "Alfonso" CC BY-SA 3.0, https://commons.wikimedia.org/w/index.php?curid=1161341

Box 6.1: (continued)

Fig. 6.4 Many South American landscapes are at great risk from soy expansion. By transforming ecosystems in the interest of producing one specific ecosystem service, strong trade-offs exist with other ecosystem services. Biodiversity and human livelihoods may also be compromised. (*Reprinted from* World Wide Fund for Nature [18]; with permission)

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Part II

Drivers and Their Risks for Ecosystems, Their Functions, and Services

7 Introduction to Part II: Drivers and Their Risks for Ecosystems, Their Functions, and Services

Cornelia Baessler and Stefan Klotz

The first framework for characterizing human impact on biological systems was developed to study biotic responses of different human impact factors like air, water and soil pollution, and general land use. This approach was defined as "biomonitoring" or "bioindication" [1]. Different levels of indication were defined from genetic, biochemical, physiological, morphological, population, and species responses, up to responses of ecosystem structures and functions. Different stressors were defined as drivers like physical stressors (e.g., changes in radiation, temperature extremes, droughts etc.), chemical stressors (e.g., air, water, and soil pollutants) and biotic stressors (e.g., alien invasive species, new pathogens, etc.). Drivers themselves are natural and/or human-induced factors having a direct or indirect impact on biotic systems. But most of the drivers and their effects are very complex and occur mostly as an assemblage of physical, chemical, and biological stressors like land use change, climate change, etc.

The framework for stressors and the related responses of ecosystems used for biomonitoring was further developed by the Organisation for Economic Co-operation and Development (OECD) [2] and the United Nations [3] by defining indicators of sustainable development. The European Environmental Agency (EEA) proposed a more detailed concept, the Driver-Pressure-State-Impact-Response (DPSIR) framework [4].

A first ranking of major drivers affecting biodiversity was published by Sala et al. [5]. Land use change, climate change, changes in the CO_2 concentration, nitrogen pollution, and alien invasive species were defined as main drivers.

In the Millennium Ecosystem Assessment Framework [6], the ecosystem services concept relates direct and indirect drivers directly to ecosystem services and human well-being, including possibilities of human interventions.

A detailed list of general drivers, sub-drivers, and their possible consequences are given by Klotz (Table 7.1) [7].

In the framework chapter of the Atlas (Chap. 1), we have distinguished two types of drivers' consequences: first order risks and second order risks. First order risks include potential losses of processes and properties of the ecosystems, whereas second order risks comprise potential losses of ecosystem services (Fig. 7.1). In the chapters of this part, the complex relations between drivers and their risks for ecosystems, their functions, and the services they provide are addressed for different systems.

7.1 Main Drivers

Understanding the driving forces of environmental and social changes, their impacts, and their relationship to decision-making constitutes a major challenge for scientists and policy makers. Drivers of change operate at various scales, which do not always match the scales that are relevant for organisms or ecosystem functions [8]. In the first chapter of this part, Henle et al. (Chap. 8) present a recently developed tool to measure the scale-sensitivity of drivers of biodiversity and ecosystem services change over multiple scales, and illustrate the scale sensitivity of selected drivers including urbanization, tourism, agriculture, and habitat fragmentation.

7.1.1 Land Use and Land Use Changes

Habitat loss and fragmentation associated with climate and land use change entail major risks for biodiversity, ecosystems, and their functions at different spatial scales and organizational levels of the systems. Based on a review, Michalski (Chap. 9) explores the possible links between genetic diversity and ecosystem services and argues that a loss of genetic diversity can constitute severe risks for community-based processes. Genetic as well as species diversity are part of ecological variables that are important for provisioning ecosystem services. To simulate changes in different ecological variables and to project ecosystem services at regional scale, Boit et al. (Chap. 10) are using Dynamic Global Vegetation Models (DGVM). Another recent method introduced by Fischer et al. (Chap. 11) is the combination of forest

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Table 7.1 Classification of drivers and their possible consequences for biodiversity, and ecosystems [7]

Driver	Sub-driver/pressures	Consequences
Land-use change	• Overexploitation • Urbanization • Fragmentation • Isolation • New landscape configuration • Application of pesticides, insecticides and synthetic fertilizers	• Habitat loss • Ecosystem loss • New anthropogenic ecosystems • Changed ecosystem structures and functions • Species loss • New species invasions • Changes in matter fluxes • Loss of genetic diversity
Climate changes	• Temperature • Precipitation • Wind • Weather extremes	• Changed phenology • New distribution ranges of species • New species composition in ecosystems • Changed ecosystem structures and functions • Genetic drift • Species loss • Species invasion
Changes in matter fluxes	• Increasing carbon dioxide concentration in the atmosphere • Pollution by nutrients • Pollution by other chemicals (pesticides and pharmaceuticals etc.)	• Food webs/interaction networks • Competition • Toxic effects • Accumulation of toxic chemicals in organisms • Species loss • Species invasion
Biological Invasions	• Plant and animal invasions • Diseases spread • Genetically Modified Organisms (GMOs)	• Species loss • Changed ecosystem structures and functions • Hybridization • Anthropogenic evolution • Global homogenization of species diversity

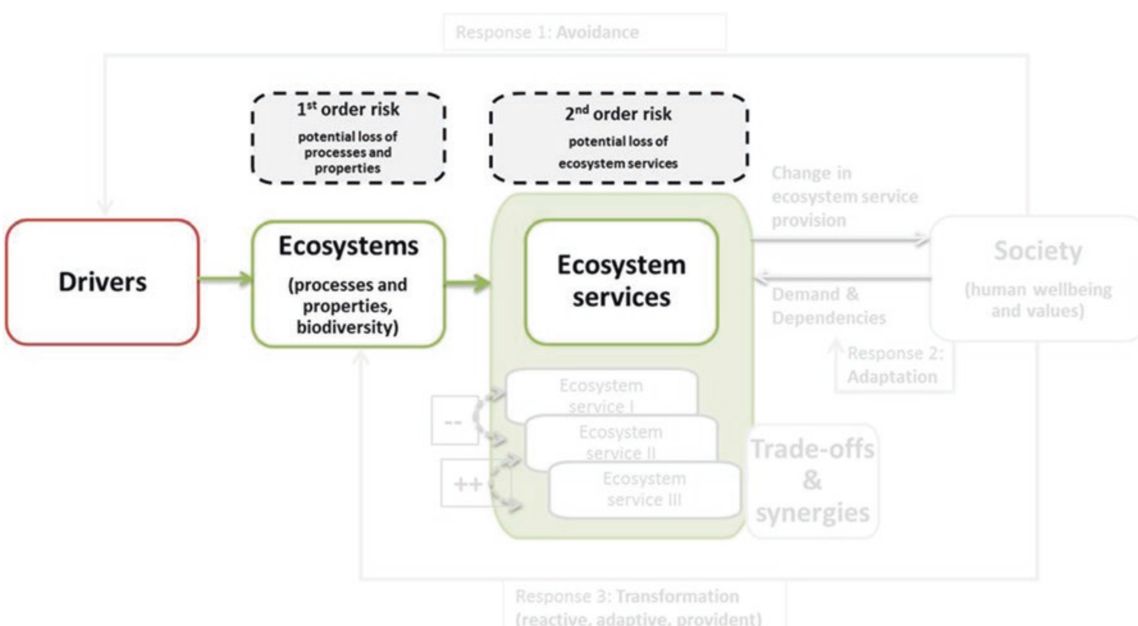

Fig. 7.1 Elements of the framework of the Atlas of Ecosystem Services addressed in this part

modelling and remote sensing. They use this method to meet the challenge of estimating forest properties like biomass or productivity for larger regions. Whereas this approach is focused on forests at different spatial scales, the concept of Land System Archetypes (LSA) introduced by Vaćlavík et al. (Chap. 12) provides a more holistic representation of global land use patterns. The authors also illustrate its use for identifying drivers of ecosystem service risks and the potential to increase resilience of particular regions. Geijzendorffer et al. (Chap. 14) assessed the temporal changes of the resilience of Mediterranean wetlands and their capacity to provide ecosystem services in the face of land use changes driven by societal demands. Land use and climate change are also main drivers for the loss of fertile soil through degradation and desertification. Vogel et al. (Chap. 13) highlight the central role of soil for the functioning of terrestrial systems and thus for the services they provide. They therefore propose that soil functions need to be evaluated based on a systemic model concept and in a site-specific way. They also emphasize the importance of sustainable landscape management. This aspect is also discussed by Loos et al. (Chap. 15) in the context of the maintenance or enhancement of functional diversity that provides ecosystem services like biocontrol and pollination. Schweiger et al. (Chap. 17) could show that sustainable land management increases resilience of pollinator communities and thus counterbalance the negative influence of climate change impacts on pollinators. Accordingly, Lautenbach et al. (Chap. 16) show that the spatial distribution of the benefits created by pollinators is uneven, and varies across different archetypes of land systems.

Infrastructures for energy production – both fossil-based and renewable – represent types of non-traditional land use that impose different kinds of stress compared to the stress imposed by traditional land use types such as agriculture. Dotzauer et al. (Chap. 18) show how biogas production pushes risk factors for the provisioning of ecosystem services. Biogas plants can provide synergies, but also tradeoffs, with ecosystem services, and can pose risks to biodiversity, soil fertility, and clean groundwater. In more general terms, Strunz et al. (Chap. 19) discuss the effect of energy infrastructure for ecosystems (e.g., by habitat fragmentation – first order risk) or the services they provide (second order risk). They also consider the important topic of the spatial match between energy production, technology, and policies in Europe. This aspect is exemplified in more detail for the case of wind power by Lauf et al. (Chap. 20), who ask which institutional and non-institutional factors drive the spatial allocation of wind power deployment and their impacts on ecosystem services. Beside bioenergy production, Priess et al. (Chap. 21) depict reforestation, expansion of urban land, organic agriculture, and change in agricultural productivity as the main factors driving land use change in Central Germany. They used scenarios to address at least some potential future developments and some of the consequences and risks. In Chap. 22, Priess et al. explore the applicability of specific scenarios for science and policy-making at different scales, including the European level and regional and local scales. The authors use these scenarios to assess the uncertainties and risks related to land use change by simulating results for a provisioning and a regulating service. Potentials and risks for provisioning services (e.g., food production) as well as for regulation services (e.g., surface water retention, air temperature regulation, pollutant filtration) in cities and the adjacent open land are shown in the next three chapters. Haase et al. (Chap. 23) and Koch et al. (Chap. 24) demonstrate the effects of the transformation of vegetated into built-up areas, i.e., land consumption, within the city and along the urban-to-rural gradient of the city of Leipzig, Germany. Banzhaf et al. (Chap. 25) assess the influence of socio-spatial differentiation on urban green infrastructure and its supporting ecosystem services in Santiago de Chile.

7.1.2 Climate Change

Climate change is already considered to be a major pressure on ecosystems and a driver of biodiversity and ecosystem changes in some of the previously introduced chapters. Not only mean values, but also extreme events, e.g. heavy precipitation, droughts, and heat waves, are affected by climate change [9]. A variety of effects on ecosystem structure and functions, e.g., changes in phenological stages, in species composition, in species interactions, in the migration of species, and in soil conditions have been observed and analyzed so far. Beside urbanization, green spaces in cities, for instance, are increasingly threatened by changing temperatures and precipitation due to climate change. Knapp et al. (Chap. 26) assess these impacts on the ability of trees to cool their surroundings. Another aspect in relation to climate change impacts are "agricultural droughts" that have already affected ecosystems and, e.g. their ability to store CO_2 [10]. Marx et al. (Chap. 27) examine the influences of climate change in general across Europe and the role of soil drought in Germany. They conclude that the impacts of climate and the resulting changes in ecosystem services demonstrate the need for sustainable management and climate adaptation strategies or climate mitigation, respectively.

7.1.3 Changes in Matter Fluxes

The third major driver of changes in biodiversity, in ecosystem functions, and thus a driver of ecosystem services risk is the pollution of ecosystems by a constantly increasing load of anthropogenic chemicals, i.e., chemicals caused or produced by humans. Wick and Chatzinotas (Chap. 28) argue that the "combined action of multiple anthropogenic chemicals are of particular concern because mixtures of chemicals may cause effects even when individual chemicals are present at concentrations too low to be individually effective." The natural microbial communities are the principal actors for decontamination. The authors describe factors that enable and limit microbial ecosystems to biodegrade anthropogenic organic chemicals, looking at both the compatibility of a chemical and the capability of the ecosystem for biodegradation. De Vries and Schulte-Uebbing (Chap. 29) show the impact of nitrogen deposition as an additional human-induced driver on forest carbon sequestration at the global scale. They also discuss appropriate management options to enhance the beneficial as well as to reduce the adverse impacts of nitrogen loads in forests.

Contamination with chemical and emerging pollutants is also one of the most important risks to the ecosystem services provided by aquatic ecosystems. In this context, Rinke et al. (Chap. 30) discuss ecosystem services provided by aquatic ecosystems and their risks in general, and explore options to manage nutrient retention in these systems. Griebler et al. (Chap. 31) then give an overview of ecosystem services provided by groundwater systems and the main anthropogenic threats on a global scale. Völker and Borchardt (Chap. 32) evaluate the drinking water quality in Europe. They conclude that rising contamination with anthropogenic chemicals and nutrients and changes in the hydrological regime due to climate changes lead to an increasing risk for drinking water quality. Using regional as well as global examples, Knillmann and Liess (Chap. 33) explain the risks of pesticides for invertebrate communities, including relevant ecosystem functions and services in flowing waters, and discuss expected developments under global climate change.

7.1.4 Biological Invasions

Biological invasion is a global process that influences all main ecosystems – marine, freshwater, and terrestrial. Global spread of species is one important cause of biotic homogenization, which leads to increasing similarity among different ecosystems. The invasion or introduction of alien species into ecosystems might be considered a proliferative problem as well. That is, such invasions may diminish the resilience of ecosystems in the face of further change. Biological invasions may thus have different impacts on native biodiversity, ecosystem functions, and their resulting services, as well as on their economic values. Knapp et al. (Chap. 34) explain exemplary existing trade-offs between the services and disservices provided by non-native species. Kraberg et al. (Chap. 35) discuss the impact as well as the central cause-and-effect relationships of non-native plankton species on marine ecosystem services. However, the introduction of a non-native species does not always have detrimental effects on ecosystems and their services, as shown by Gutow and Buschbaum (Chap. 36) with regard to the Pacific oyster in the Wadden Sea. As the consequences of species invasions are often unpredictable, it is necessary to distinguish between those species and their characteristics that provide services and those that promote invasions or provide disservices.

Finally, Koellner et al. (Chap. 37) ask whether the import of biomass leads to negative impacts on ecosystem services in exporting regions as another aspect, beside biological invasions, that results from international trade and global flows of ecosystem services. The relationships of this aspect are explained by the authors using the example of EU soybean imports.

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8 Scaling Sensitivity of Drivers

Klaus Henle, Joseph Tzanopoulos, Pascal Marty, Vesna Grobelnik, Raphaël Mathevet, and Anna V. Hetterley

Understanding drivers of change, their impacts on biodiversity and ecosystem services, as well as their relationships to decision-making, constitutes a major challenge for scientists and policy makers. As we move across scales, the intensity as well as the spatial distribution of a driver may change. Drivers' impacts on biodiversity and ecosystem services are thus scale sensitive, and it is necessary to analyse and describe the way drivers of change operate over multiple scales. Here we present a recently developed tool to assess the scale sensitivity of drivers of biodiversity and ecosystem services and illustrate the scale sensitivity of selected drivers (urbanization, tourism, agriculture, habitat fragmentation) in Europe. For example, Poland and Germany tend to show a similar pattern in the share of surfaces affected by agricultural conversion. However, mapping similar data at a regional level reveals some strong regional contexts. Globally, Polish regions have medium rates of conversion, whereas a contrasting pattern is observed in Germany. While strong agricultural conversion can be observed in East Germany, 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. However, not all drivers are scale sensitive. For example, change in the evenness and intensity of Gross Domestic Product is minimal as we move across scale. Based on the different reaction of drivers to the scale of assessment, we derived a typology of scale sensitivity of drivers. Finally, we illustrate that the results of assessments of habitat fragmentation are also highly scale sensitive.

8.1 Introduction

Why do we go on losing biodiversity and ecosystem services? The answer depends on the scale at which we view the world. At one scale it may be habitat fragmentation or homogenization, whereas at another scale it might be climate change. For instance, farmland species living in extensive agricultural landscapes are heavily impacted by intensive farming [1, 2] and it is the same for pollination services [3]. Management of the living world will be effective only if we understand how problems and solutions change with scale.

Understanding drivers of change, their impacts on biodiversity and ecosystem services, as well as 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., the 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 organisation of the analysis; in other words, the scale of the analysis. This is because drivers of change operate at various temporal and spatial scales that do not always match the scales that are relevant for understanding organisms, ecosystem functions, and the ecosystem services they provide. In addition, the way drivers operate or appear over multiple scales is non-linear [4–6]. Indeed, as we move across scales, the intensity as well as the spatial distribution of a given driver may change. Drivers' impacts on biodiversity and ecosystem services are thus scale sensitive, and it is necessary to analyse and describe the way drivers of change operate over multiple scales.

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 [4] and ecosystem services [3, 7]. Depending on the administrative organization of a country and how competences for environment are distributed in different levels, conservation of a habitat covering several administrative units may be addressed in different ways. These mismatches may make our efforts to manage the living world ineffective and even counterproductive.

Here we present a recently developed tool to assess the scale sensitivity of drivers of biodiversity and ecosystem services and to illustrate the scale-sensitivity of selected drivers in Europe. More examples of the scale-sensitivity of drivers can be explored with the SCALETOOL (http://scales.ckff.si/scaletool/index.php?menu=1&submenu=0&pid=5&nut=0) [8] that was developed by the SCALES project [9].

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8.2 Development of a Tool for Quantifying and Assessing the Scale Sensitivity of Drivers

To quantify the scale sensitivity of indicators, Tzanopoulos et al. [5, 6] used two key variables across administrative levels, "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 Fig. 8.1. Change in intensity (I) is assessed by measuring the relative change in the median of a driver at a given administrative level compared to NUTS level 0. The term NUTS stands for "Nomenclature of territorial units for statistics" where NUTS 0 indicates countries, NUTS 1 major socio-economic regions

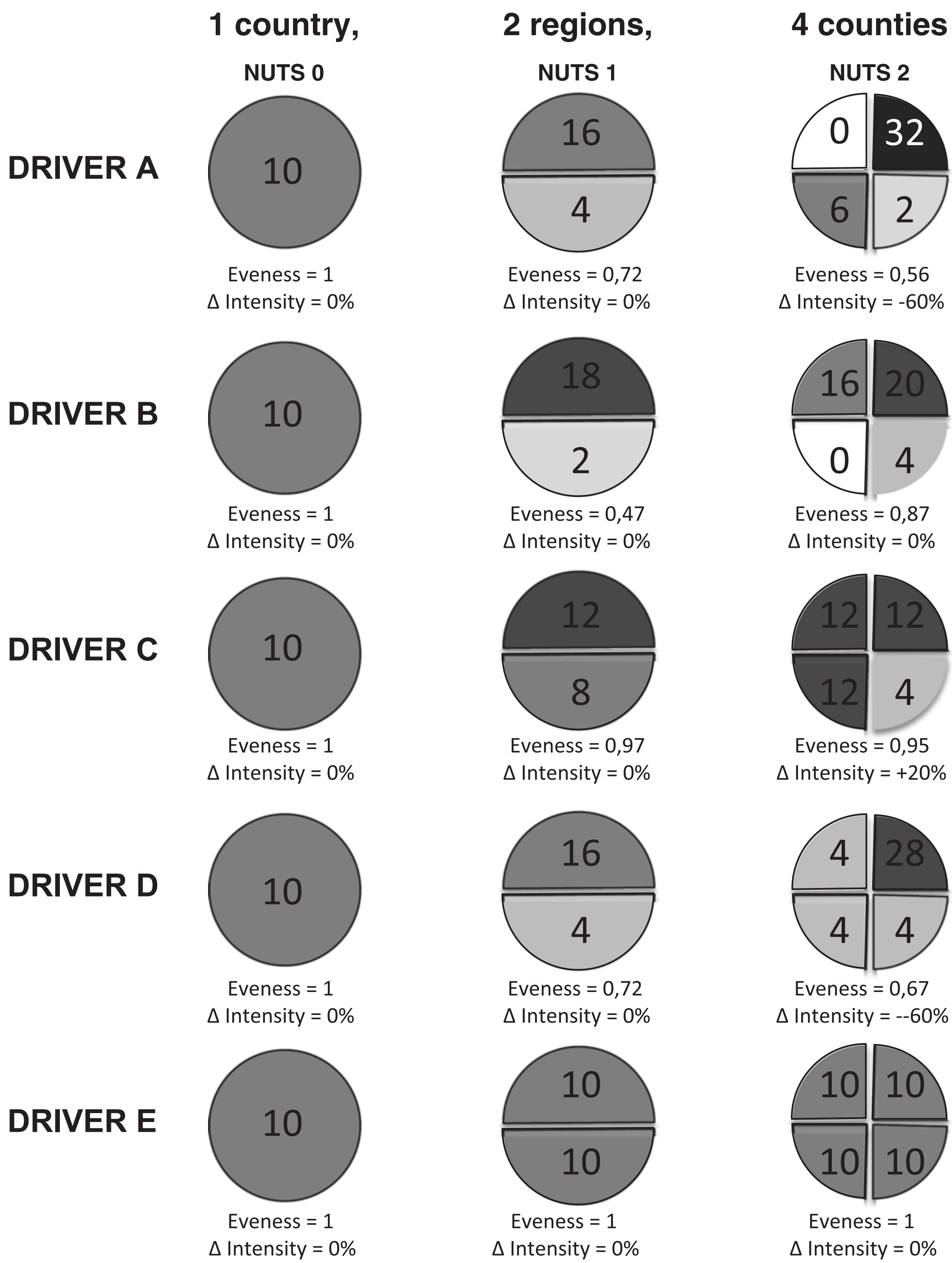

Fig. 8.1 Conceptual framework to assess the scale sensitivity of drivers (**A**–**E**). Change in intensity is measured as a change in median intensity. Change in evenness is measured as the difference in the Shannon's Evenness Index to the next higher NUTS level. (*Based on the approach of* Tzanopoulos et al. [6])

within countries, NUTS 2 basic regions, and NUTS 3 small regions (see http://epp.eurostat.ec.europa.eu/portal/page/portal/nuts_nomenclature/introduction).

Intensity is set equal to zero at the highest level, in the case of the EU at NUTS level 0. It can either be positive or negative for other levels. Evenness is measured using Shannon's Evenness Index (E), which is derived from Shannon's diversity index. The values of the above two variables for each driver across all different administrative levels then can be plotted on a two-axes graph, which provides a visual summary of the relative scale sensitivity of each driver of change.

8.3 Mapping the Scale Sensitivity of Drivers

Maps provide a visual impression of spatial structures and trends in the distribution of drivers over Europe. They allow highlighting main spatial patterns, including disparities and clustering. For example, Fig. 8.2 shows trends of agricultural conversion over Europe (between 1990 and 2000) with the exception of Finland, Sweden, and Cyprus. Agricultural conversion is defined as agricultural land that has been transformed to forests and semi-natural areas,

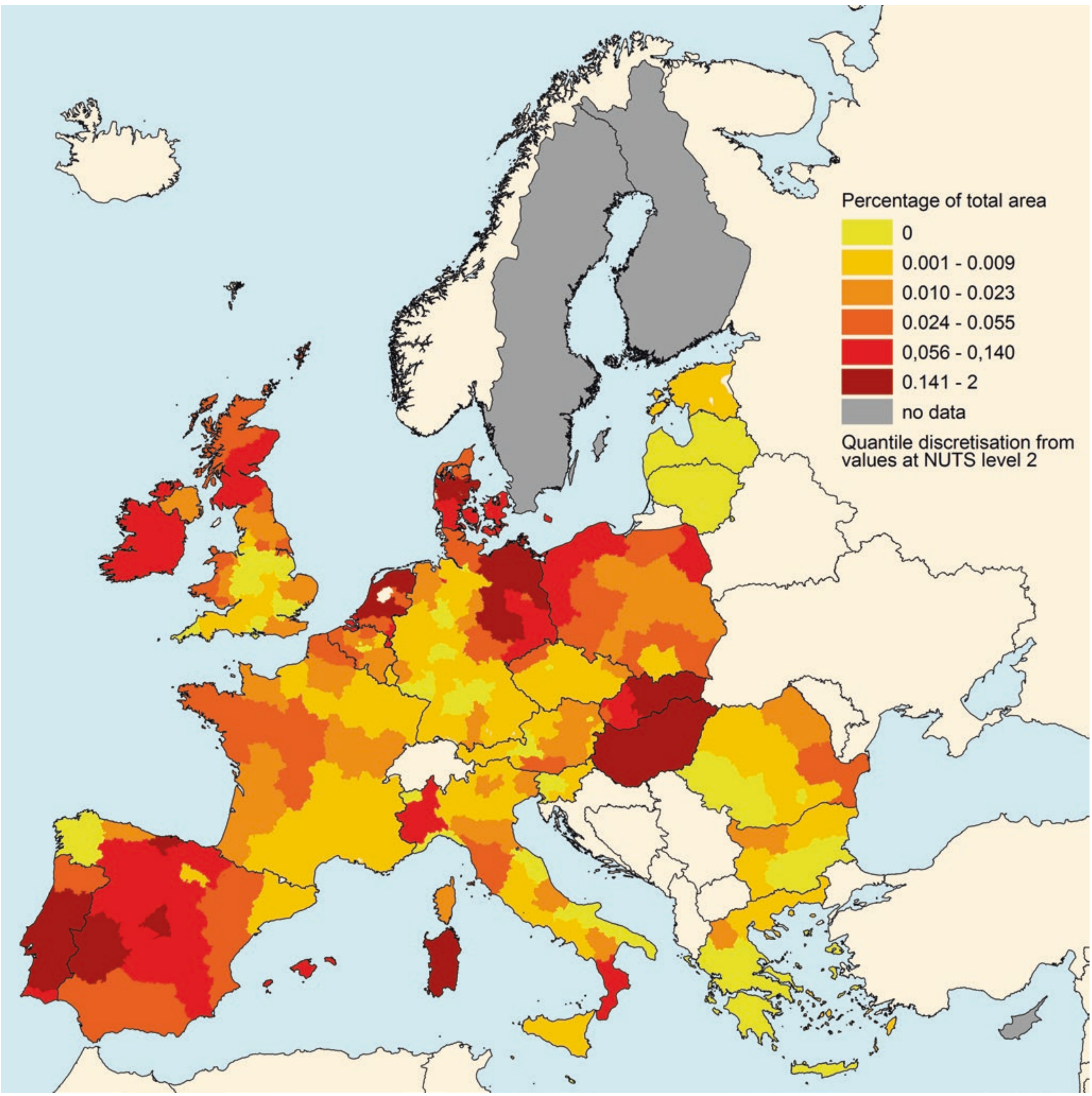

Fig. 8.2 Agricultural conversion at NUTS 2 in Europe between 1990 and 2000. (*Data from* CORINE Land Cover https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

excluding any conversion to residential use, and it is expressed as a percentage of total area. High values of conversion tend to be concentrated in Eastern and Southern Europe, whereas some regions seem to be spared from this process. A combination of reasons can explain agricultural conversion, such as soil quality, terrain, climate, shifts in agricultural policy, modernization of agricultural sector, and land reforms [1, 10, 11, 12].

Furthermore, mapping enables the visual representation, exploration, and comparison of the scaling properties of drivers (i.e., change in evenness, change in intensity). An example of visual representation of change in intensity across scale is provided in Fig. 8.3, where change in urbanisation between 1990 and 2000 is mapped at multiple administrative levels (NUTS 1–NUTS 3). This figure shows that as we move from NUTS 1 to NUTS 3, urbanisation appears more intensive. This is because urbanisation is widespread and there are many NUTS 3 areas that experience an increase. However, at a higher level of aggregation (NUTS 1) there is a smoothing effect. The opposite trend can be observed – for example, for wetland loss which is taking place only in specific localities – and therefore the median intensity at NUTS 3 is lower than the median intensity at NUTS 0 (Fig. 8.4).

On the other hand, an example of visual representation of change in evenness across scale is provided in Fig. 8.5, which maps change in tourism intensity between 1990 and 2000. Tourism at NUTS 0 appears more unevenly distributed at the national level (NUTS 0) compared to the local level (NUTS 3). This is because although tourism intensity differs considerably among countries, there are many hotspots of tourism development scattered within each country. However, not all drivers are scale sensitive. For example, change in the evenness and intensity of Gross Domestic Product (GDP) is minimal as we move across scale (Fig. 8.6).

Examples of the difference in the overall scale sensitivity of drivers are also presented in Figs. 8.7 and 8.8. Figure 8.7 focuses on the situation in Germany and Poland. At NUTS level 0, both 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 (e.g., Eastern Germany was strongly affected by decollectivization and transition to market economy). Conversely, mapping GDP in southern Sweden at NUTS levels 2 and 3 does not highlight many differentiations (Fig. 8.8). The underlying phenomenon stands in the homogeneous GDP distribution over the five NUTS 3 regions.

8.4 Typology of the Scale Sensitivity of Drivers

A typology of the scale sensitivity of drivers can be developed summarising in five classes indicators with common characteristics regarding their scale behaviour (Table 8.1). Classes behave very differently as we move across levels. In class 1, the change in evenness and intensity is small. The already high evenness suggests that there are limited differences in the change in intensity of the corresponding drivers across EU regions. Class 2 shows a similar kind of behaviour. However, slight differentiations across administrative levels are more apparent: evenness is lower than in class 1 and tends to increase when moving to lower levels. The following classes (from 3 to 5) can be characterized as much more scale sensitive. Class 3 displays both an increase in evenness and an increase in intensity when moving to lower levels. This class groups together drivers that tend to be relatively widespread over EU administrative units. Classes 4 and 5 show an increase in evenness and a decrease in intensity at lower administrative levels. They differ in the amplitude of variation, class 5 having much stronger variations. Such changes entail the existence of spatial clustering and hotspots in the repartition of the corresponding drivers.

8.5 Scaling of Habitat Fragmentation

Habitat loss and fragmentation are key consequences of several drivers, including scale-sensitive ones such as urbanization or agricultural conversion. In turn, the degree of habitat loss and fragmentation and their effects on biodiversity and ecosystem services, e.g., pollination, will be scale sensitive [7]. Species differ substantially in their dispersal ability. Therefore, the scale at which the landscape is perceived by each of these species varies and has a differing effect on their ability to survive and the ecosystem services they provide.

Analysis of fragmentation and habitat loss are very sensitive to scale. The example given below uses CORINE Land Cover Maps from the European Environment Agency to examine changes in the structure of the landscape and to highlight the most vulnerable regions. We examined changes in the structure and abundance of land cover types between 1990 and 2006. Fragmentation of the CORINE land covers 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) [13]. Figures 8.9 and 8.10 illustrate

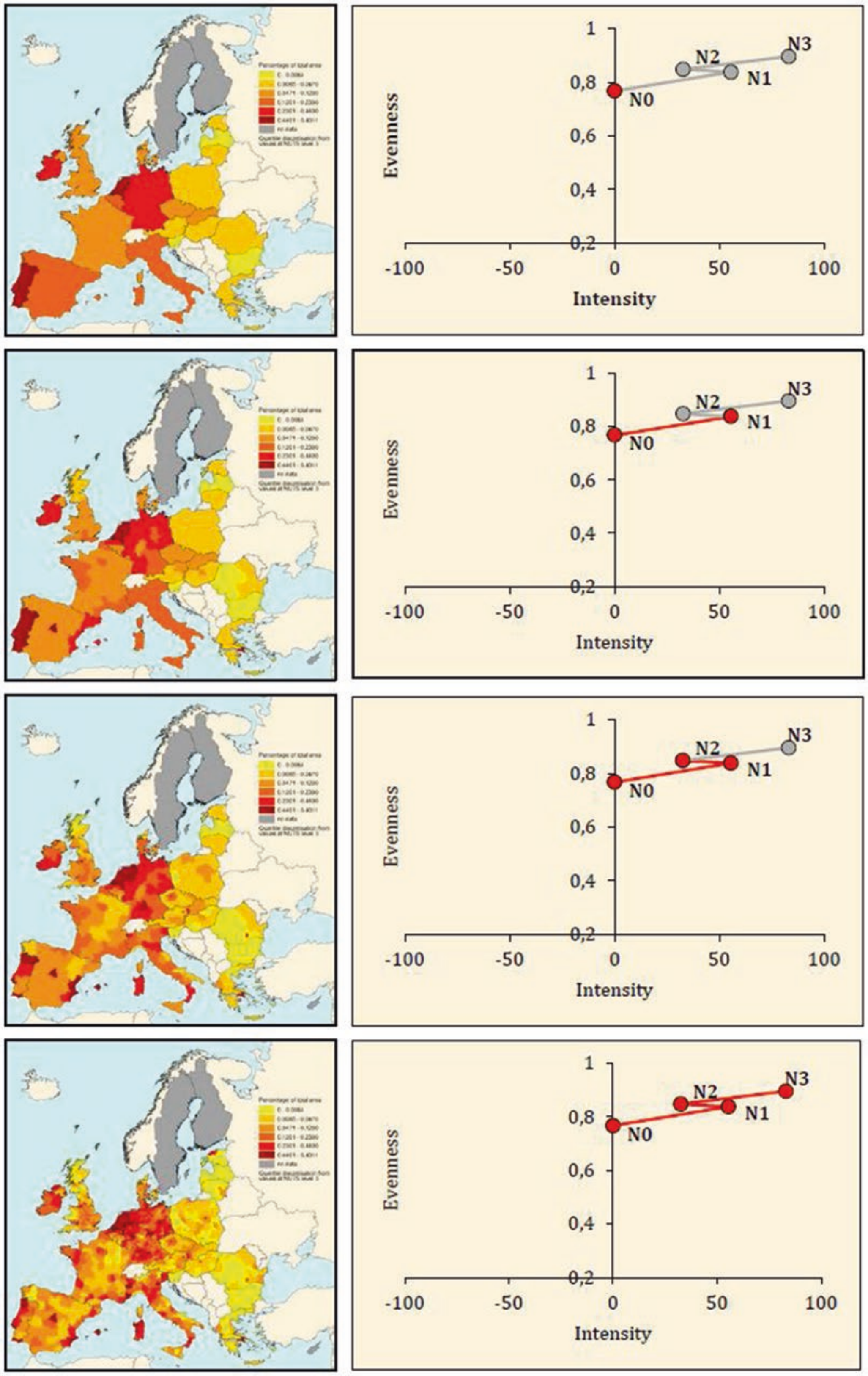

Fig. 8.3 Scale sensitivity of Urbanization in Europe between 1990 and 2000. From top to bottom: Moving from NUTS 0 to NUTS 3 showing the change from one level to the next higher level. The right panels show the chances in intensity and evenness. (*Data from* CORINE Land Cover https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

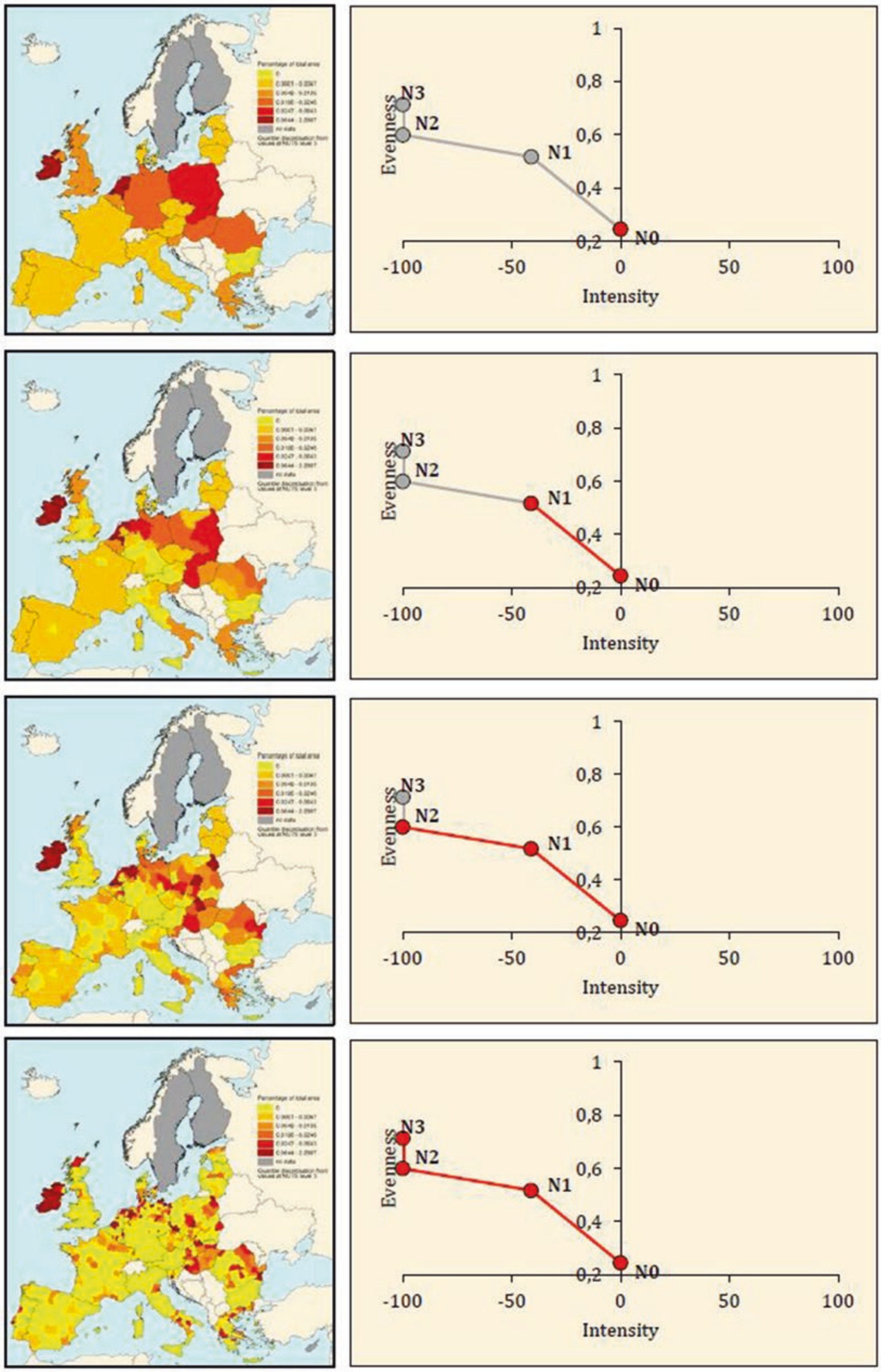

Fig. 8.4 Scale sensitivity of Wetland loss in Europe between 1990 and 2000. From top to bottom: Moving from NUTS 0 to NUTS 3 showing the change from one level to the next higher level. The right panels show the chances in intensity and evenness. (*Data from* CORINE Land Cover https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

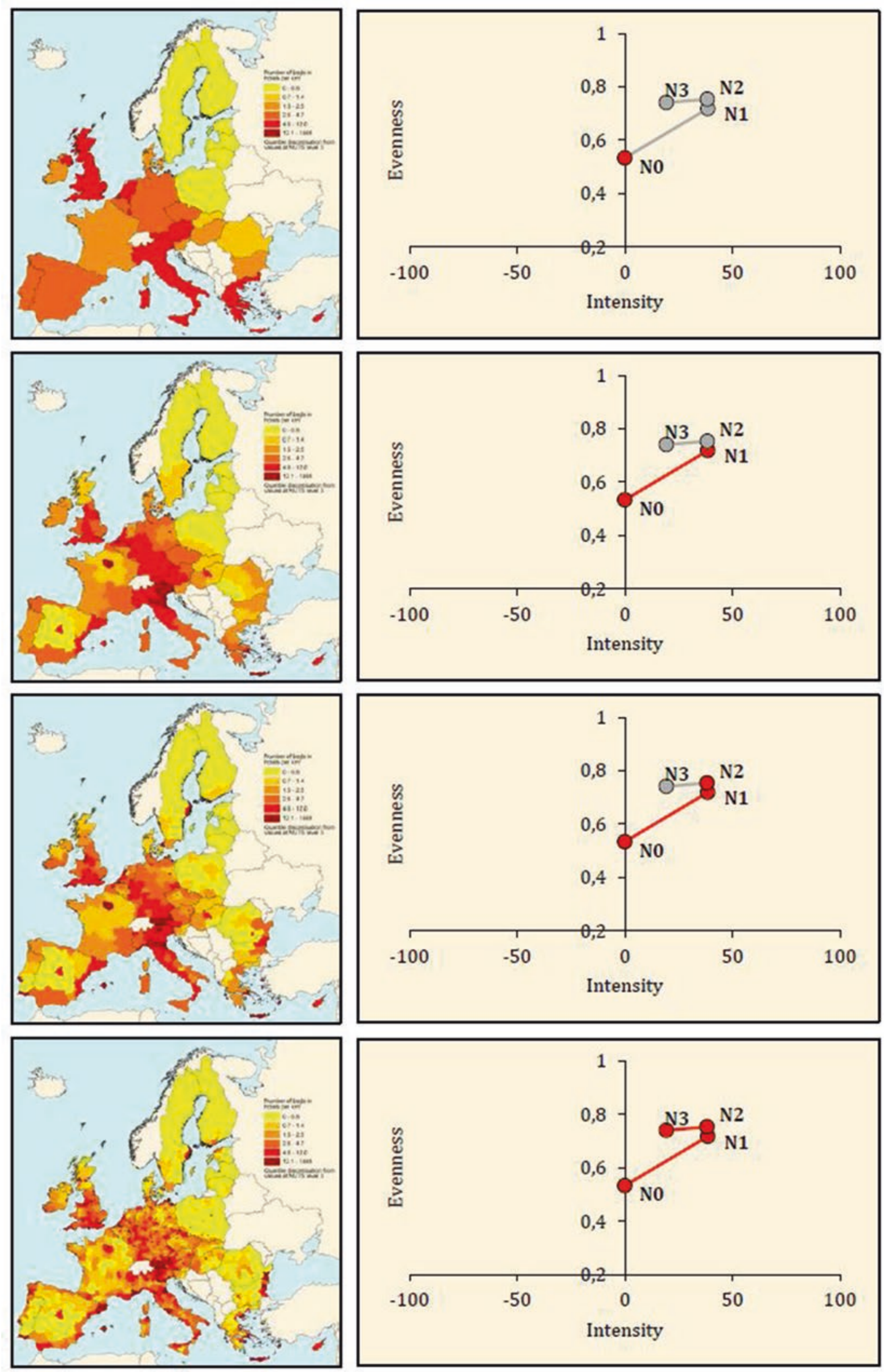

Fig. 8.5 Scale sensitivity of Tourism. From top to bottom: Moving from NUTS 0 to NUTS 3 showing the change from one level to the next higher level. The right panels show the chances in intensity and evenness. (*Data from* CORINE Land Cover 1990 and 2000 https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

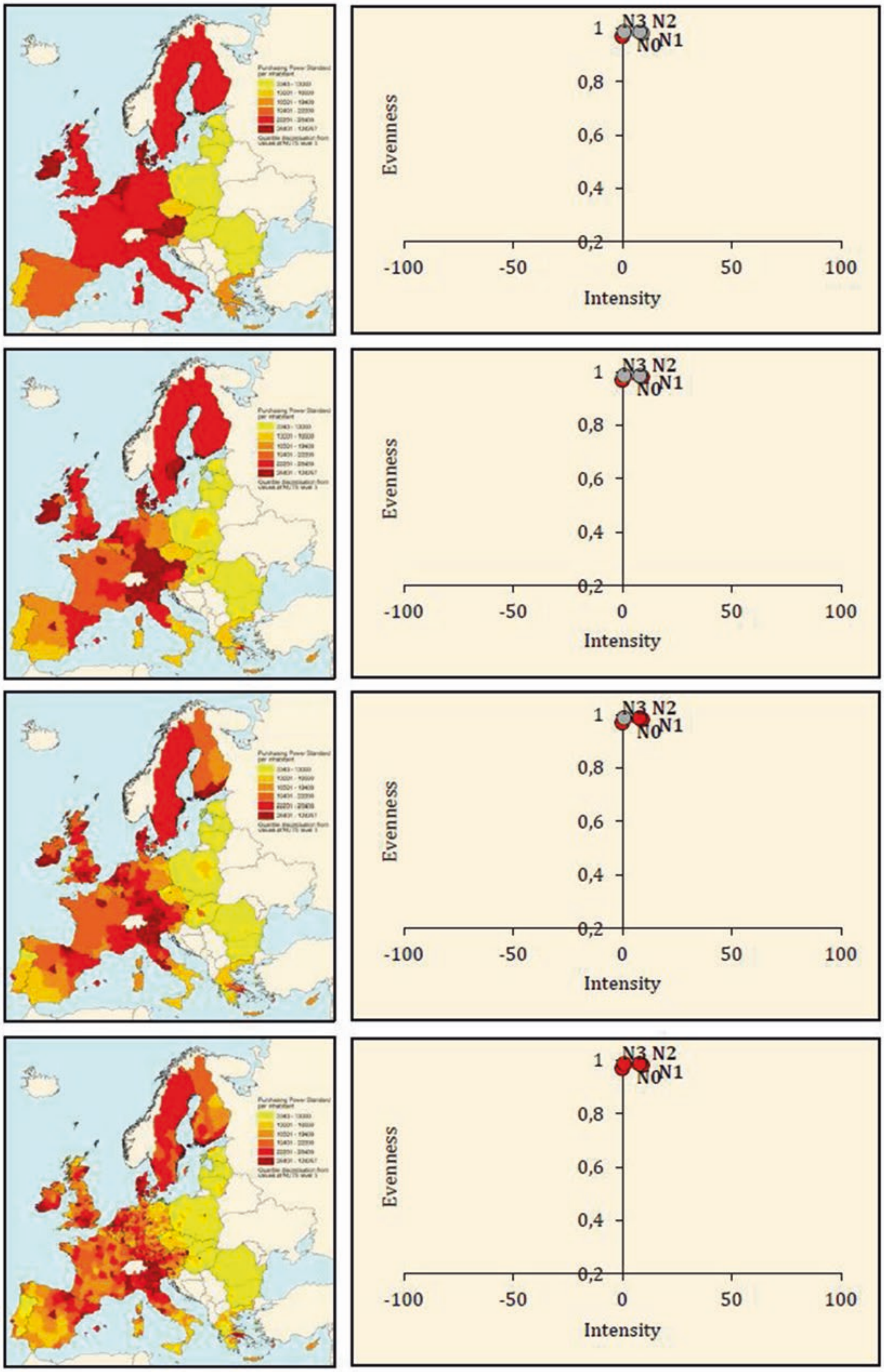

Fig. 8.6 Scale sensitivity of Gross Domestic Product (GDP). From top to bottom: Moving from NUTS 0 to NUTS 3 showing the change from one level to the next higher level. The right panels show the chances in intensity and evenness. (*Data from* CORINE Land Cover https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

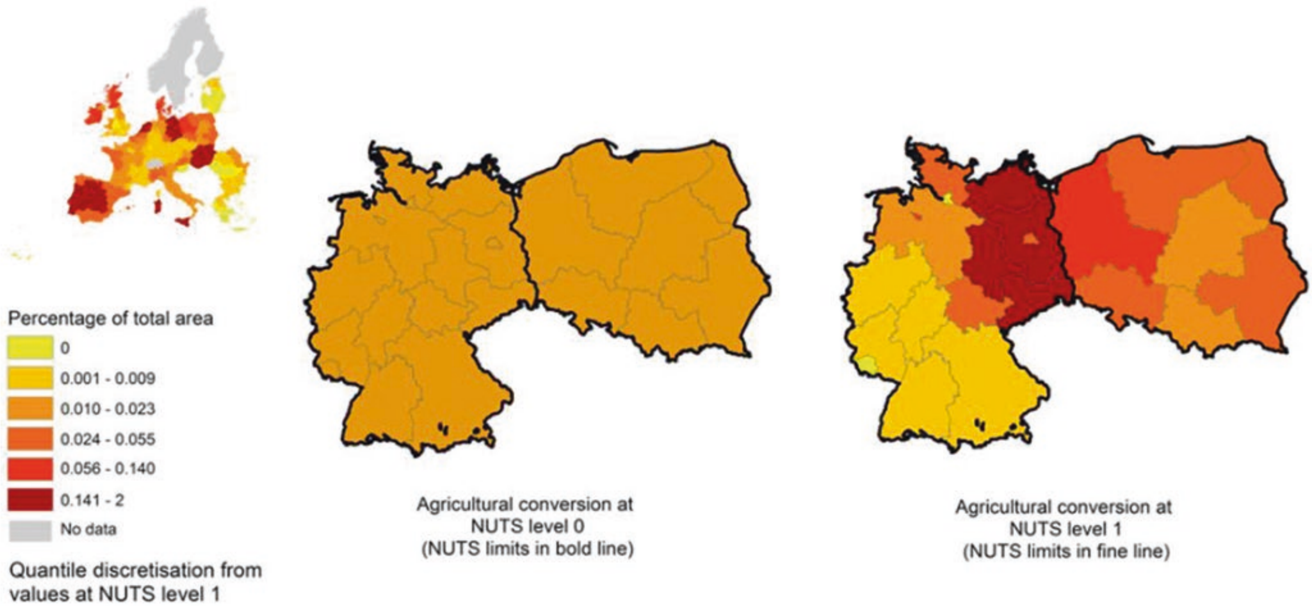

Fig. 8.7 Agricultural conversion at NUTS 0 and NUTS 1 in Germany and Poland between 1990 and 2000. (*Data from* CORINE Land Cover https://www.eea.europa.eu/data-and-maps/data/copernicus-land-monitoring-service-corine)

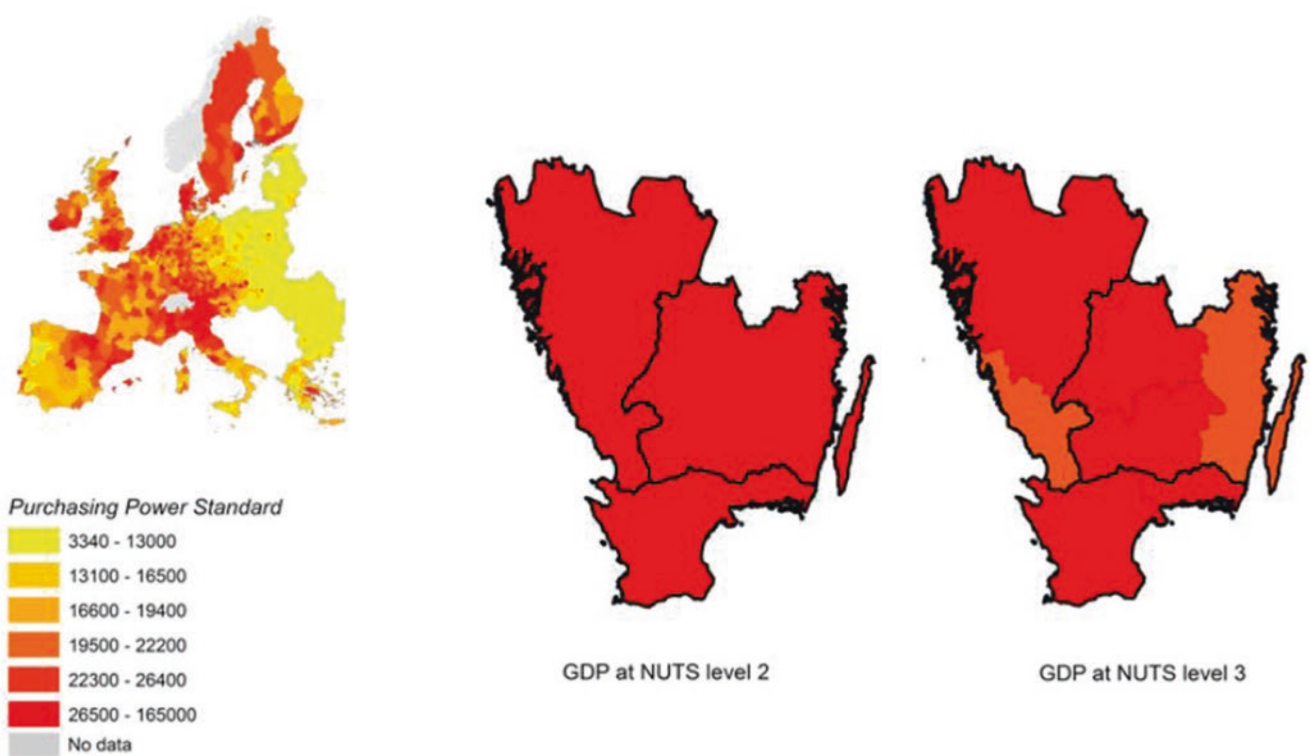

Fig. 8.8 Gross Domestic Product at NUTS 2 and NUTS 3 in Småland and South Sweden (Sweden), mean for the decade 1990–2000. Data source: Eurostat

Table 8.1 Typology of scale sensitivity

Class	Drivers	Scale sensitivity	Evenness	Change in intensity
1	Age structure, Mortality, Gross Domestic Product Employment in industry, Employment in services, Unemployment, Utilised agricultural area, Arable area	Very low	Almost no change (0)	Almost no change (0)
2	Employment in agriculture, Farm margin, Farm size, Farmers training, Livestock density, Forest area, Pasture area	Low	Slight increase (↑)	Almost no change (0)
3	Population density, Tourism infrastructure, Urbanization, Irrigation	Moderate	Moderate increase (↑)	Moderate increase (↑)
4	Permanent crop area, Afforestation, Deforestation, Agricultural conversion	High	Moderate increase (↑)	Large decrease (🡻)
5	Wetland loss	Very high	Large increase (🡹)	Large decrease (🡻)

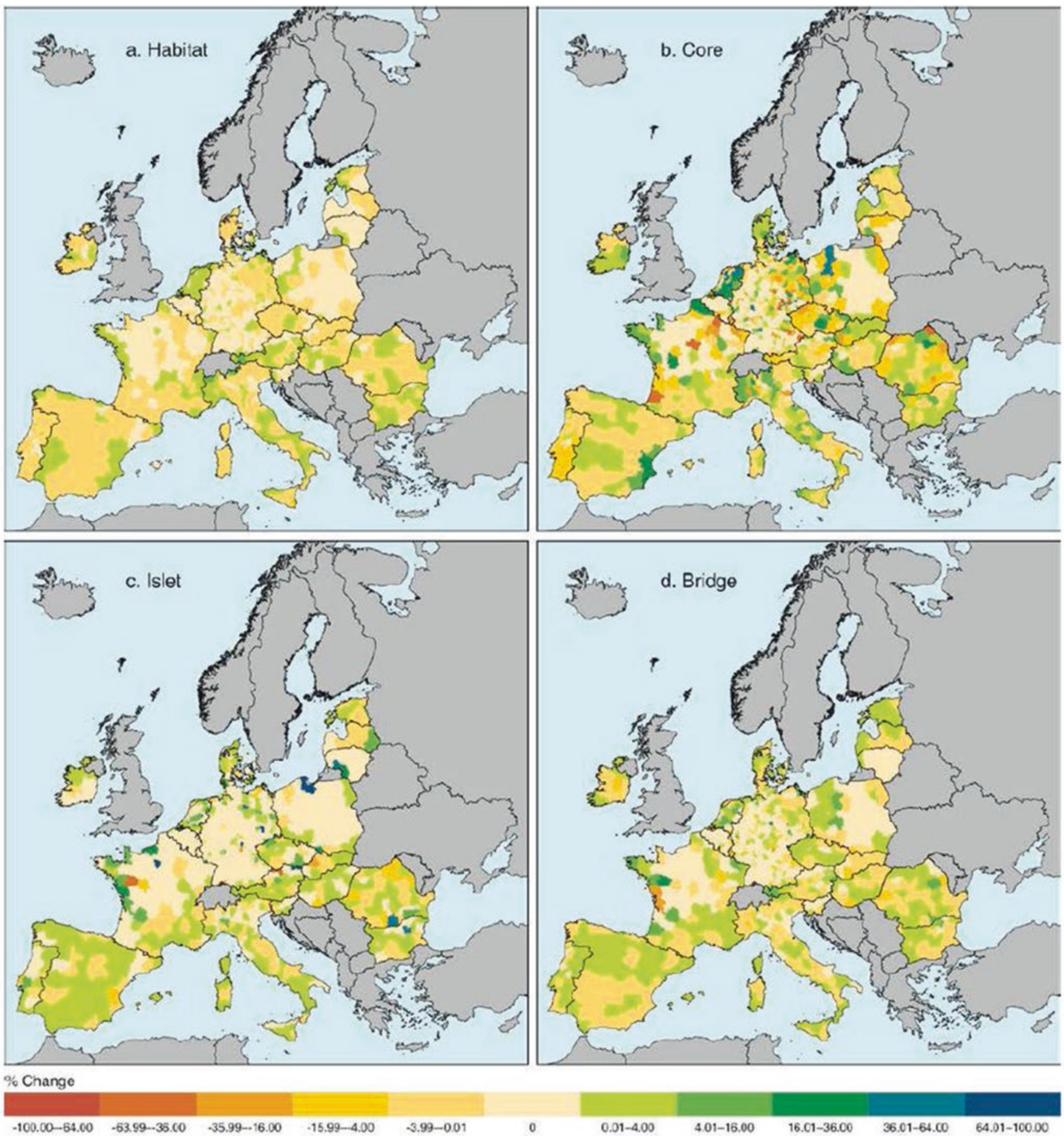

Fig. 8.9 Scale sensitivity of the assessment of grassland fragmentation based on CORINE land cover data: (**a**) changes in percentage of natural grassland at NUTS 3 between 1990 and 2006; (**b**)–(**d**) changes in percentage of natural grassland classified as 'core', islet and 'bridge', respectively, between 1990 and 2006. (*From* Scott et al. [14])

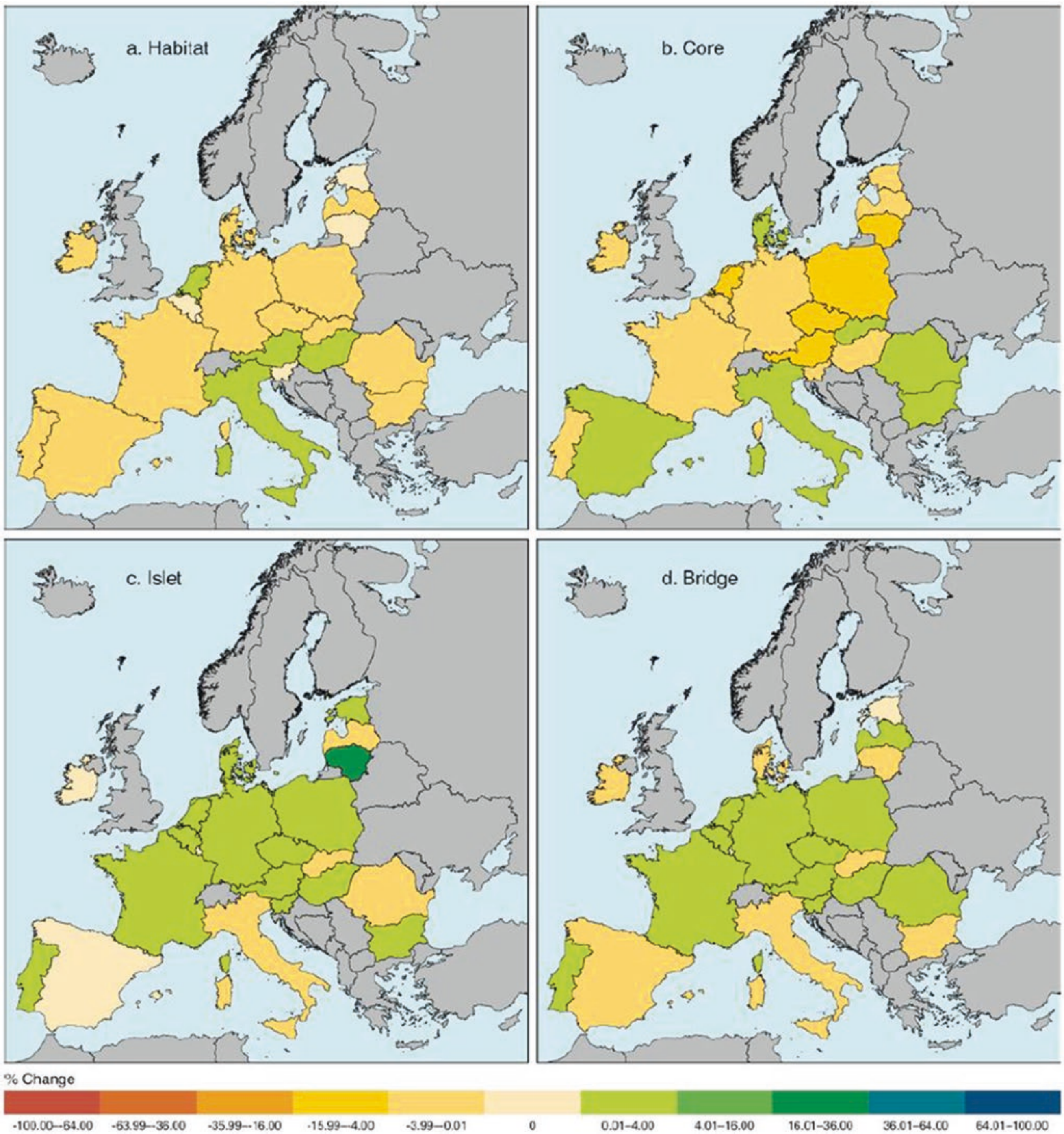

Fig. 8.10 Scale sensitivity of the assessment of grassland fragmentation based on CORINE land cover data: (**a**) changes in percentage of natural grassland at NUTS 0 between 1990 and 2006; (**b**–**d**) changes in percentage of natural grassland classified as 'core', islet and 'bridge', respectively, between 1990 and 2006. (*From* Scott et al. [14])

scale-dependent differences in changes of grassland habitat to the proportion of core, isolated (islet), and bridge habitats at two spatial scales (NUTS 0 and NUTS 3) [14].

This example shows that natural grasslands across Europe decreased by approximately 2.4% (1900 km^2) between 1990 and 2006. At the country scale (NUTS 0), these changes appear very evenly distributed, with most countries displaying changes of less than ±4%. However, at the more regional scale (NUTS 3), there is a much wider variation in the percentage changes, with some regions experiencing much higher losses or gains in core habitat and/or isolated (islet) habitat. This suggests that strategies for protecting biodiversity and securing the ecosystem services it provides cannot always be devised and implemented at a national scale without the regional effect of fragmentation and habitat loss being considered. By observing fragmentation and other landscape changes across different scales, the risks to different species can be assessed. Practitioners and policy makers are better able to understand the way in which their sites, networks, regions, and countries might be changing, and can consequently highlight areas, habitats and species, and ecosystem services provided by them, that are at greatest risk.

8.6 Conclusions

Scale sensitivity has important implications for policy making. Depending on how public bodies address environmental issues, policies may be designed at the state level or at a regional/sub-scale level. In large countries or political bodies (e.g., the European Union), it is likely that environmental and socio-economic characteristics offer important spatial variability. In that case, the evidence of scale sensitivity of drivers is a strong argument in favour of thinking and elaborating policies not only at state or country scale but also at regional/local level. For example, policies addressing direct drivers of change (such as land conversion) need to be scale sensitive and must take scale into consideration during the designing process. 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.

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9 The Evidence for Genetic Diversity Effects on Ecosystem Services

Stefan G. Michalski

9.1 Introduction

The unprecedented rates of climate and land use change associated with habitat loss and fragmentation entail major risks for biodiversity in general. A loss of within species level variability, i.e., genetic diversity, can fundamentally threaten the ability of populations and species to persist, adapt, and migrate. Apart from these direct effects, a loss of genetic diversity may also have an impact on community and ecosystem level (Fig. 9.1). Indeed, experimental and empirical evidence shows that genetic diversity may substantially drive ecosystem processes [1]. However, in contrast to the value of species diversity for sustaining ecosystem services [2] and similar underlying mechanisms, i.e., additive and non-additive mechanisms, such as the sampling effect and niche partitioning or facilitation, respectively, the role of genetic diversity still remains poorly understood. Here, based on a survey of available studies (N = 59) that experimentally manipulate plant genetic diversity or genotypic richness, the possible links between genetic diversity and ecosystems services are explored, and it is argued that a loss of genetic diversity can constitute severe risks for community-based processes.

Which ecosystem services are addressed? Supporting: Primary productivity, carbon sequestration, soil formation, nutrient cycling, pollination, Provisioning: Crop yield, Regulating: Stability and maintenance of biodiversity, erosion control, water purification, pest and disease control.

What is the research question addressed? Does genetic diversity have an impact on ecosystem services?

Which method has been applied? Literature review.

What is the main result? Genetic diversity is affecting ecosystem services, but effect sizes as compared to species diversity are largely unknown.

What is concluded, recommended? A loss of genetic diversity can constitute severe risks for community-based ecosystem services.

9.2 Supporting Services

9.2.1 Primary Productivity/Carbon Sequestration

Many studies that manipulate plant genetic diversity report on aboveground biomass production as a main function for other ecosystem processes. A positive diversity-productivity relationship for at least one of the treatments applied was found in 17 out of 27 surveyed studies (63%) that report on above-ground productivity [3, 4]. However, studies on multi-species, community level, manipulating genetic diversity of at least one component species showed either no effects [5] or ambiguous effects [6] on community productivity, raising the question of whether results for single species can be generalized for multi-species communities. However, the temporal stability of biomass production in grassland communities was found to be primarily controlled by genetic diversity [7]. Still, if the positive genetic diversity-productivity relationship holds true also for more complex ecosystems, genetic diversity should indirectly increase the system's ability to sequester carbon because processes that return more biomass and increase soil organic matter also enhance carbon sequestration [8].

9.2.2 Soil Formation/Nutrient Cycling

A number of experimental studies manipulating intraspecific litter diversity report positive effects of diversity on decomposition rates and/or soil respiration and hence nutrient

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Fig. 9.1 Genetic variation is a fundamental level of biodiversity. In wetland ecosystems the retention and degradation of contaminants, and hence water quality, is positively affected by plant genetics within and across species diversity as well as the associated microbial community. Photo by Stefan Michalski

cycling [9]. The question of whether genetic diversity of plant species impacts on soil formation and nutrient cycling by effects on diversity of soil organisms has yet to be explored in more detail. Wang et al. [10] report a higher abundance and richness of soil animals in mixtures compared to monocultures of the invasive *Solidago canadensis*; however, root fungal diversity did not respond to increased genotypic richness in a mesocosm experiment where genetic diversity of multiple plant species was manipulated simultaneously [11]. In general, the responses of soil ecosystem functions to plant genetic diversity seem to be complex and multifactorially controlled [12].

9.2.3 Pollination

It has been argued that diversity of native pollinators is essential to sustaining pollination services because of year-to-year climatic and environmental variation [13]. Genetic diversity of both crop and crop-associated plants could impact on this service if higher intraspecific diversity sustains also a greater number and diversity of pollinators. The available evidence for such a relationship has been explored by Hajjar et al. [8], emphasizing the potential benefits of genetic diversity. Experimental studies support this view by reporting an increase in (a) arthropod richness and diversity; (b) flower visitors; (c) flower display; or (d) flowering duration with increasing genotypic richness [14–18].

9.3 Provisioning Services

The cultivation of crop species in more genetically diverse varietal mixtures instead of monocultures is a long-known low-tech agricultural approach aimed at increasing and stabilizing yield and reducing pesticide application [19, 20]. A recent meta-analysis focusing on studies using wheat and barley varieties showed that mixing significantly increased grain yield, disease resistance, and weed suppression, and that this effect increased with the effective number of component varieties [21].

9.4 Regulating Services

9.4.1 Stability and Maintenance of Biological Diversity

Genotypic diversity of individual populations often relates to fitness [22] and hence can be expected to positively affect stability and maintenance of ecosystems in general. Apart from fitness, increased genotypic diversity in monospecific stands of *Zostera* seagrass species resulted in higher resistance and/or resilience to disturbances [23, 24]. Genetically more diverse populations were also found more resilient to invasion in some studies [25, 26]. Others, however, could not find an effect [27]. Perhaps more relevant for natural communities, genetic diversity within species has also been found to stabilize species diversity in experimental grasslands [5, 11].

9.4.2 Erosion Control

Plant root density and characteristics are known to have an impact on the ability of soils to withstand erosion [28]. Plant root architecture is under both environmental and genetic control and hence is very likely to mediate a diversity-erosion susceptibility relationship. The few relevant studies on this topic provide a rather positive support. For example, root biomass increased significantly in *Pseudoroegneria spicata* when pairs of genetically less-related individuals were planted together, compared to genetically more similar pairs [29]. Similarly, belowground biomass increased by 15% in nine-genotype polycultures compared to monocultures of *Andropogon gerardii* [30].

9.4.3 Water Purification

So far only little direct evidence has been reported for the effects of genetic diversity on the potential of ecosystems to maintain and enhance water quality. Using a wetland mesocosm experiment, Tomimatsu et al. [31] found that genotypic polycultures of the Common Reed can significantly reduce the concentration of inorganic nitrogen in the outflow water compared to monocultures. This effect was positively correlated with the abundance of a denitrifying gene (*nosZ*) in the bacterial soil community, suggesting a positive plant-soil microbiome feedback in response to an increased genotypic diversity. Though without a clear monotonic pattern, genotypic diversity was also found to impact on sediment oxygen availability [32], which can control water quality in aquatic systems.

9.4.4 Pest and Disease Control

In principle, genetic diversity within host populations can either increase or decrease pathogen transmission and disease risk [33]. Though not yet widely employed in modern agricultural ecosystems, genotypically diverse plantings, i.e., cultivar mixtures, have been repeatedly used for successful disease management [34]. Experimental evidence for effects of diversity on plant health and pest resilience from non-crop plant species is more equivocal and mostly based on observations on herbivory. A number of studies report no effect or an increase in herbivore richness and damage with increasing diversity [35, 36], but contrasting results have also been published [37, 38]. Various feeding guilds of herbivores have been found to respond differently to genotypic diversity of the same species [39], emphasizing the need for more detailed investigations on the mechanism behind diversity effects on plant-herbivore interactions and pest control in general.

9.5 Effect Sizes

Very similar to species diversity, mechanisms driving genetic diversity – ecosystem functioning relationships – are based on genetically based trait variation. Consequently, for a study investigating the effect of genetic diversity in *Phragmites australis* on primary productivity and water quality in a constructed wetland ecosystem [31], the observed effects showed the same direction as three very similar studies in which species diversity was manipulated instead [40–42], but effect sizes were smaller (Fig. 9.2).

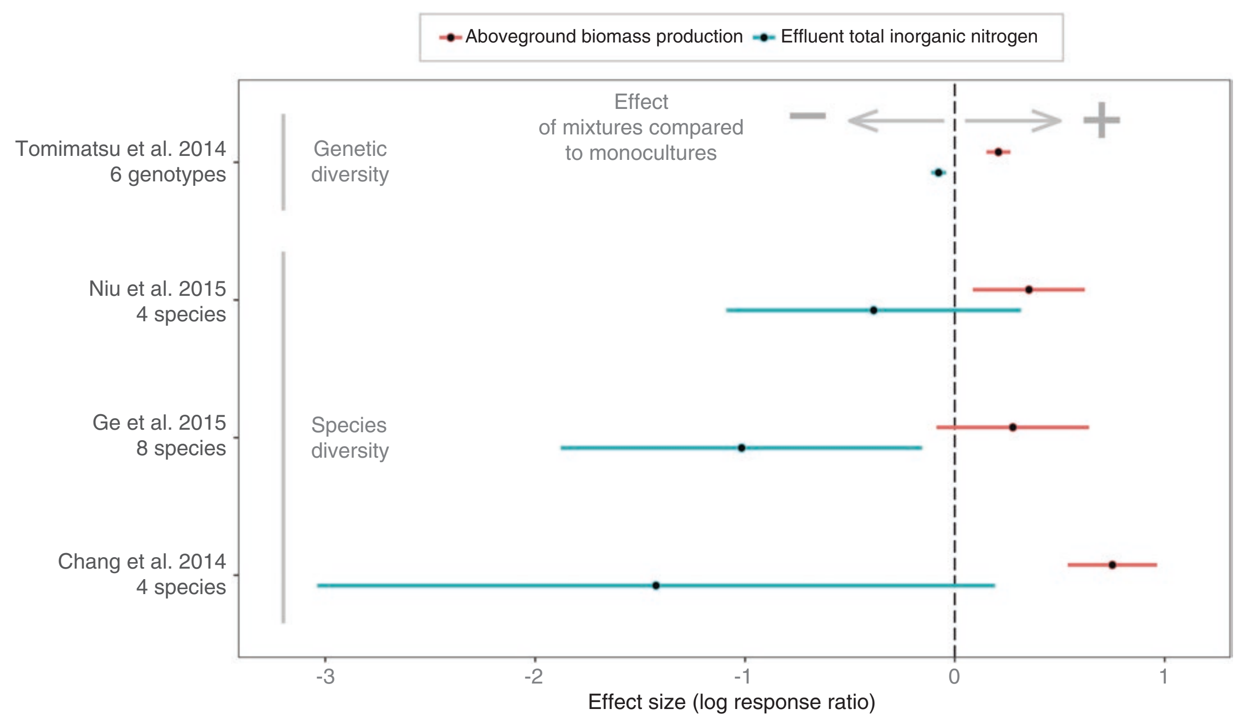

Fig. 9.2 Effects of increased species and genotypic diversity on plant productivity (aboveground biomass) and water quality (total inorganic nitrogen concentration in effluent water) in experimental wetland ecosystems. Displayed are effect sizes and respective standard errors. For each study, effect sizes are expressed as log response ratio between the mean performance for the mixtures with the largest number of species or genotypes and the mean for respective monocultures

Across a species range this trait variation can be substantial and even exceed among-species variation [43]. Indeed, in some cases the effects of genetic diversity on ecosystem processes have been described as comparable in magnitude to the effects of species diversity [1, 44]. At the local community scale, intraspecific trait variation will generally be lower than interspecific variation, suggesting the strength of an effect in natural ecosystems might on average be lower for genetic diversity compared to species diversity [45]. For artificially assembled communities to be used, for example, in ecological restoration or for remediation measures, the scale-dependency is less relevant as genetic diversity can be maximized, which also would ensure the potential for long-term adaptive evolutionary changes. In summary, the evidence for significant effects of genetic diversity on community and ecosystem level processes, and hence for providing a large variety of ecosystem services, has accumulated in recent years; the relative magnitude of these effects, however, remains largely unknown. It can be hypothesized that the risks for ecosystem services associated with a loss of genetic diversity will be relatively higher in species-poor ecosystems like wetlands or agroecosystems, which, however, makes them suitable model systems for future research in this field (Fig. 9.3).

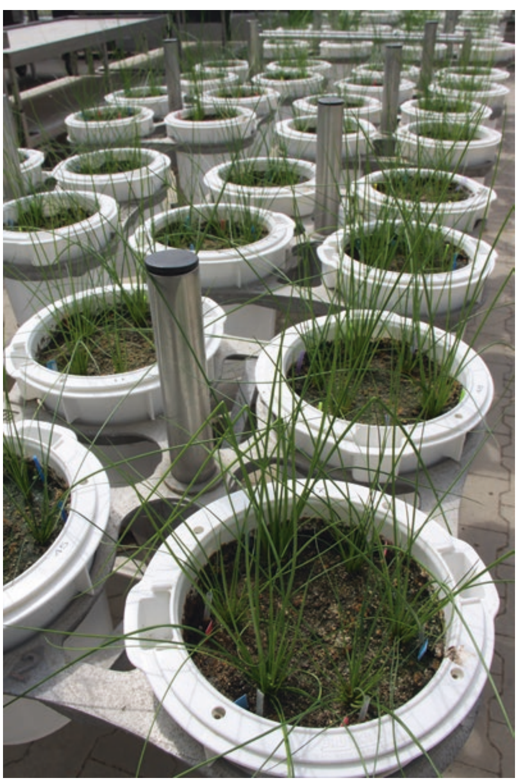

Fig. 9.3 Ongoing experiment investigating the impact of genotypic richness of the wetland plant *Juncus effusus* on ecosystem processes (Stefan G. Michalski 2017, unpubl.)

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Using Dynamic Global Vegetation Models (DGVMs) for Projecting Ecosystem Services at Regional Scales

10

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10.1 A Case Study from Latin America: Projecting Vegetation Change Driven by Climate and Land-Use Change with a Regionalized DGVM

Which ecosystem services are addressed? Regulating and provisioning.

Which method has been applied? Dynamic Global Vegetation Model used to simulate changes in ecological variables that are important for provisioning of ecosystem services.

What is the major result? Climate and land-use change cause biome shifts that affect provisioning of regulating and provisioning services. Important to quantify changes in ecosystem-level consequences of these drivers.

What is concluded, recommended? Use of simulated changes in ecological variables to compile impacts on ecosystem services. These should be used further, together with other methods, in valuation schemes and in policy recommendations.

Latin America is experiencing land-use change, often for beef and crop production, and this is likely to continue in the future. Climate change might have severe impacts on Latin American biomes. These anthropogenic changes put the provisioning of multiple ecosystem services at risk. We therefore made use of regionally specific socio-economic development scenarios for Latin America [1]. Dynamic Global Vegetation Models (DGVMs) offer the possibility to integrate large amounts of geospatial data to quantify and project a large range of ecological variables that describe changes in vegetation cover (biome shifts) and that are important for ecosystem service provisioning under future scenarios. We combined the land-use change (LUC) of a best-case scenario (i.e., low rates of LUC) and a worst-case scenario (i.e., high rates of LUC) to demonstrate the climate forcing of both the least and the most severe scenarios of climate change (i.e., representative concentration pathways (RCP) 2.6 and 8.5 [2] using the DGVM LPJmL [3, 4]). Figure 10.1 shows the spatial distribution of projected vegetation change in 2099 for these two integrated scenarios. The best-case scenario describes low-intensity climate change *(RCP 2.6) under low rates of LUC* (i.e., Shared Socio-Economic Pathways SSP1) while the worst-case scenario quantifies severe climate change *(RCP 8.5) and high rates of LUC (SSP5)*. The land-use change scenarios were simulated using an agent-based model which translates population growth, socio-economic development, and environmental policies into spatio-temporal patterns of land-use change. For illustration, the climate forcing of the Global Circulation Model (GCM) Hadley Centre Global Environment Model version 2 (HadGEM2- ES) was selected due to its suitable model performance for South America [5]. In the best-case scenario (Fig. 10.1a), projected land cover in 2099 indicates few changes in vegetation cover that are attributed to climate change during this century (Fig. 10.1b). In the worst-case scenario, climate-driven changes in vegetation cover become apparent (Fig. 10.1d, e) predominantly during the second half of this century (Fig. 10.1f). In both scenarios, forest transformation from closed to open forest and further to shrubland represents the greatest proportion among the climate-driven biome shifts, whereas forest transformation to cropland or pastoral land represents the greatest proportion among the land-use-driven biome shifts in Latin America (Fig. 10.2). Areas where the natural vegetation is vulnerable to future climate change may be transformed by land-use change well before they undergo climate-driven biome shifts, so that the land-use effect on biome state masks the potential climate impact.

The implications of such projected vegetation and land cover changes for ecosystem service provisioning are not immediately clear. We will outline next how DGVMs can serve in ecosystem service assessments.

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10.2 Projecting Changes in Ecosystem Services Using DGVMs

10.2.1 Ecosystem Services Bundles and Cascades in DGVMs

Ecosystem services do not exist in isolation from one another—humans benefit from multiple services at the same time, and services within such an ecosystem service bundle often trade off with one another. For instance, transforming a forest into pasture may provide space necessary for food production, but gaining this service trades off with timber production, non-timber forest products, carbon storage, climate regulation, and the cultural value of largely intact forest systems. Hence, such an ecosystem service cascade [6] often links the two ends of an ecological "production chain" [7, 8]. Because the structure and productivity of vegetation systems forms the basis of many ecosystem services, spatially explicit vegetation models that link plant functional traits to ecosystem functioning with derived services are a promising approach to address such complex social-ecological questions [9].

With DGVMs we are able to derive the supply for provisioning and regulating ecosystem services from model output variables. Such process-based models are powerful tools for projecting how multiple drivers will affect ecosystem service supply. As DGVMs were originally designed to investigate biogeochemical cycling at large spatiotemporal scales [10], the challenge is to provide projections at landscape-to-regional scale to be more relevant to ecosystem service management. DGVMs become increasingly refined to incorporate aspects of nutrient limitation and biodiversity change at smaller spatial scales [11, 12]. An even more varied portfolio of ecosystem services can be approximated from them (Table 10.1). Some of the variables in Table 10.1 were already used to quantify the vegetation and land cover change shown in Figs. 10.1 and 10.2 (e.g., biomass, vegetation structure, and evapotranspiration), while other model variables are still waiting to be employed for quantifying ecosystem service provisioning (e.g., plant biodiversity).

DGVMs that simulate patterns and processes of natural and managed ecosystems provide data relevant to the provisioning of those ecosystem services related to carbon and water cycles, and agricultural production. These data are increasingly related to biodiversity, thereby covering continuous temporal and spatial coverage. By applying scenarios of climate and land-use change, impacts of socio-economic and biophysical drivers can be analyzed.

A next step in model development would be to incorporate feedbacks between human demand and natural resources [13]. A fully coupled DGVM—one that resolves plant biodiversity, interactions between anthropogenic drivers of land cover change, and feedbacks between biogeochemical processes on a landscape-scale spatial grid—would computationally be very expensive. Implementing human decisions in Integrated Assessment Models following the example of agent-based functional types [14] is an important step forward, which could be extended to decision-making based on the sustainable use of ecosystem services to produce respective land-use change scenarios. As an intermediate step, DGVMs can already be used to analyze trade-offs and synergies arising from impacts of climate and land-use change on the provision of ecosystem services [15]. Models such as DGVMs can be further combined with national statistics, field estimations, and remote sensing data to quantify the supply, delivery, and value of ecosystem services [16–18]. Decision-support tools and payment schemes might have to be adapted to combine such new data sources [19, 20].

10.3 Outlook

The way forward in quantifying, projecting, and evaluating ecosystem services with DGVMs would be to

- **Quantify ecosystem service bundles:** In DGVM projections, not only provisioning services should be quantified, but extended to include previously less well-defined regulating services and their influence on provisioning services. Cultural services have to be defined in their own right as DGVMs are not a suitable tool to model them directly.

10.4 Conclusion

Quantifying and projecting ecosystem service bundles requires a multi-model approach of spatially explicit models, which links biodiversity to ecosystem function with the benefits that social-ecological systems provide in a multi-functional context. Using examples of modelling studies with DGVMs in regions where ecosystem service provisioning is threatened by global change, we outlined a strategy for how DGVM projections could become more useful in quantifying ecosystem service bundles. We hope our contribution encourages ecologists to engage in interdisciplinary and transdisciplinary work together with economists and social scientists to establish integrated ecosystem service analyses that inform policy-makers and ecosystem management practitioners in the face of global change.

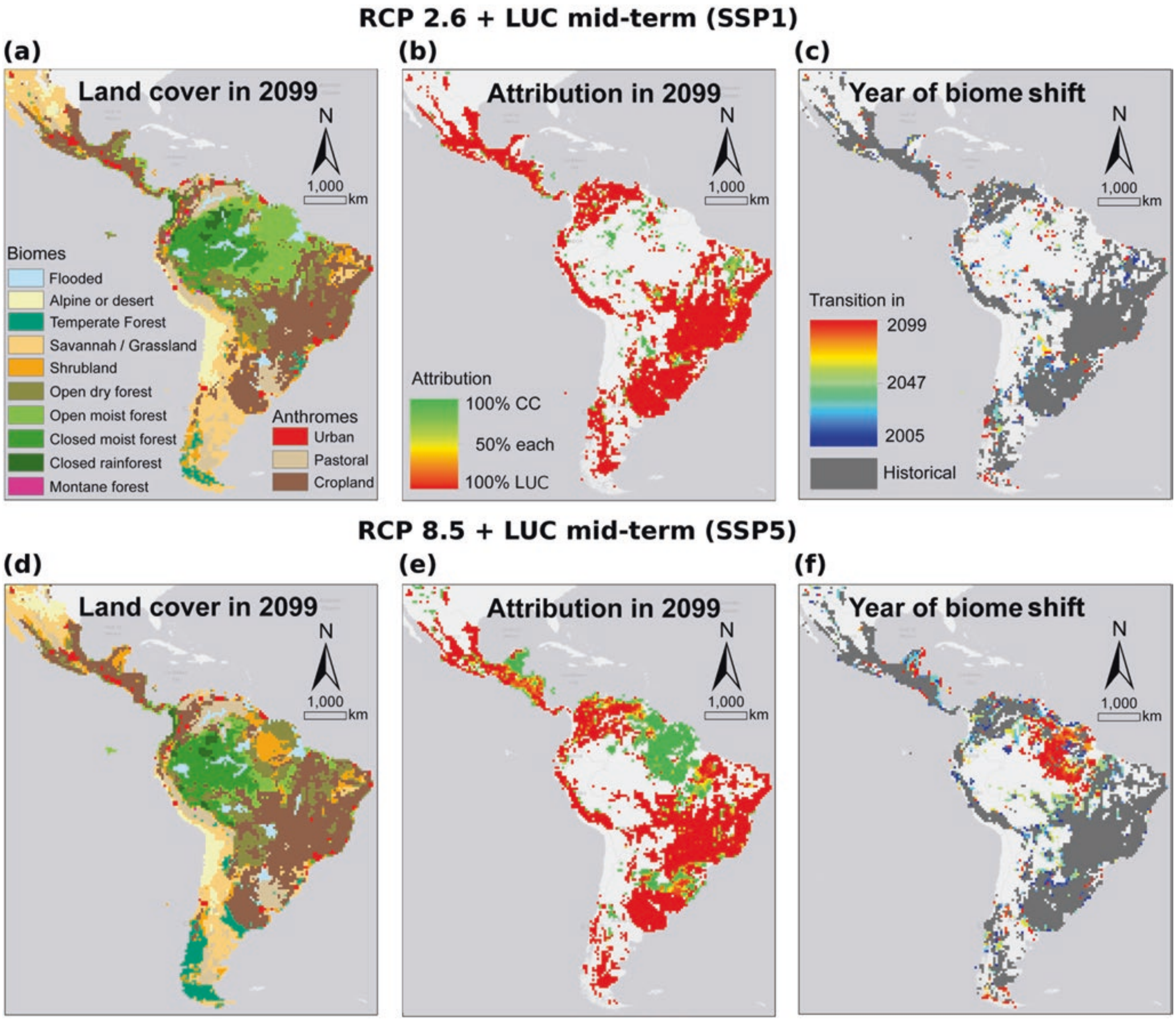

Fig. 10.1 Projected land cover, attribution of vegetation cover change to climate change (CC) and land-use change (LUC), and the year of biome shifts in Latin America. Maps for the best-case scenario (A–C) and the worst-case scenario (D–F) under climate change. (**a**, **d**) Land cover types divided into mostly natural biomes and strongly human-modified landscapes (so-called anthromes). (**b**, **e**) Attribution of vegetation cover to climate change and land-use change, respectively. The relative contributions add up to 100% in each affected grid cell (**c**, **f**). Year of detection of the shift in land cover class ("biome shift") leading to the final vegetation state shown in **a** and **d**. Dark grey areas depict vegetation cover change caused by historical land-use change until 2005. (*From* Boit et al. [1])

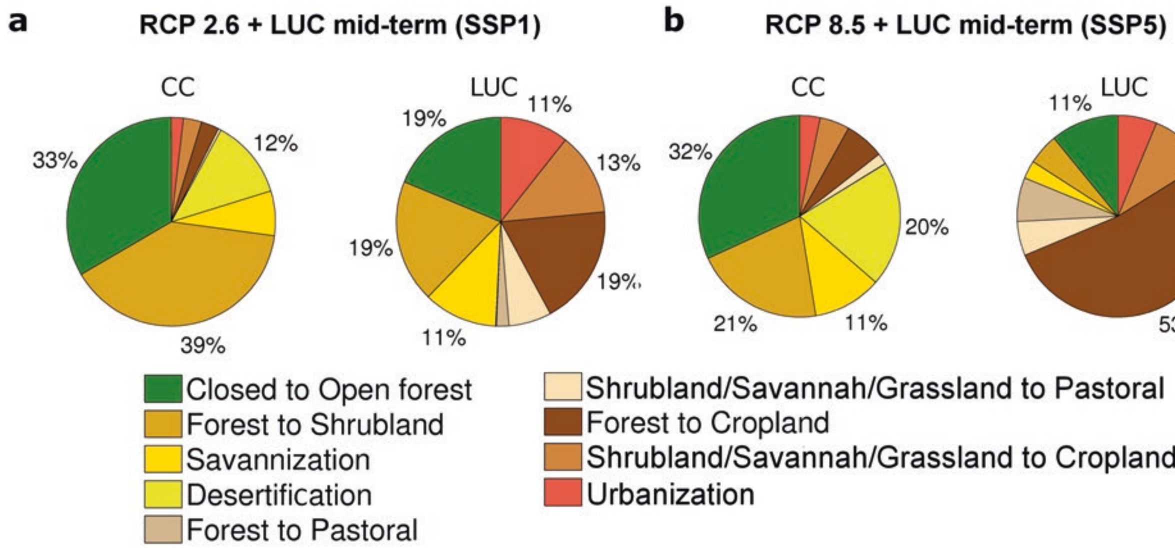

Fig. 10.2 Projected changes between natural and anthropogenic transformed land cover types from 2005 to 2099. Each diagram in **a** and **b** shows the relative proportions of changes between land cover types attributed either to climate change (CC) or land-use change (LUC), respectively, in the best-case (**a**) and worst-case (**b**) scenario. (*From* Boit et al. [1])

Table 10.1 Ecosystem services available from DGVM output variables

Ecosystem service type	Name	DGVM variable useable as proxy for ecosystem service supply	Modelled with DGVM
Regulating	Climate regulation (Carbon stocks and uptake)	Total biomass	Southeast Asia [15] Africa [22]; Global [17, 22, 23]
		Carbon sequestration (net ecosystem balance = GPP – emissions [1])	
	Climate regulation	Evapotranspiration [2]	Global [24, 25]
Provisioning	Water	Ground water supply, infiltration, green and blue water, discharge	Global [17, 26] Southeast Asia [15]
	Wood	Woody biomass	Global [17],
	Crop	Crop harvest	Global [17, 27]; Southeast Asia [15]
	Fodder	Biomass from managed grasslands	Global [28]
	Biofuel	Biomass from bioenergy tree and grass types	Global [29]
Regulating and cultural	Biodiversity	Species or trait distributions, ecosystem characteristics and diversity	Amazon rainforests [12]; East Asia [30]

Examples of ecosystem services which can be derived from DGVM output variables as indicators of service supply. Naming of ecosystem services following [18]; GPP- gross primary production [1], emissions include autotrophic and heterotrophic respiration from dead organic matter, fire-related emissions and crop harvest [2]. Evapotranspiration has been modelled using the simpler vegetation schemes of land surface models coupled to Earth System Models, but the more complex DGVMs are increasingly used to perform this task.

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11 Remote Sensing Measurements of Forest Structure Types for Ecosystem Service Mapping

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11.1 Introduction

Twenty-five percent of Earth's land surface is covered by forests, and they are habitat for more than 70% of all terrestrial species [1–3]. Forests represent an important pool in the global carbon cycle as they bind huge amounts of carbon in their living biomass [4–6]. They are also able to regulate the water cycle through processes of evapotranspiration, which is important for stabilizing the global climate [7]. Additionally, forest management is an important economic sector in Europe.

Global forests are characterized by complex patterns and structures. Forest dynamics are driven by processes that act on different spatial and temporal scales. Consequently, biomass stocks and carbon fluxes are variable in space and time. Therefore, estimating forest properties such as biomass or productivity for larger regions is a major challenge. The IPCC (Intergovernmental Panel on Climate Change) reported that missing knowledge on biomass distribution is one large source for uncertainty in the global carbon cycle [6, 8].

Forest canopy height, derived from active remote sensing systems such as lidar or radar, is often used as predictor for forest biomass. However, a significant amount of variance remains unexplained. One approach to improving height-to-biomass relationships is to consider horizontal and vertical forest structure, as forest structure is a key element for forest properties. Ground-based tests have shown that classifying stands according to structure indices, e.g., the stand density index [9] and modified species profile index [10], can lead to more accurate height-to-biomass relationships within each structure class. Hence, an important goal is to classify forest stands into structure types (horizontal and vertical structure) based on remote sensing measurements. For each forest structure type, biomass and productivity of forests could then be estimated more accurately compared to a general estimation (see Fig. 11.1).

Which ecosystem services are addressed? Ecosystem services provided by forests: forest biomass and forest productivity. Knowledge about these ecosystem services can be used to deliver further forest ecosystem services related to climate regulation, soil protection, biodiversity protection, water regulation, disturbance regulation, and bioenergy

What is the research question addressed? How can we estimate forest structure from remote sensing, and what is the role of forest structure for forest biomass and productivity estimations?

Which method has been applied? Linking forest inventory data, forest model simulations, and remote sensing

What is the main result? Forest structure can be estimated from remote sensing by using structural indices. Additionally, we show that over a broad range of forest stands, forest structure is the important driver for estimating forest biomass and forest productivity

What is concluded, recommended? Future remote sensing missions will provide information on forest structure with a high degree of detail. This will lead to more accurate estimations of forest biomass and productivity

11.2 Methods

11.2.1 Study Site

For this study we used ground-inventory data from the new forest megaplot Traunstein (https://forestgeo.si.edu/sites/europe/traunstein). It is a large permanent research plot, established as a new super test site (25 ha, 30,000 measured trees) in a highly diverse structured forest district in the German alpine upland—including even- and uneven-aged forest stands and 26 tree species. In 2016, a full tree survey was performed (stem diameter and posi-

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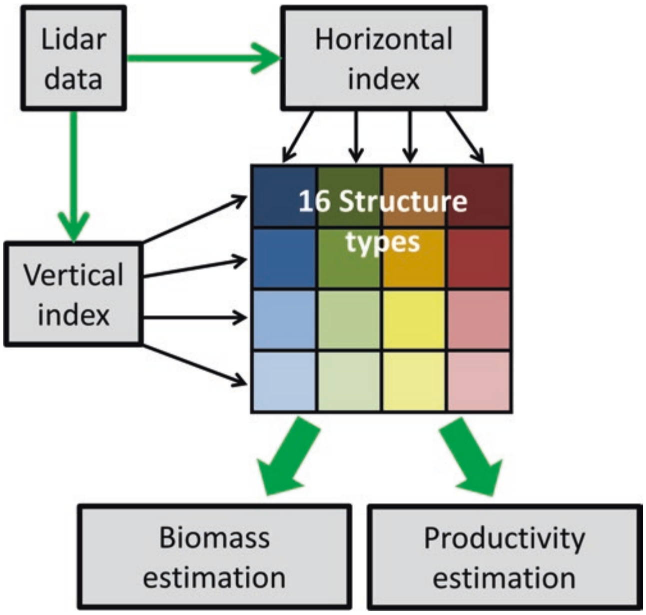

Fig. 11.1 Workflow to classify forest stands into 16 structure types and predict forest attributes (i.e., biomass and productivity)

tion for each tree) and an airborne lidar campaign was conducted by the German Aerospace Center (DLR). Since 2017 this unique research plot is part of the global Smithsonian Tropical Forest Institute Network (ForestGEO).

To apply our local findings from Traunstein to Germany, we used the German national forest inventory data ('BWI'; [11]), which consists of 48,562 field plots distributed across Germany.

11.2.2 Describing Forest Structure by Structural Indices

Different structural indices can be used to characterize the horizontal and vertical structure of forests. For each forest stand we calculated a horizontal and vertical structure index and related this stand to a forest structure type (Fig. 11.1). Horizontal structure can by described, for example, by stand basal area BA [m^2], which is the sum of all tree basal area values:

$$BA = \sum_{trees} \frac{\pi}{4} d^2,$$

where d [m] is the stem diameter of a tree.

Vertical structure can be quantified by tree height heterogeneity σ_{height} [m]:

$$\sigma_{height} = \sqrt{\frac{1}{n-1} \sum_{trees} \left(h - \bar{h}\right)^2},$$

where h [m] is the height of a tree and $\bar{h}$ [m] the mean tree height of a stand.

We focus in this study on these two indices (basal area and tree height heterogeneity), but other indices can also be applied. Another possible index for horizontal structure is, for example, stand density index SDI [−]:

$$\mathrm{SDI} = \mathrm{N} \cdot \left(\frac{25}{\bar{\mathrm{d}}}\right)^{-1.605},$$

where N [1/ha] is the number of trees and $\bar{d}$ [cm] the quadratic mean stem diameter (i.e., square root of the mean of the squares) of all trees of a stand. For the vertical structure, the modified species profile index S [−] can also be used, describing the basal area distribution in different height layers:

$$S = -\frac{1}{\ln 3} \sum_{1}^{3} p_i \bullet \ln p_i, \quad p_i = \frac{BA_i}{BA_{tot}},$$

where BA_i [m^2/ha] is the basal area in height layer i and BA_{tot} [m^2/ha] is the total basal area of a stand. Three height layers were used, which were equally spaced between the ground and the maximum height.

11.2.3 Estimating Structural Indices from Lidar Remote Sensing

To find relationships between field-based forest structure and lidar, we used ground-inventory data from the Traunstein megaplot (see Sect. 11.2.1). Additionally, we analysed the airborne lidar campaign conducted for this forest (see Sect. 11.2.1). We explored relationships between field-based forest structure (here, basal area and tree height heterogeneity) and lidar on different scales (e.g., 20 m, 100 m). Depending on the spatial scale, we found good relationships between field-based and lidar-based structure index for horizontal forest structure (e.g., $r^2 = 0.77$ for field basal area vs. lidar top-of-canopy height at the scale of 20 m). Relations for the vertical index are more challenging (e.g., $r^2 = 0.41$ for tree height heterogeneity vs. lidar 90% height quantile at the scale of 20 m).

11.2.4 Classifying Forest Stands into Structure Types

Basal area and tree height heterogeneity can be used to classify forest stands into different structure types. For this we divided both indices into four classes: basal area (m^2/ha) as horizontal structural descriptor: H1: 0–15, H2: 15–25, H3: 25–35, H4: >35, and tree height heterogeneity (m) as vertical structural descriptor: V1: 0–1, V2: 1–2, V3: 2–3, V4: >3. With this classification scheme we assign each forest stand to a vertical and horizontal structure class. Both indices can be estimated from lidar due to the derived relationships between field-based metrics and lidar.

This classification scheme was applied to the German national forest inventory (BWI) data [11]. As no wall-to-wall lidar data for Germany was available, we generated lidar data for the BWI plots using a lidar simulation model [12]. At the end we developed a Germany-wide forest structure map estimated from lidar remote sensing. The same type of maps can be derived also from radar (e.g., L-Band) as radar measurements can be also used to quantify forest structure. Similar maps can be generated using other structural indices (like SDI, not shown).

11.2.5 Forest Biomass and Productivity

In a second step, 300,000 virtual forest stands were analysed to identify the importance of forest structure for biomass and productivity estimations ("forest factory approach," [13]). The virtual stands were generated with the individual-based forest model FORMIND [14]. We calculate for each forest stand (400 m^2) biomass and productivity (here, aboveground woody productivity AWP) and relate them to the structural indices: here, basal area *BA* and standard deviation of tree heights σ_{height}.

11.3 Results and Discussion

11.3.1 Classifying Forest Stands into Structural Classes Using Field Data and Lidar

A first application of the workflow shows how forest structure types can be derived from remote sensing (here, lidar). The maps for vertical and horizontal structure types in Germany estimated from lidar are shown in Fig. 11.2, the frequency distribution of the structure type classes in Fig. 11.3.

According to this analysis, most forest stands in Germany have a high basal area >35 m^2/ha (see Fig. 11.3). The amount of forest area with heterogeneous vertical structure (σ_{height} > 2 m) is as high as the forest area with homogenous vertical structure (σ_{height} < 2 m).

11.3.2 The Importance of Forest Structure for Biomass and Productivity Estimates

Analysing all 300,000 virtual forest stands (using FORMIND), we find that forest productivity (AWP) is hardly affected by species diversity. Instead, forest structure emerges as the key variable [12]. Here, we group the forest stands into sixteen forest structure classes, four horizontal and four vertical structure classes like in the presented forest structure maps (Fig. 11.2). We find an increase in biomass with basal area (basal area as proxy for horizontal structure), whereby biomass in forests with low basal area is much more influenced by vertical heterogeneity than forests with large basal area (Fig. 11.4a).

Forest productivity increases with basal area for stands with a low vertical heterogeneity (Fig. 11.4b). However, with increasing vertical heterogeneity, the positive effect of basal area on productivity diminishes. The reason is that large trees shade smaller trees, which reduces the productivity of smaller trees. To sum up, for forest state estimations (like biomass) the horizontal forest structure plays a key role (here, basal area; Fig. 11.4). For productivity estimations, however, the horizontal and vertical structures are relevant.

Using the derived forest structure maps for Germany, we can estimate forest biomass and forest productivity distributions for forests and explore relationships between forest structure and other forest properties. We show that over a broad range of forest stands, forest structures are the important drivers for estimating forest biomass and forest productivity. Knowledge about ecosystem functions like biomass and productivity is crucial for evaluating timber volume for forestry, estimating disturbance state of forests, and understanding the effects of climate change. The concept of forest structure types in combination with forest modelling and remote sensing has high potential for applications at larger scales. Future remote sensing missions (like BIOMASS, GEDI, Tandem-L) will provide information on forest structure with a high degree of detail.

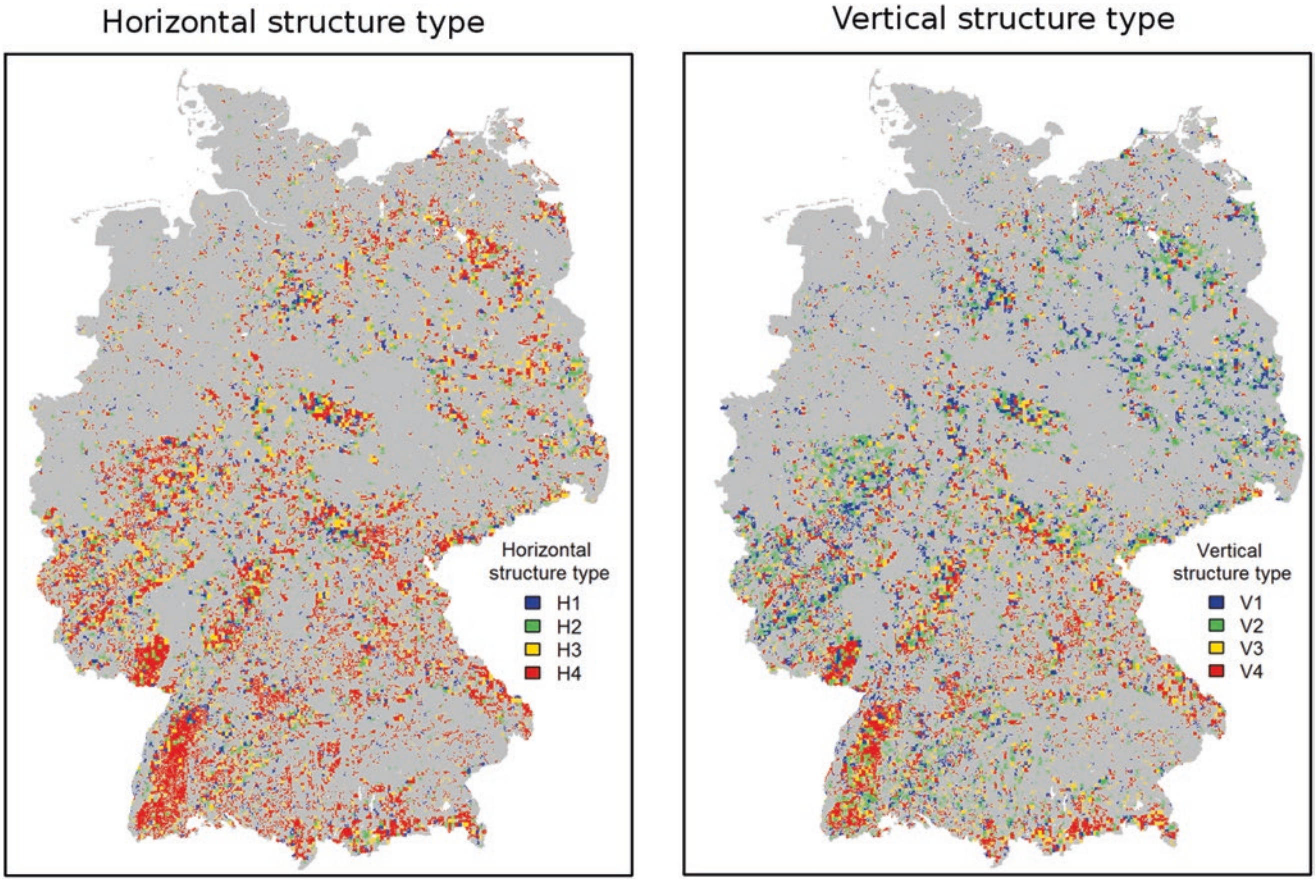

Fig. 11.2 Map of horizontal (H) and vertical (V) forest structure types in Germany estimated from simulated lidar data based on the German national forest inventory [11]. As horizontal index we used basal area (H1: 0–15, H2: 15–25, H3: 25–35, H4: >35 [m²]), as vertical index the tree height heterogeneity (V1: 0–1, V2: 1–2, V3: 2–3, V4: >3 [m]). Class 1 stands for a homogenous structure, class 4 for a heterogeneous structure

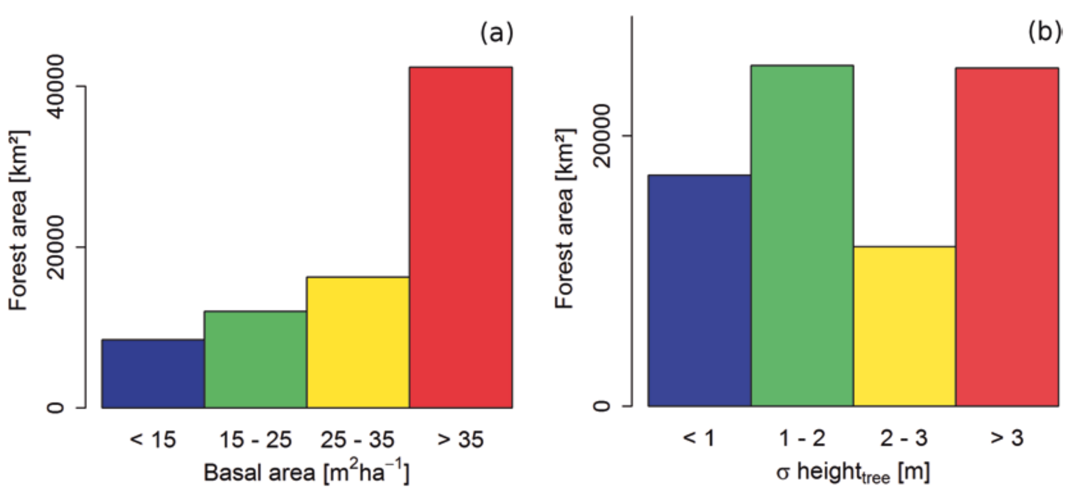

Fig. 11.3 Amount of forest area with a specific forest structure in Germany: horizontal (**a**) and vertical (**b**) forest structure types, estimated from simulated lidar data based on the German national forest inventory [11]. As horizontal index we used basal area, as vertical index tree height heterogeneity

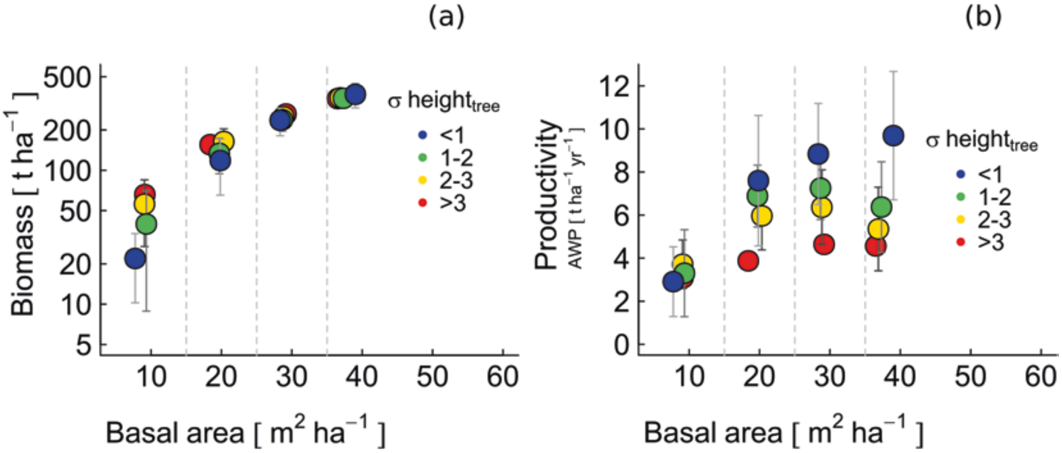

Fig. 11.4 Mean above-ground biomass (**a**) and mean aboveground wood productivity AWP (**b**) vs. forest structure classes. As horizontal structure index we used basal area (m^2 / ha), as vertical structure index we used tree height heterogeneity (m; in colours). Error bars indicate the standard deviation

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Mapping Land System Archetypes to Understand Drivers of Ecosystem Service Risks

12

Tomáš Václavík, Sven Lautenbach, Tobias Kuemmerle, and Ralf Seppelt

12.1 Identifying Land System Archetypes

Land system archetypes are unique patterns of land-use intensity within prevailing environmental and socio-economic conditions that occur repeatedly across the terrestrial surface of the earth [1]. We identify these archetypical patterns based on 32 land-use indicators available at the global scale (Table 12.1). The intensity indicators characterize land use in terms of inputs (e.g., extent of cropland, fertilizer input, irrigation), outputs (e.g., crop yields, production indicators) and properties of the social-ecological system (e.g., yield gap representing the difference between actual production and potential agro-ecological productivity) [2]. Environmental indicators include climate, soil, and vegetation characteristics that are known to drive and constrain the intensity and form of land use. Socio-economic indicators characterize the social, economic, and political background of land systems (e.g., population density, gross domestic product, political stability, accessibility). Using self-organizing maps (SOM), an unsupervised clustering technique that reduces high-dimensional data by grouping observations based on their similarity and location, we characterize and map twelve land system archetypes at the global scale.

The map of global archetypes reveals a clustered pattern of land systems across the world, ranging from barren and marginal lands with low land-use intensities, through pastoral and forest mosaic systems, to intensive cropping systems dominated by high agricultural inputs (Fig. 12.1). The combination of land-use indicators and the underlying conditions that best characterize each archetype is summarized in Fig. 12.2. The results show unexpected similarities in land systems in many regions (e.g., the extensive cropping systems archetype in East Europe, India, Argentina and China), but also a diversity of land-use forms at a sub-national scale, such as in China or India. These archetypical patterns imply that place-based approaches are needed to develop regional strategies for sustainable management of land and ecosystem services.

Which ecosystem services are addressed? Food provisioning, crop production

What is the research question addressed? Which are the archetypical patterns in global land systems? What insights do land system archetypes provide into potential drivers of and impacts on ecosystem services?

Which method has been applied? Self-organising maps applied to more than 30 land system indicators

What is the main result? The results identify land systems with risks to food provisioning due to soil erosion and regions with potential to increase crop production and resilience in terms of food security

What is concluded, recommended? Mapping global land systems archetypes allow providing science-based recommendations for regions with certain land-use types on how to avoid or mitigate negative consequences of land use. It represents a first step towards better understanding the spatial patterns of human-environment interactions and the environmental and social drivers of ecosystem service risks

12.2 Insights into Ecosystem Service Risks

The archetype approach facilitates an integrative understanding of land systems and provides insights into potential drivers of and impacts on ecosystem services, which may remain uncovered if they are studied in isolation. For example, archetypes help to identify generic patterns of land pressures and ecosystem service risks. They also allow providing science-based recommendations for regions with certain land-use types on how to avoid or mitigate the negative consequences

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Table 12.1 Datasets used for classification of land system archetypes (*Reprinted from* Václavík et al. [1]; *with permission from Elsevier*)

Archetype indicator	Spatial resolution	Unit
Land-use intensity indicators		
Cropland area	5 arc-minutes	km^2 per grid cell
Cropland area trend	5 arc-minutes	km^2 per grid cell
Pasture area	5 arc-minutes	km^2 per grid cell
Pasture area trend	5 arc-minutes	km^2 per grid cell
Nitrogen fertilizer	0.5 arc-degrees	kg ha^{-1}
Irrigation	5 arc-minutes	ha per grid cell
Soil erosion	5 arc-minutes	Mg ha^{-1} $year^{-1}$
Yields (wheat, maize, rice)	5 arc-minutes	t ha^{-1} $year^{-1}$
Yield gaps (wheat, maize, rice)	5 arc-minutes	1000 t
Total production index	national level	index
HANPP	5 arc-minutes	% of NPP_0
Environmental indicators		
Temperature	5 arc-minutes	°C × 10
Diurnal temperature range	5 arc-minutes	°C × 10
Precipitation	5 arc-minutes	mm
Precipitation seasonality	5 arc-minutes	Coeff. of variation
Solar radiation	5 arc-minutes	W m^{-2}
Climate anomalies	5 arc-degrees	°C × 10
NDVI – mean	4.36 arc-minutes	Index
NDVI – seasonality	4.36 arc-minutes	Index
Soil organic carbon	5 arc-minutes	g C kg^{-1} of soil
Species richness	calculated from range polygons	Number of species per grid cell
Socio-economic indicators		
Gross domestic product (GDP)	national level	$ per capita
Gross domestic product in agriculture	national level	% of GDP
Capital stock in agriculture	national level	$
Population density	2.5 arc-minutes	persons km^{-2}
Population density trend	2.5 arc-minutes	persons km^{-2}
Political stability	national level	index
Accessibility	0.5 arc-minutes	minutes of travel time

HANPP human appropriation of net primary production, *NDVI* normalized difference vegetation index

of land use on ecosystem services (see Response 1 of the ecosystem service risk framework).

12.2.1 Example 1: Risks to Food Provisioning Due to Soil Erosion

Based on the considered land-use indicators, several regions in tropical Latin America and Southeast Asia are classified as "degraded forest/cropland systems in the tropics" (Fig. 12.3). These systems are characterized by extremely high soil erosion and represent areas where patches of rainforest were converted to cropland. Although soil erosion occurs in other systems too, these regions are particularly affected by the loss of soil fertility because of their high agricultural inputs, relatively poor economy, and strong dependence on agricultural production. The underlying socio-economic data, showing that food production is important for the national economy of the local countries, emphasize the need to develop and apply erosion control measures for these regions. Therefore, this archetype pinpoints regions that may require similar policy responses and highlights heterogeneity (e.g., within countries), of which decision makers should be aware. Although data on forest management intensity are not available globally, this archetype matches well with the hotspots of forest cover change [3].

12.2.2 Example 2: Opportunities to Increase Resilience of Land Systems

It has been recognized that new approaches to agriculture that would prevent cropland expansion, close yield gaps, and increase cropping efficiency should be implemented to sustain future food demands while shrinking agriculture's environmental footprint [4]. Analyses of land systems can help identify strategies for particular regions and

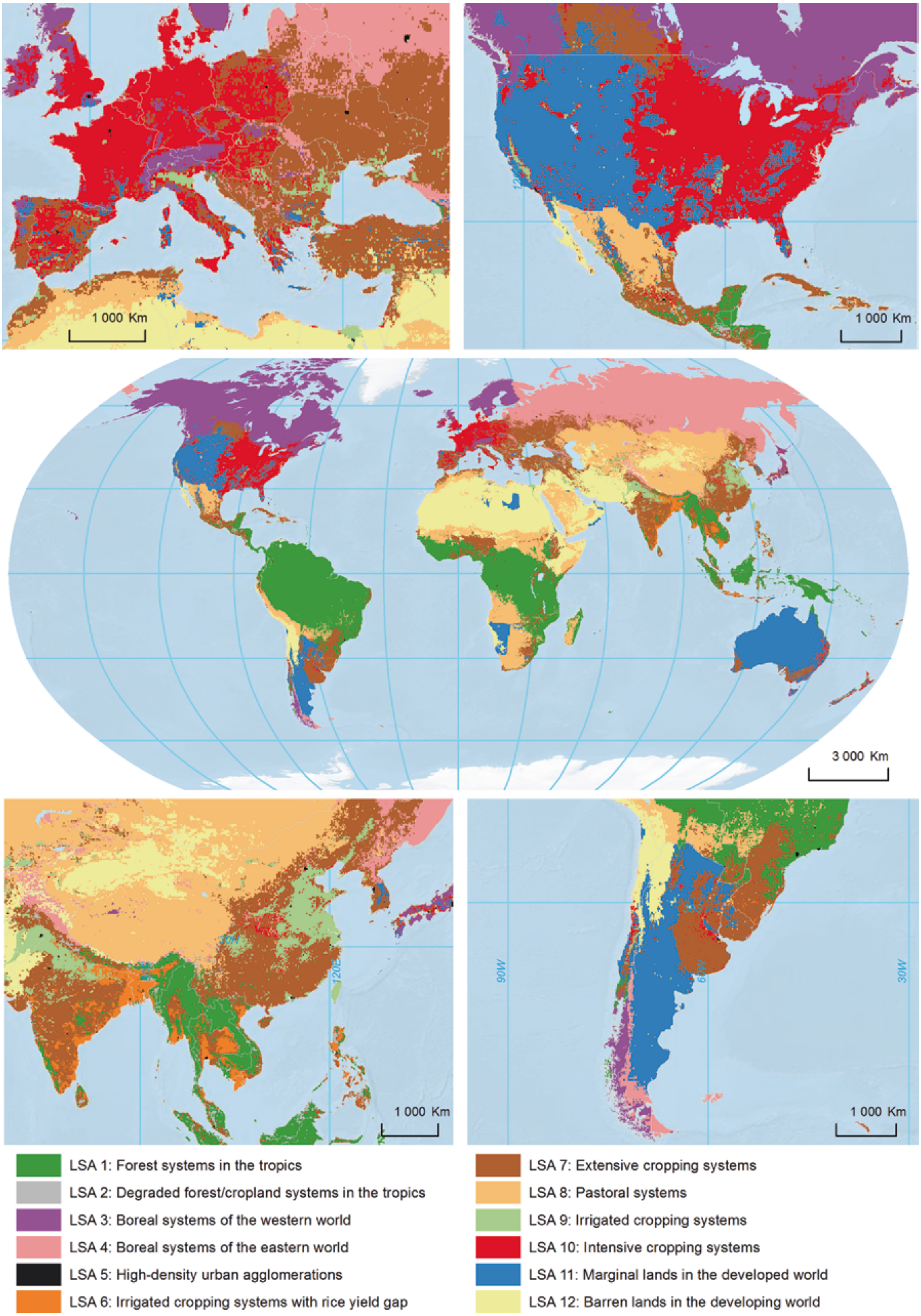

Fig. 12.1 Global land system archetypes (LSAs): world map and regional areas (*Reprinted from* Václavík et al. [1]; *with permission from Elsevier*)

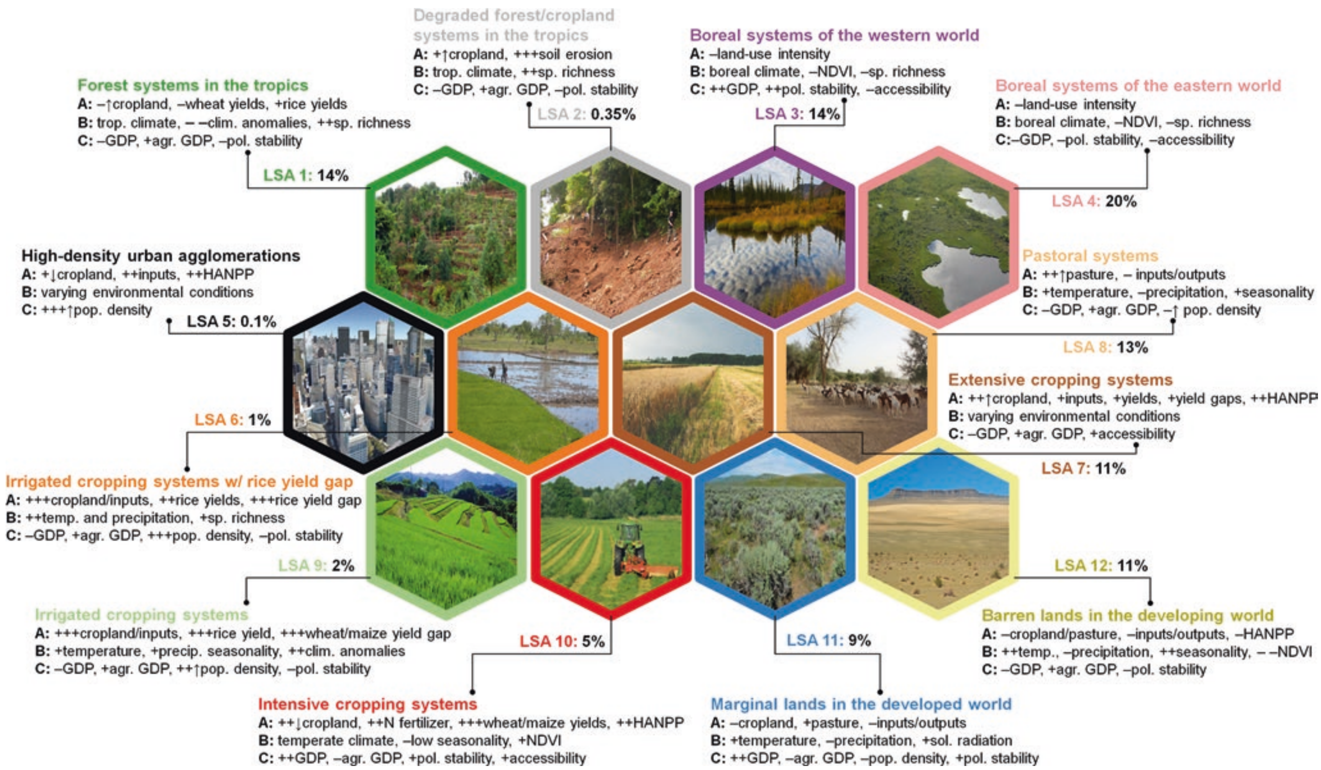

Fig. 12.2 Overview of land system archetypes (LSAs), summarizing major land-use intensity indicators (**a**), environmental conditions (**b**), and socio-economic factors (**c**) that best characterize each archetype. The + and – signs show whether the factor is above or below global average (+ is up to 1 s.d., ++ is 1–2 s.d., +++ is >2 s.d.); the ↑ and ↓ signs signify increasing/decreasing trends within the last 50 years; the numbers represent percentages of terrestrial land coverage. (*Reprinted from* Václavík et al. [1]; *with permission from Elsevier*)

support the development of solution portfolios relevant for a particular place. For instance, while the differences between realized and attainable yields are relatively small in "intensive cropping systems," considerable opportunities for yield improvements exist in the "extensive cropping systems" archetype (Fig. 12.4). This is in congruence with other studies [5–7] showing that Eastern Europe and Sub-Saharan Africa represent relatively easily achievable opportunities for intensification of wheat and maize production through nutrient and water management. Such regions have high potential for enhancing their resilience in terms of food security by increasing their provisioning ecosystem services to only 50% of attainable yields. Considering that many of these regions are characterized by a considerably low political and economic stability, any type of land management, whether focused on adaptation to climate change or on closing yield gaps, needs to consider the limitations of land-use options due to social and political constraints.

Mapping global archetypes of land systems represents a first step towards better understanding the spatial patterns of human-environment interactions and the environmental and social drivers of ecosystem service risks. The archetype approach should be seen not as a static typology but as an adaptable blueprint for land system characterization that can be refined for a specific region and group of ecosystem services [8]. Such assessment requires finer-scale data and a particular set of indicators that are relevant to decision-makers for managing ecosystem services and their risks.

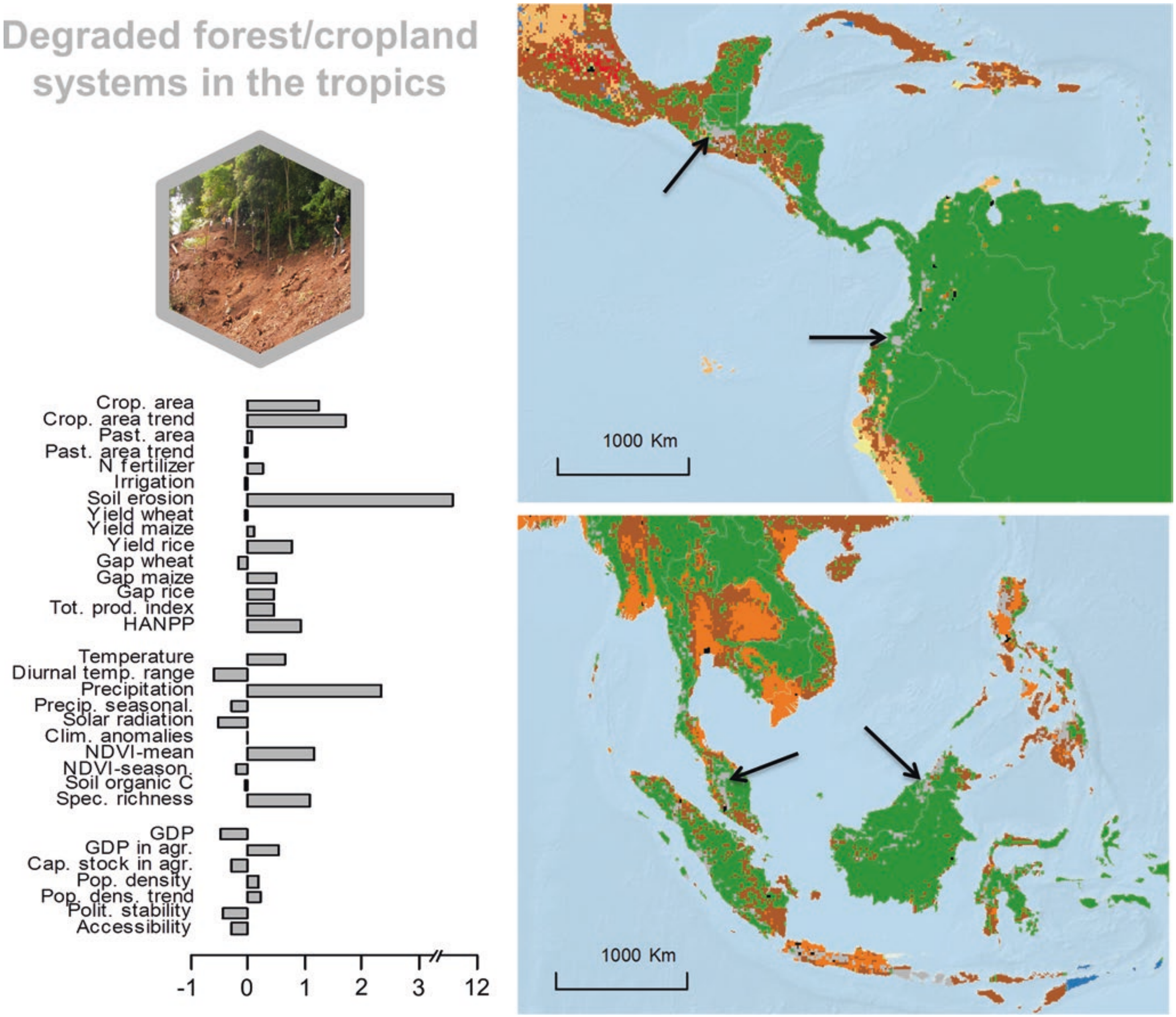

Fig. 12.3 Degraded forest/cropland systems in the tropics). An example of areas with high risks to agricultural production due to extreme soil erosion. The graph shows the combination of normalized values of land-use indicators that best characterize this archetype. Zero on the x-axis is the global mean, so the bars show whether and how much an indicator is above or below the global mean (units in s.d.). (*Reprinted from* Václavík et al. [1]; *with permission from Elsevier*)

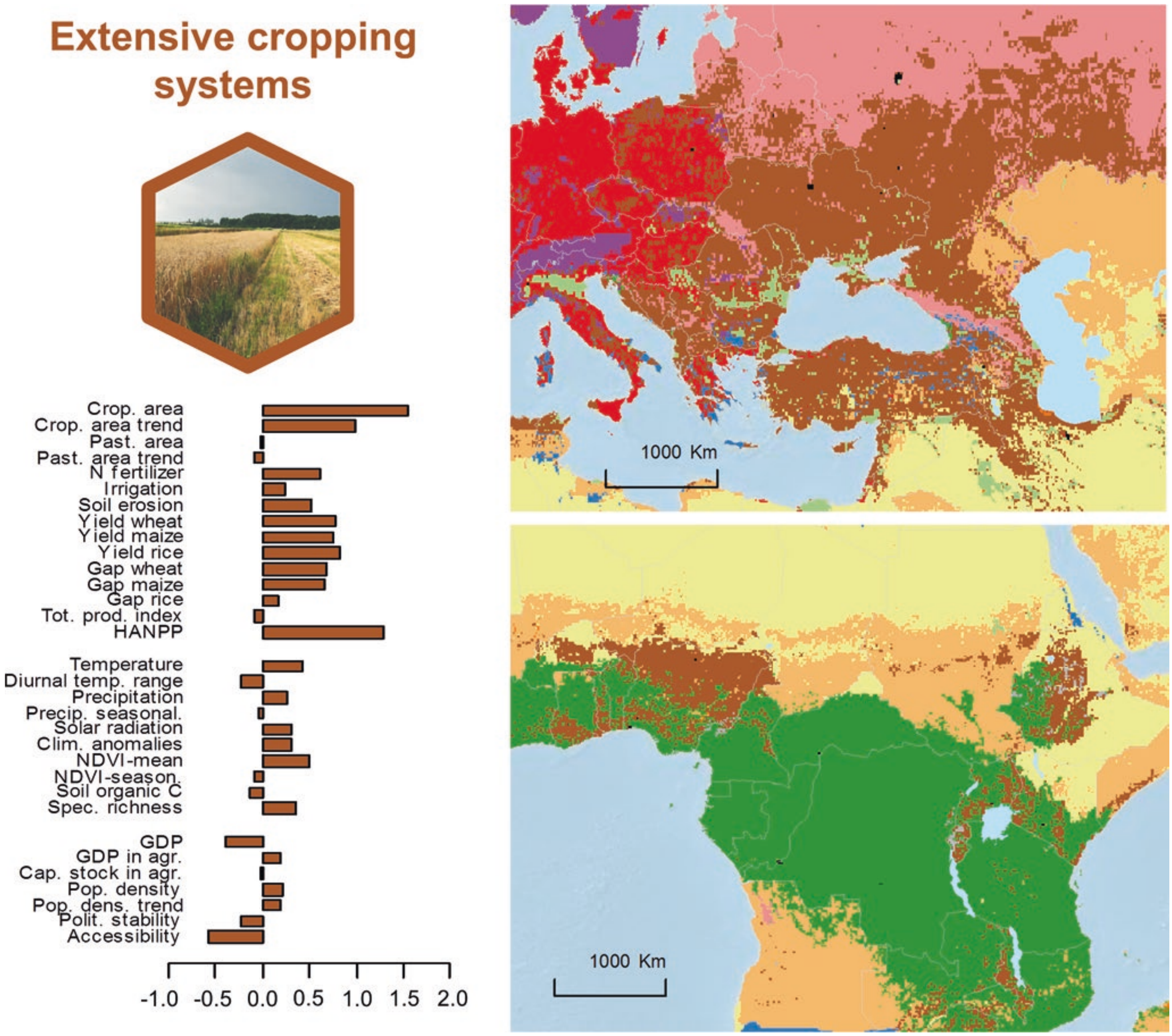

Fig. 12.4 Extensive cropping systems. An example of areas with a high potential for closing the yield gap and thus increasing the regions' resilience in terms of food security. The graph shows the combination of normalized values of land-use indicators that best characterize this archetype. Zero on the x-axis is the global mean, so the bars show whether and how much an indicator is above or below the global mean (units in s.d.) (*Reprinted from* Václavík et al. [1]; *with permission from Elsevier*)

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Assessment of Soil Functions Affected by Soil Management

13

Hans-Jörg Vogel, Ute Wollschläger, Katharina Helming, Uwe Heinrich, Matthias Willms, Martin Wiesmeier, David Russell, and Uwe Franko

13.1 Introduction

Soil plays a central role in the functioning of terrestrial systems. This role is at risk given the enormous loss of soil through desertification and degradation amounting to 12 million hectares per year [1]. In addition, a considerable amount of soil is lost through conversion to building areas (e.g., 70 hectares per day in Germany). Most of the remaining soil is managed for agriculture and forestry, and its functioning might be jeopardized by non-sustainable management practices. Because of the multitude and complexity of processes involved, it is a formidable challenge for soil science to predict the positive and negative impacts of soil management practices on the ensemble of soil functions within our terrestrial environment. Robust, science-based, predictive capabilities are a prerequisite to appropriately assessing soils and their functions and to providing informed recommendations for decision-making.

To reach this ambitious goal, there are a number of critical steps that need to be tackled. It starts with the identification of soil functions that are essential for the functioning of terrestrial systems and the livelihood of human society. Then, we need to develop concepts to observe and quantify the actual and potential contribution of a given soil to this set of soil functions. Finally, we need to establish the required understanding of processes and interactions within soil to actually predict the impact of soil management on the set of soil functions. In the following, we will discuss the actual state and possible developments along these lines.

Which ecosystem services are addressed? Provision of food, fibre, raw materials

Regulation: flood mitigation, filtering and recycling of nutrients, carbon storage and greenhouse gas regulation, habitat for biological activity

What is the research question addressed? How do we quantify soil functions based on indicators and systemic modelling?

Which method has been applied? Literature review

What is the main result? The evaluation of soil functions need to be based on a set of functional soil properties providing integral information on physical, chemical, and biological processes

What is concluded, recommended? Soil function needs to be evaluated based on a systemic model concept and in a site-specific way

13.2 Soil Functions

Sustainability of soil management refers to safeguarding soil functions. The most prominent function of soil is to provide the basis for plant growth, including water and nutrient supply and disease suppression. Almost all food production and a substantial fraction of raw materials and energy are derived from plants growing on soil. However, besides this fundamental basis of agriculture and forestry, there are more soil functions and related ecosystem services essential for the functioning of terrestrial systems and for human well-being, respectively. These include the role of soil in recycling of organic matter, the importance of soils as an efficient filter to produce clean drinking water, the soil's storage capacity for water as a critical contribution to flood protection, the storage of carbon and water relevant for climate control and, last but not least, the soil's capability to harbor a myriad of different organisms which are the "engine" for a multitude of soil processes [2] (Fig. 13.1).

Obviously, soil has more important functions when adding aspects of engineering and human culture as done, for example, by the European Commission [3]. In the following, however, our focus is on natural soil functions, which are essential for the functioning of terrestrial systems as illustrated in Fig. 13.1.

Blum was the first to frame a systemic concept linking soil processes via soil functions to services for environment

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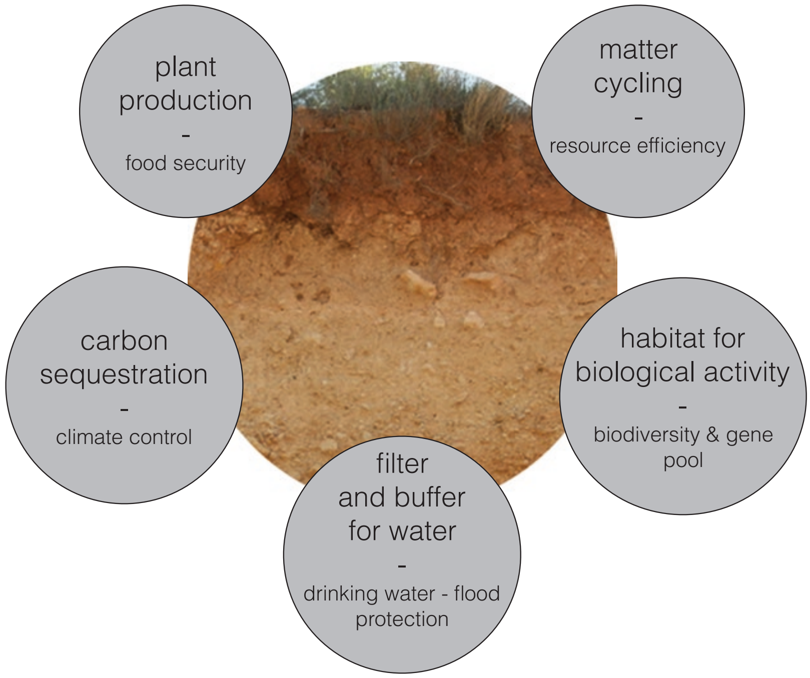

Fig. 13.1 Natural soil functions essential for the functioning of terrestrial systems and related ecosystem services

and society [4]. The latter is referred to as soil's Ecosystem Services and implies a societal valuation system for soil functions, the implementation of which is still a matter of debate [5]. The need for such a concept seems to be obvious, since it requires including expected changes in soil functions into decision-making within a socio-economic context. It is important to note that we consider soil functions to be observable properties that emerge from complex interactions of natural processes and are the basis for the functioning of terrestrial ecosystems, while ecosystem services can only be defined in the context of the actual human perception and may change according to the societal context [6].

13.3 Quantification of Soil Functions

The five soil functions illustrated in Fig. 13.1 cannot be measured directly. There are no simple sensors to measure the capacity of soil to store carbon, filter water, produce biomass, transform matter, or serve as habitat for organismal activity. These soil functions are emergent properties, which are generated by complex interactions between a multitude of physical, chemical, and biological processes. These processes are coupled in many ways, and the change in soil properties in response to external perturbations (i.e., by agricultural practices) is mostly non-linear as is typical for biological systems. This is why soils are considered to be complex systems characterized by the process of self-organization [7]. Because of this complexity, the behavior of soil cannot be reproduced by just combining the small-scale physical, biological, and chemical processes, which might be well understood on their own—the whole is more than the sum of its parts.

As a practicable approach we may choose to rely on macroscopic soil properties, which can be observed and measured, and which contain some substantial amount of information with respect to one or more of the soil functions. Such proxies are typically referred to as indicators. While indicators have long and widely been used for various applications, there is at this time no generally agreed concept. When scanning through the more recent literature [8, 9] on soil quality indicators, the set of chosen indicators is quite similar. There are physical indicators related to soil structure (bulk density, water capacity, macro-porosity and, aggregate stability) and chemical indicators related to matter turnover (organic carbon content, cation exchange capacity, pH). The use of biological indicators frequently assesses the abundances and composition of soil organisms having a significant impact on soil structure formation, the incorporation of organic matter, and the stimulation of microbial activity [10, 11]. Microbial diversity has been found to be characterized by an enormous redundancy with respect to matter turnover, such that the structure and diversity of microbial communities in soil seems to be way above some critical limit, so that it might provide only limited information on overall soil functions [12]. Instead of concentrating on taxonomic community parameters of biodiversity, a more meaningful approach regarding soil functions would stress community functional diversity as well as key species driving specific processes such as bioturbation or nutrient transformation [10, 13, 14]. While it is beyond dispute that soil functional biodiversity renders soil resistant and resilient to disturbance and stress [15, 16], the extent to which this is conferred by the linkages and interactions in soil biogenic networks is far from being completely understood [16]. Therefore, limited sets of biological indicators (i.e., microbial and faunal organism groups, activity measurements) are currently considered most appropriate for assessing the multitude of biological drivers of soil functions [17].

An additional general difficulty for the evaluation of soil functions is the fact that soils are different. Depending on the parent material of soil formation, local topography, and climatic conditions, there is a multitude of soil types differing in their physico-chemical properties and biological inventory. Hence, the valuation of indicators with respect to soil functions can only be performed in a site-specific way. A general framework for doing so still needs to be developed; a single well-defined universal indicator does not exist.

13.4 Dynamics of Soil Functions

To evaluate and predict the impact of soil management on soil functions, we need to understand how management practices such as fertilization strategies, tillage systems, and crop rotations affect these functions. However, this is only a first step. In addition, we need to understand how soil processes counteract external perturbations. Soil possesses a considerable potential for recovering from compaction induced by traffic or from structure disturbance due to tillage. This is mainly brought about by biological activity, including that of plants [18], and is reflected by the observed stability and resilience of soil functions. However, as is typical for complex systems, there are critical limits of perturbations, where internal feedbacks may lead to a bifurcation between stability and degradation. Examples include (1) critical soil compaction that exceeds the biological potential of structure reformation; and (2) a critical removal of organic matter needed for biomass production, so that the return to soil drops below a level sufficient to fuel the soil biological engine [19]. Identifying such limits is crucial when assessing the risk of losing essential soil functions.

As discussed in the previous sections, the dynamics of soil functions can be evaluated by analyzing the dynamics of a set of suitable indicators or, more generally, of functional soil properties. One approach is to integrate current process understanding into appropriate models to describe the dynamics of soil functions or related indicators. Such models are typically designed for specific processes such as soil carbon dynamics, soil water flow, soil compaction, or the emission of greenhouse gases. In many cases, these models are parameterized according to local functional soil properties, which are considered to be fixed material properties. However, the change in soil functions coincides with the change in these properties and these properties are indeed affected by soil management practices (e.g., soil bulk density, water capacity, pH, soil carbon, earthworms, etc.). There is, thus, a need for model approaches to describe the dynamics of such functional properties.

An alternative systemic approach is currently being developed by the BonaRes Centre for Soil Research in the framework of the BMBF funding program BonaRes (www.bonares.de). The underlying concept is to characterize a local soil as a specific combination of functional soil properties. Hence, what is traditionally known as "soil type" following some classification scheme may translate into a characteristic combination of functional properties (e.g., bulk density, organic carbon content, functional diversity of soil biota). To describe the dynamics of the whole system, the interaction between functional properties is described based on available process understanding or based on empirical relations found under specific site conditions. This approach is still in its infancy, but may open new avenues for a systemic evaluation of soil functions.

Irrespective of the chosen approach, there is an urgent need for information on the spatial distribution of soils and their properties (i.e., indicators) at the regional and global level. Thereby, the spatial resolution of this information needs to correspond to the characteristic length scale of the distribution pattern of different soil types which is at the scale to some tens of meters. There is still a long way to go, and there are considerable differences in available soil information between regions and federal and national states. However, the tools to gather valuable spatial information, especially through various techniques of proximal and remote sensing [20], are steadily improving and the culture developing towards open data is promising [21]. This type of information can be directly used to estimate and infer soil attributes at the systemic level or to parameterize targeted model tools. An example for the latter is given in the following section.

13.5 Example: The Potential of Carbon (C) Sequestration

Soil carbon dynamics are typically modeled by describing the fate of fresh organic matter input through a cascade of decomposition processes leading to mineralization, storage, and biomass. The site-specific dynamics (i.e., rate parameters) depend on soil properties such as soil texture and soil structure (related to C storage capacity) and biological inventory, but also the climatic boundary conditions in terms of temperature and precipitation distribution. In the following example (Fig. 13.2), the CCB model [22, 23] was used to classify the potential of German arable soils for additional carbon sequestration in the case that the tillage system is

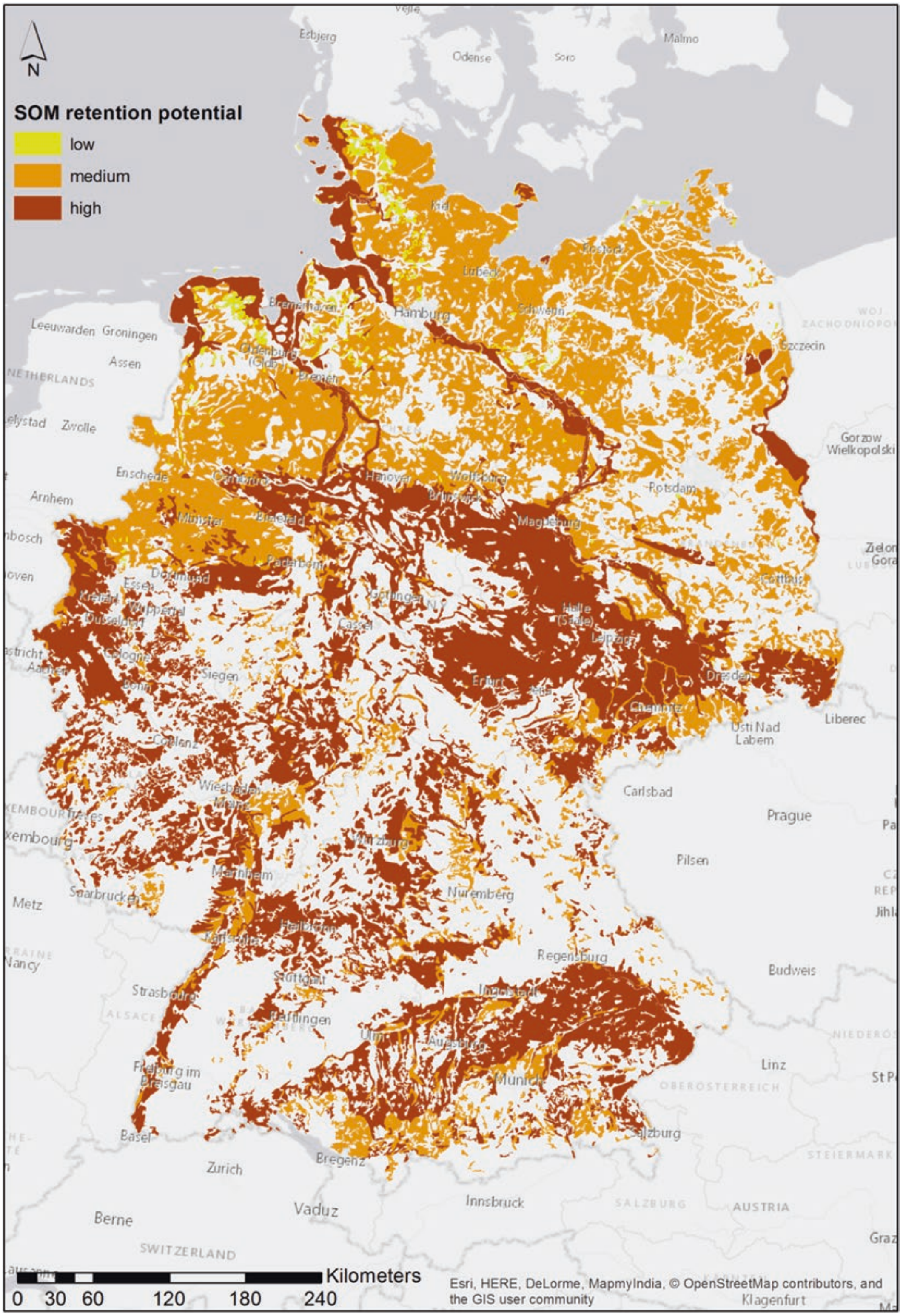

Fig. 13.2 The potential of arable soils in Germany for additional carbon sequestration when changing from conventional to conservation tillage as predicted by the CCB model. The classification levels low, medium or high refers to the effect of conservation tillage to increase soil carbon storage. *SOM* soil organic matter

changed from conventional (i.e., plowing) to conservation tillage with limited disturbance of soil structure. The basis for the results shown in Fig. 13.2 is the German Soil Map (BÜK 1000, [24]) and average climate data from 1981 to 2010 that were available on a 1 × 1 km grid [25]. The model reflects an increased potential of silty and loamy soils for additional carbon sequestration. This is due to the fact that the secondary structure of fine-textured soils in contrast to sandy soils physically protects carbon when not disturbed by conventional tillage.

As with all modeling approaches, this is based on simplifying assumptions so that the results are afflicted by considerable uncertainty. In this case, among other simplifications, only the upper soil horizon (0–30 cm) is considered and the impact of local climate (i.e., water supply and temperature) is averaged for a whole year and lumped into an efficient parameter denoted as "biologically active time" [26]. Hence, some aspects known to be relevant, such as crop rotation, precipitation pattern, biological assemblages, or the depth distribution of organic matter within the topsoil [27], are not considered. Moreover, it should be noted that the choice of the tillage system might impact other soil functions, such as soil productivity or the quality of drinking water, as influenced by different inputs of herbicides accompanying different tillage systems [28].

However, despite these limitations, this example demonstrates the potential of combining process understanding represented by a suitable model with spatial knowledge on the distribution of relevant soil properties (i.e., functional soil maps). In the present example, soil data are at a rather coarse scale and provide information mainly on soil texture classes. More generally, this type of model-based evaluation of scenarios can be extended towards additional drivers such as the impact of changing climatic conditions and additional processes related to other important soil functions.

13.6 Summary

Soil functions are critical for the functioning of terrestrial systems on earth and for the livelihood of human societies. Nonetheless, a quantitative evaluation of soil functions is difficult, but it is essential if soil functions are to be adequately included in decision-making with respect to land management. Such an evaluation needs to be done in a site-specific manner because soils and their properties and potentials vary vastly along the landscape. A future need and a prerequisite for a reliable risk assessment is to develop systemic model approaches accounting for the temporal change of functional soil properties in response to measures of soil management. This change can be abrupt and followed by a long period of time required for soil recovery or the development towards another stable state. This is due to the long timescale of the underlying natural processes of physical, chemical, and biological feedbacks. For upscaling of local understanding of different soil types to the regional scale, existing soil maps need to be improved in terms of spatial resolution and information on functional soil properties.

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14 Mediterranean Wetlands: A Gradient from Natural Resilience to a Fragile Social-Ecosystem

Ilse R. Geijzendorffer, Thomas Galewski, Anis Guelmami, Christian Perennou, Nadege Popoff, and Patrick Grillas

14.1 Introduction to Mediterranean Wetlands

Wetlands in countries around the Mediterranean Sea have provided ecosystem services to its population for more than 6000 years (Nile, Mesopotamian civilizations) [1–3]. Initially these wetlands provided drinking water, fishing and hunting grounds, as well as protection against flooding from rivers and seas. Later, these services extended to the supply of water for agriculture, households, energy and industries, and the supply of material for construction (e.g., *Phragmites australis*). In addition, wetlands have been increasingly managed for the cultivation of crops, such as rice, and grazing for livestock. These ecosystem services have resulted from a social-ecological system and are predominantly co-produced, not in the least due to the importance of highly human-controlled hydrological systems. Over time, this close co-existence of man and nature has produced multiple cultural traditions related to the characteristics of Mediterranean wetlands, such as the development of wetland adapted livestock in various places (e.g., local breeds of horses and cattle in the Camargue, France, and local breeds of cattle in Tuscany [Maremma], Italy, or Prespa [Greece], Menorca [Spain]).

The interplay of nature, climate, and society around the Mediterranean basin has resulted in global recognition as a biodiversity hotpot that attracts tourists from far (Fig. 14.1). For the wetlands, the emblematic bird species are an especially great source of attraction, as well as the endangered species and the sheer diversity that can be found on any particular day. Unfortunately, this same interplay poses a serious threat to at least 1000 species (Fig. 14.2). As natural heritage and diversity is the most frequently included ecosystem service in global sustainability targets [4], this is very worrying.

Rising population numbers, consumption patterns that increase demands on resources, and reduced water renewal rates have been putting existing social-ecological interactions under considerable stress. This renders both ecosystems and people more vulnerable to naturally existing hazards, because it increases the likelihood that a hazard will occur and increases the potential damage that may be caused.

Which ecosystem services are addressed? Hazard regulation (flooding) is the most important one
For the rest we mention:
Provisioning: fishing, freshwater
Regulating: habitats for biodiversity
Cultural: bird watching

What is the research question addressed? How is the resilience of ecosystem services provided by Mediterranean wetlands changing over time?

Which method has been applied? Literature review

What is the main result? Examples of how different aspects of ES flows are being affected on the one hand through changes in the ecosystems and on the other hand through changes in the societal demand

What is concluded, recommended? The general trends of declining biodiversity and reducing water availability, as well as an increase in demand for ecosystem services, make Mediterranean wetlands and the people that depend on them less resilient and increasingly exposed and vulnerable to physical and economical hazards that naturally occur in the Mediterranean basin

14.2 Ecosystem Capacity to Provide Ecosystem Services at Risk

The capacity of Mediterranean wetlands to provide ecosystem services has been reducing rapidly in recent years. Although this is clear for many of the actors depending on the services from Mediterranean wetlands, the lack of general

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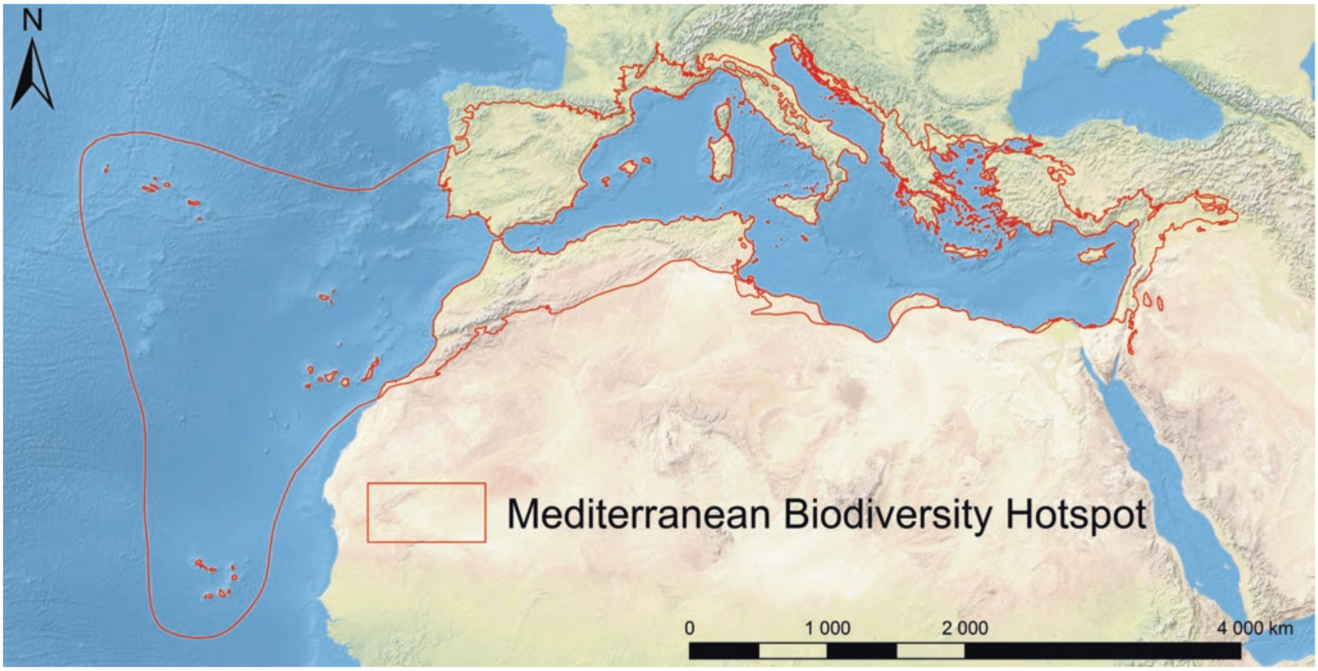

Fig. 14.1 Hotspots of biodiversity around the Mediterranean Sea [17]

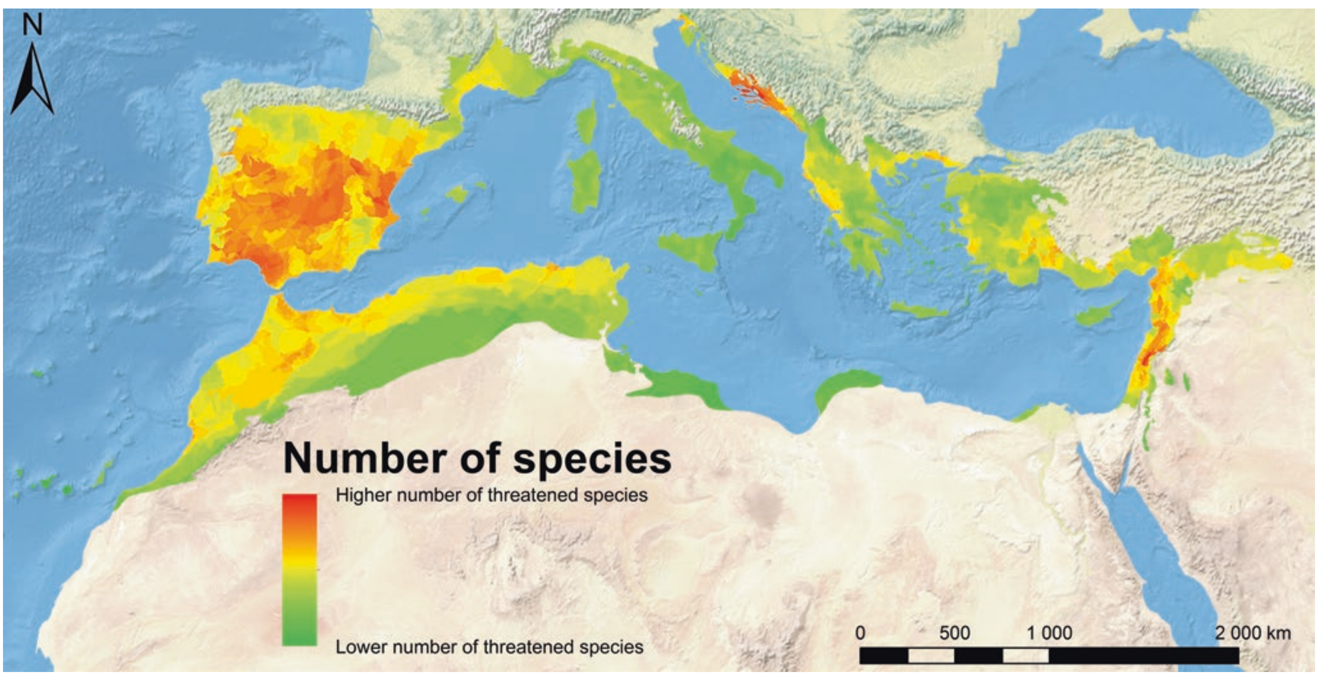

Fig. 14.2 Map of the wetland species threatened with extinction in the Mediterranean biodiversity hotspot. Data extracted from the IUCN database and BirdLife International. Species included for this analysis are all mammals, birds, reptiles and amphibians associated with wetlands as well as all freshwater fishes and odonates. Especially the watersheds with endangered endemic freshwater fish species stand out in Spain, the south of Croatia/Bosnia, the Danube, the rivers of Oronto and Jordan and Syria/Israel

baseline data is a real problem for quantifying the actual loss of services. For instance, the first estimation of where Mediterranean wetlands are and how much might have been lost up to the present time did not exist until very recently, in 2012 (Figs. 14.3 and 14.4, respectively) [5, 6]. Additionally, where historical baseline information does become available, it can radically shift our perception of the ecological state of the wetlands (see for an example Galewski and Devictor [7] for the revaluation of trends in bird species using a pre-Industrial era baseline). But even if we can only base our estimation of the state of the ecosystems to produce ecosystem services on data that is available, it is clear that

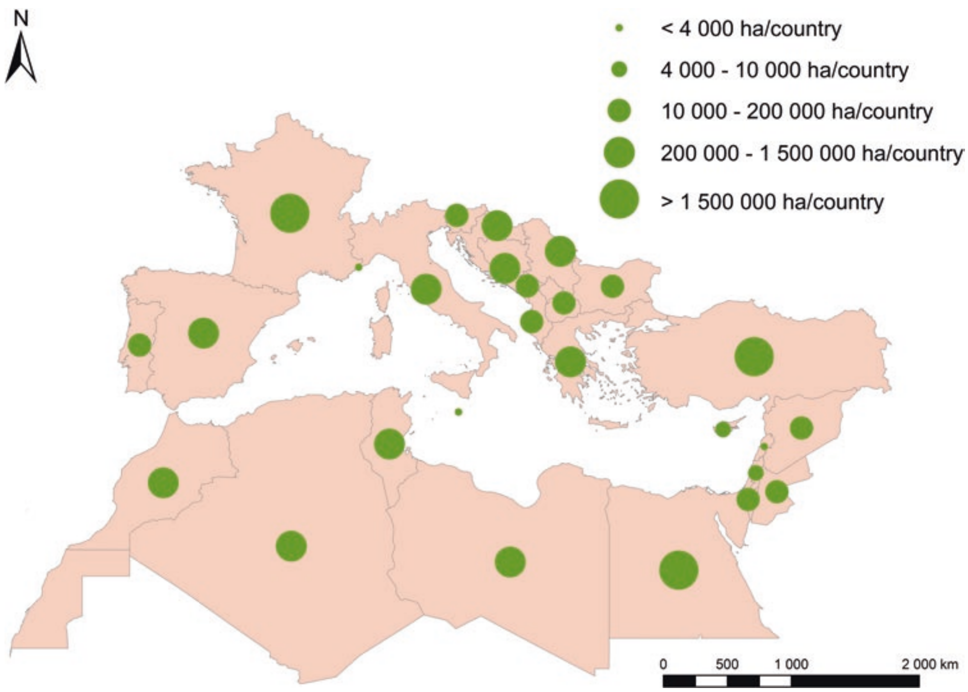

Fig. 14.3 Estimated surface and location of 15–22 million ha of Mediterranean wetlands [5]

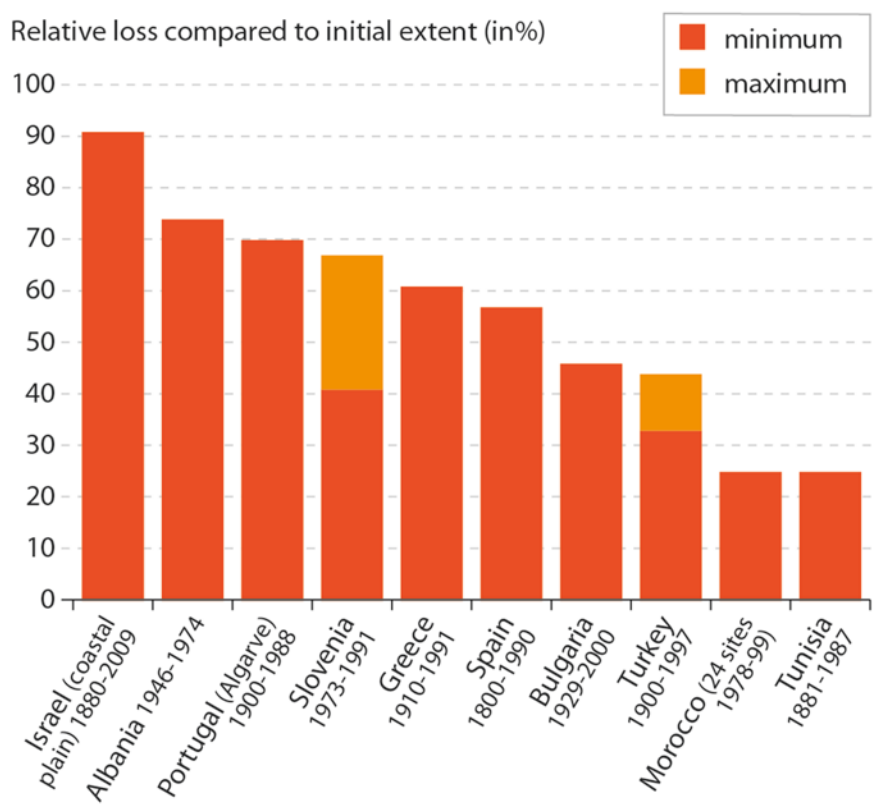

Fig. 14.4 Estimated loss of natural wetlands in selected countries/regions of the Mediterranean. Baseline year is 1900. In the absence of data, an estimate was made on the potential minimum (red) and maximum (orange) loss of the wetland surface [5, 6]

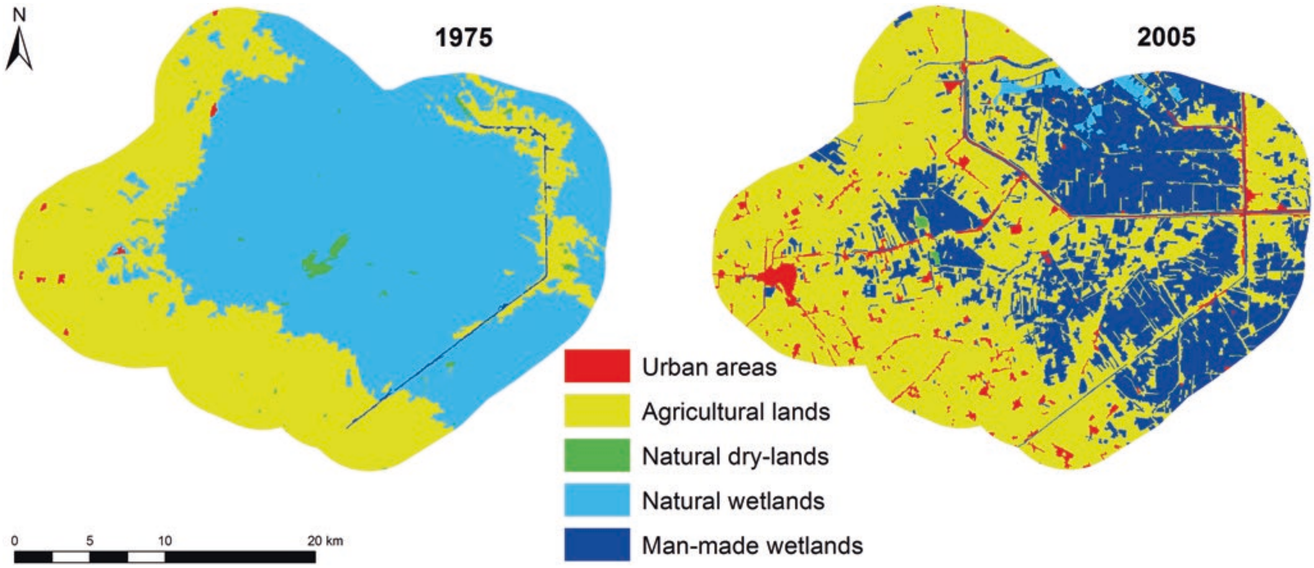

Fig. 14.5 Irreversible land cover changes in the wetlands of Sinnera and San El Hagar in the Nile Delta (1975–2005) [8]

due to the severe reduction of the surface of wetlands (Fig. 14.5), this capacity at local scale and at the scale of the Mediterranean basin has already been severely affected [8].

For many ecosystem services in general, the actual connection to ecological functions remains to be demonstrated, but recent work by Newbold et al. [9] looked at trends in species richness and abundance to identify that many habitats have already passed, or are on the brink of passing, the planetary safety boundary for resilient ecological functions. Although current research is on the way to define baselines for species richness and abundances in Mediterranean wetlands, many taxa show a startling decrease, and therefore a reduced resilience of the ecological functions can be assumed. Plants and animals combined, one out of three species living in Mediterranean wetlands is threatened with extinction [5]. At least 40 species of freshwater gastropods and fish already went extinct in Mediterranean countries in the past decades. Higher numbers of endangered species are found especially where there are "concentrations" of endemic wetland species: Iberian Peninsula, Balkans, southern Turkey, Near East, and northern Maghreb (Fig. 14.2).

14.3 Risk to Societal Use of Ecosystem Services

The reduction of the ecosystem capacity to provide ecosystem services is an effect not only of the extent and state of habitats and species, but the quantity and quality of available water, especially for Mediterranean wetlands, is also a crucial factor. Artificialization of the hydrological systems has occurred in nearly all Mediterranean wetlands, which is

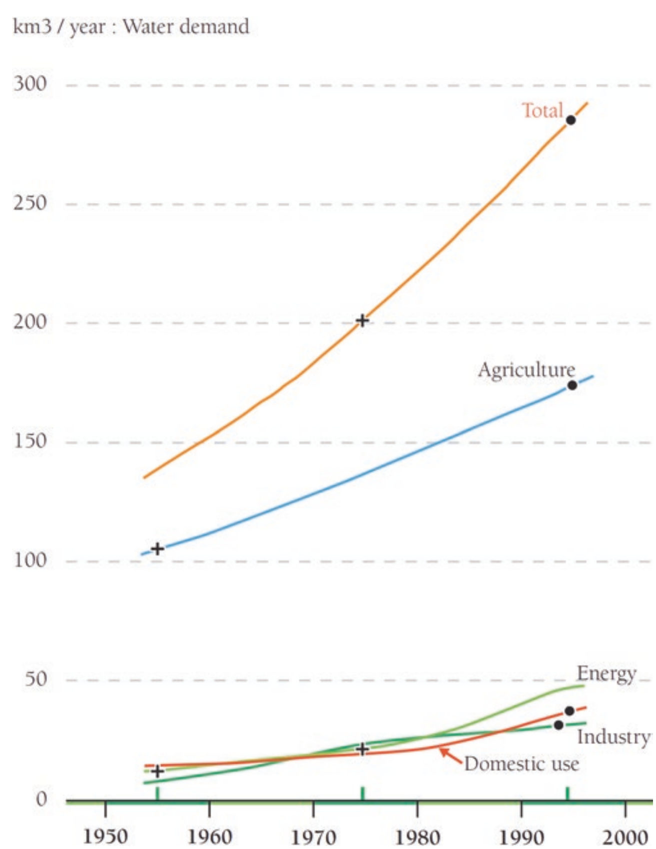

Fig. 14.6 Increase of freshwater use in the Mediterranean, with irrigation being the most dominant end use [5]

facilitating the withdrawal of increasing amounts of water from river flows, lakes, and aquifers for agricultural, industrial, and household consumption purposes (Fig. 14.6).

Related pollutants end up in the freshwater sources, thus additionally affecting water quality. Furthermore, changes in precipitation patterns and temperatures related to climate change have led to a reduction of freshwater renewal, and of the pollution dilution capacity of freshwater ecosystems [5]. The reduction in quantity and quality of freshwater arriving in Mediterranean wetlands directly affects the potential supply for ecosystem services, such as clean drinking water, fisheries, rice production, and medicinal plants, whereas many other ecosystem services are indirectly affected through the impact on underlying ecological processes and biodiversity.

Certain hazards have always been present in the Mediterranean basin, such as the flooding by rivers or the sea, occasional fires, periods of prolonged droughts, and heat waves. These hazards have always had negative impacts on human well-being, and society's capacity to cope with these risks and with the impacts of increasingly frequent hazards requires ever-growing investments and resources.

For one, human population numbers have been steeply increasing, especially in coastal areas and around areas with freshwater availability (Figs. 14.7 and 14.8). On the one hand, this has caused both infrastructure and people to be situated in areas prone to flooding. On the other hand, the related artificialisation of hydrological systems further reduced the capacity of ecosystems to provide regulating services (Fig. 14.5). The combined result is a higher number of people exposed to increasingly more frequent hazards, such as flooding or periods of droughts.

In addition, there has been an increasing demand for and consumption of energy, food, and water by individual households, agriculture, and industry. For some of these resources, importation of goods and services is an option that solves an immediate need, but it makes societies more dependent on international market prices. In addition, it contributes to externalizing the pressures on water, as measured by the international virtual water flows [10]. For other resources, import is a less obvious option, and poorer households especially suffer the consequences of local deficits in ecosystem services [11]. For instance, in North Africa, one out of three species of freshwater fish and aquatic plant species is used by local populations, providing them with direct socio-economic advantages [12, 13]. Similarly, among the 86 endemic freshwater plant species that only occur in northern Africa, 11 (12.8%) are known to be harvested by people for a direct use or for increasing their income [13]. Also in Mediterranean coastal lagoons, the European eel (*Anguilla anguilla*) catch, which forms an important resource for local fishermen, has steadily declined in the last decades [14] as a result of a combination of various pressures on wetlands and rivers. With increasing human population numbers and decreasing natural resources, increasing mismatches between the supply and the demand for ecosystem services are a prevailing trend for Mediterranean wetlands.

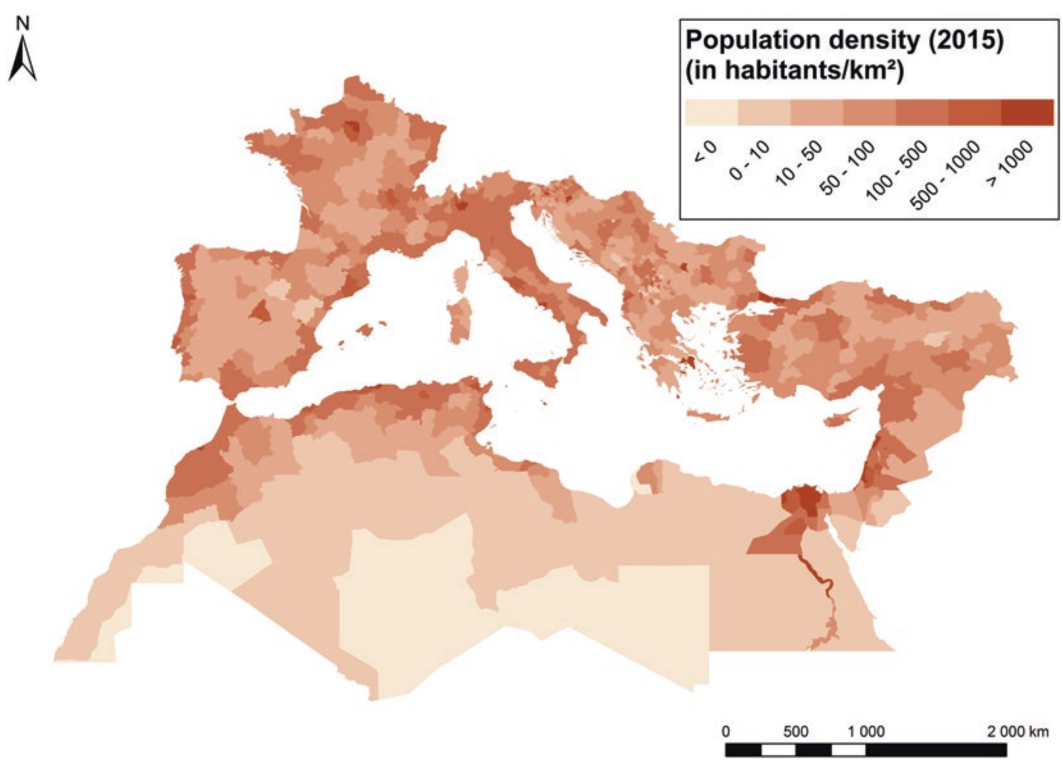

Fig. 14.7 Population density in the Mediterranean in 2015 [18]

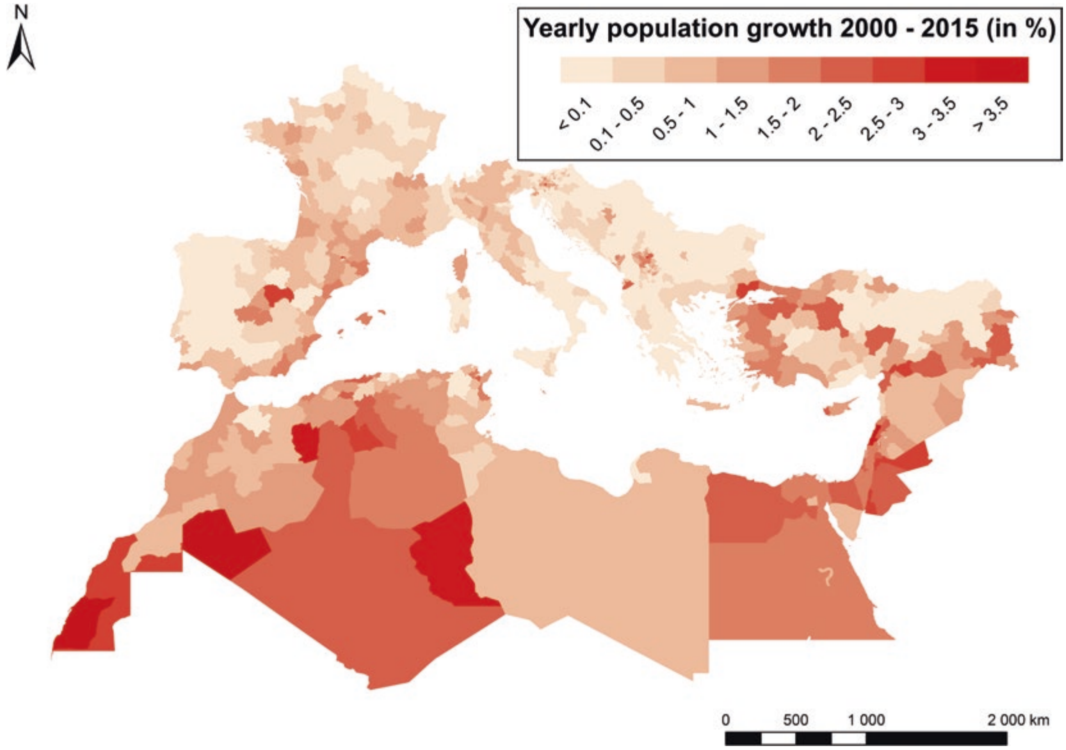

Fig. 14.8 Average annual population growth (in %) in the Mediterranean between 2000 and 2015 [18]

14.4 A Dangerous and Unpredictable Cocktail

Although hazards have always been present in the Mediterranean, the interplay of biological, societal, and climate processes have increased the uncertainty of the resilience of existing social-ecological systems of Mediterranean wetlands. The decreasing capacity of ecosystems to provide services, as well as the increasing vulnerability of the people depending on these services, provide a dangerous cocktail of potentially high risks and significant impacts.

In the case of flood protection regulation, the decrease of the surface of wetlands has greatly impacted the capacity of the ecosystem to provide the service, the risk of flooding has increased due to artificialisation of riverbeds, and the society has become more vulnerable to the risk as more people live in the sensitive areas [15, 16].

The end of supply for some services has already been predicted, such as in the case of eel fishing. The combined impacts of a high demand for eels, pollution, introduced pathogens, and the impact of dams on river connectivity and dispersal capacities, predict the disappearance of both this species as well as this traditional fishing practice [14].

For the other services, such as bird watching tourism, the number of people visiting wetlands continues to increase, regardless of or perhaps stimulated by, the number of red-list species that can be found in Mediterranean wetlands (internal report of the French Observatoire National des zones humides).

The general trends of declining biodiversity and reducing water availability, as well as an increase in demand for ecosystem services, make Mediterranean wetlands and the people that depend on them less resilient and increasingly exposed and vulnerable to physical and economical hazards that naturally occur in the Mediterranean basin. This fragility is likely to only further increase as these trends have yet to be countered and the frequency and intensity of hazards are likely to increase under the influence of climate change and other anthropogenic pressures. Conservation and sustainable use of Mediterranean wetlands are therefore a serious challenge, that must be faced with increasing urgency if we are sincerely concerned about human well-being in the Mediterranean basin.

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15 Vulnerability of Ecosystem Services in Farmland Depends on Landscape Management

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15.1 Introduction

Forty-four percent of Europe's terrestrial surface is covered with agricultural land. Thus, agriculture strongly influences Europe's environment, including ecological functions and processes. Agriculture provides direct benefits to humanity, such as food, feed, fuel, and fiber. In addition to agricultural production, farmland plays an important role for regulating services, such as carbon sequestration, water capture and retention, biological pest control, and pollination. As an interface between nature and human activities, agricultural landscapes endow people with a sense of place, enable livelihoods, ways of living, and offer space for recreation [1]. These and several other ecosystem services constitute the multifunctionality of the agricultural landscape that European agricultural policy seeks to achieve and maintain. Hence, ecosystem service management needs to navigate trade-offs between competing interests from local to landscape scales.

Two processes, land use intensification and land abandonment, are the main drivers of current changes in European agroecosystems. The consequences of these changes for human well-being have been only fairly explored. On the one hand, production of agricultural goods increases, either through the expansion of agricultural land or, more frequently, by intensification on existing farms. This happens through the use of higher yielding crop varieties, increased input of agrochemicals, and simplification and shortening of the crop rotation. Intensification also aims at higher cost-effectiveness in the short term, which involves consolidation of field sizes and the removal of semi-natural landscape elements such as hedgerows, field margins, and tree lines [2]. The consequences of intensification include landscape simplification, nutrient leaching, soil compaction, loss of soil fertility, and loss of biodiversity. On the other hand, land abandonment might also lead to a loss of landscape heterogeneity through biotic homogenization, thereby eroding habitats for open-land species.

Which ecosystem services are addressed? Multifunctionality of agricultural landscapes with particular focus on crop pollination and biological pest control

What is the research question addressed? What is the relative importance of local and landscape management for maintaining or enhancing functional biodiversity that provides ecosystem services such as biological control and pollination?

Which method has been applied? Mixture of literature review, experts' opinions, and case studies on pest predation and crop pollination in farmland in response to changing landscape heterogeneity

What is the main result? The multifunctionality of agricultural landscapes calls for managing trade-offs between ecosystem services. Enhancing functional biodiversity for pollination and biocontrol at a landscape scale requires a minimum of approx. 20% of semi-natural habitat, but improved cropland and fallow management may allow reducing this percentage. Measures to enhance biocontrol and pollination are most efficient in simple, but not complex or fully cleared landscapes. Scattered semi-natural habitat across regions and countries maintains dissimilarity of communities (beta-diversity) and resulting functional redundancy

What is concluded, recommended? EU policy should tailor its agri-environmental schemes at the landscape scale to increase its effectiveness. Regulations to minimize agrochemical use need to be implemented to reduce hostility of cropland, thereby allowing spillover of functionally important biodiversity between local and landscape habitats. Such management should promote functional complementarity and insurance of ecosystem service delivery in times of environmental changes

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15.2 Biodiversity as Integral Part of Ecosystem Services

Agroecosystems are pivotal for the conservation of biodiversity in Europe. Biodiversity, in terms of species richness, trait diversity, and biotic interactions, affects ecosystem functions and their stability [3] by, e.g., promoting soil-supporting services, pollination, or biological pest control. In a political context, biodiversity conservation is often justified to ensure human well-being via the supply of ecosystem services. Notwithstanding, conserving a wide range of species, including those that are rare and endangered, may serve as an insurance and complementation strategy for safeguarding ecosystem functions under changing environmental conditions. Despite a huge body of experimental approaches [3], our knowledge about the relationship among biodiversity, ecosystem functions, and ecosystems services in agricultural landscapes is still fragmented and ambiguous. This relationship is most likely non-linear and depends upon interacting field and landscape-scale effects.

Pollination through insects and biological pest control are two ecologically and economically important agroecosystem services. Production of 75% of all major crops, especially fruits, nuts, and vegetables, benefits from or even relies on insect pollination. Wild pollinators such as bumblebees and solitary bees are usually the most effective pollinators for many economically important crops [4]. Pollination rates may increase with the number of species present in a site due to functional complementarity. However, the majority of pollination service is delivered through few common species [5]. Thus, the relationship between pollination rates and the number of species levels off at a particular point, which means that additional species only marginally increase the ecosystem service of interest. Under changing environmental conditions, however, these species may play an important role in maintaining the resilience of the ecosystem.

For pest control, both success and failure are possible with increasing numbers of natural enemies, but despite the context dependency, enemy diversity appears to generally increase biocontrol [6]. In a systematic re-analysis of aphid pest control across Europe and North America, Rusch et al. found consistent negative effect of landscape simplification on the level of natural pest control, despite interactions among enemies [7]. The average level of pest control was 46% lower in homogeneous landscapes dominated by cultivated land, as compared with more complex landscapes. There is thus a huge potential to support natural pest control through counteracting homogenization of farmland.

15.3 Landscape Heterogeneity Determines On-Farm Biodiversity and Ecosystem Services

The field and the landscape are intricately interconnected and constitute heterogeneity [8]. Both landscape compositional and configurational heterogeneity can affect biodiversity [9]. Landscape compositional heterogeneity increases with the diversity of habitat types, while landscape configurational heterogeneity increases with high amounts of edges and small crop fields. Ongoing research shows that increasing configurational heterogeneity at a landscape scale is at least as important for keeping biodiversity as the switch to organic farming [10]. Landscape composition and configuration at different spatial scales explained species richness of plants, bees, and butterflies [8, 11], and the presence of pest enemies in agricultural landscapes [12]. Many other ecological studies confirm that landscape characteristics influence biodiversity patterns at different spatial scales [8]. Moreover, heterogeneity can mitigate adverse effects of local land use intensification [13].

Semi-natural habitats and crop diversity are two important components of compositional and configurational heterogeneity in agricultural landscapes that affect biodiversity at the landscape scale [9]. Semi-natural habitats in agricultural landscapes play an important role as source habitats for many species, such as wild bees that pollinate crops [14] and natural enemies of pests [15]. However, the amount of semi-natural habitat is not the only factor that determines biodiversity at a landscape scale; the quality, in terms of resource availability, is also important to consider from an agroecological perspective. For example, conservation management of set-aside or fallows contributes to landscape complexity, but set-aside that is agronomically managed may not differ from cropland [16]. Enhancing functional biodiversity for pollination and biocontrol on a landscape scale requires a minimum of ca. 20% of semi-natural habitat, but improved cropland and fallow management may allow a reduction of this percentage [16].

The crop production area itself is often ignored and considered as undifferentiated matrix [9], although it greatly varies in its heterogeneity (e.g., field size or diversity of crops). In a recent study, we found that both configurational and compositional heterogeneity of the cropland influence predation rates on aphids, which indicates a higher success of pest control in more heterogeneous cropland (Fig. 15.1). Furthermore, fewer cereal aphids were present in farmland comprising spatial and temporal heterogeneity represented

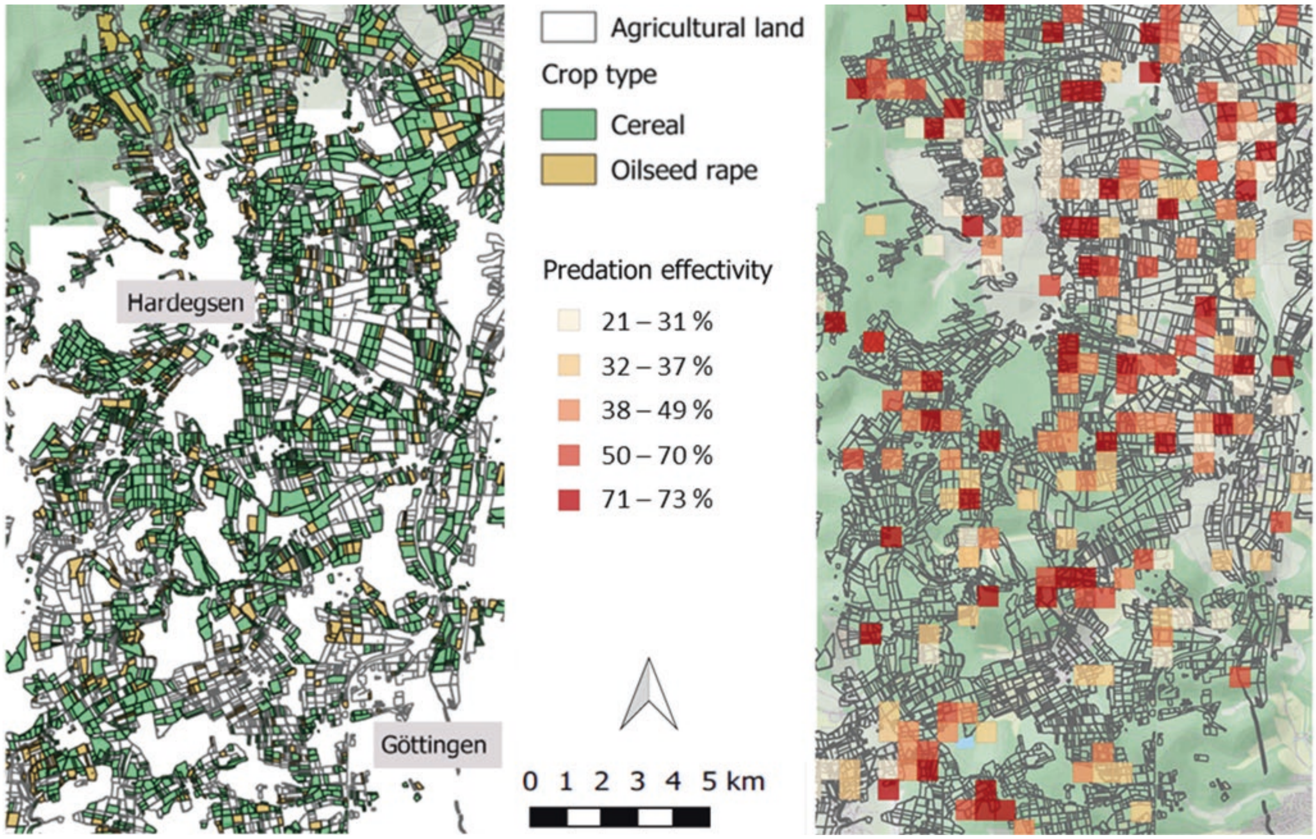

Fig. 15.1 Predicted predation effectivity in 52 agricultural landscapes in the Leinetal, Lower Saxony. The prediction is based on a comprehensive study on aphid predation rates in 104 cereal fields and 52 oilseed rape fields with different compositional and configurational heterogeneity of crops in the surrounding (Aliette Bosem-Baillod [Agroscope, Reckenholz] and Annika Hass [Agroecology, Georg-August University, Göttingen], unpubl. data). Information on the predation rates of aphid cards were collected during the summers of 2013 and 2014. Predation rate was used as a response variable in a generalized linear mixed model using the landscape as random effect and heterogeneity of the landscape as predictors. The results of this model were then extrapolated to the entire agricultural landscape in the Leinetal to predict pest control based on landscape heterogeneity

through small field sizes and high cover of field margins [17]. Consequently, ecological effectiveness through, e.g., pest control and pollination, interacts with heterogeneity of the landscape at local and landscape scales (Fig. 15.2) [18, 19]. However, measures to enhance biocontrol and pollination (e.g., by implementing field boundaries or hedges) are most efficient in simple landscapes rather than in complex or fully cleared landscapes [18]. We assume that this positive relationship between landscape complexity (i.e., the presence of semi-natural habitats) and the presence of natural enemies and pollinators may prove to be beneficial for crop yield (Fig. 15.2c).

Other ecosystem services may also be affected by landscape-scale characteristics and their interaction with local scale conditions [14]. Knowledge of such interacting effects can improve the planning of agriculture for specific ecosystem services. Mass flowering crops, for example, may serve as complementary resource for pollinators (Fig. 15.3) [20]. This complementarity effect, however, calls for assessments not only of local species' richness and related ecosystem services, but for a stronger focus on larger-scale species turnover (beta-diversity) among habitats, as well as total landscape diversity (gamma-diversity). Measures to increase semi-natural habitat and cropland heterogeneity across regions and countries promise to keep dissimilarity of communities (beta diversity). Higher beta-diversity, in turn, increases the likelihood of functional redundancy and may stabilize the capacity of a system to sustain its service provision.

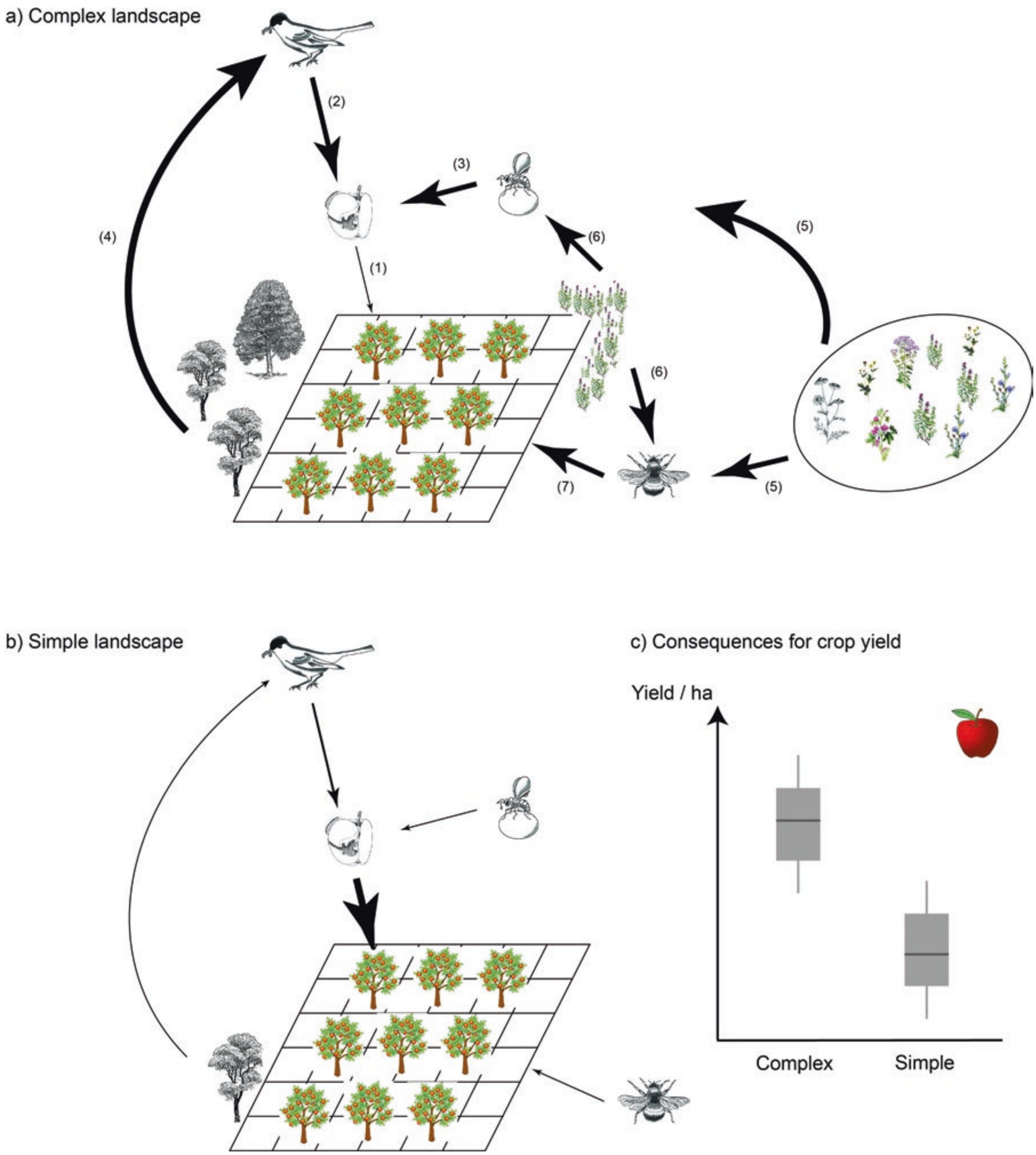

Fig. 15.2 Hypothesized consequences of landscape complexity for ecosystem service delivery and crop yield. **1**, Pest damage to apple fruits is often caused by the codling moth (*Cydia pomonella*). **2**, Insectivorous birds can suppress adult codling moths. **3**, Similarly, *Trichogramma* wasps are egg-parasitoids of codling moths, reducing codling moth damage in apple orchards when released. **4**, Trees and hedges in the landscape surroundings provide nesting habitat and food for insectivorous birds, increasing their biological control potential. **5**, Similarly, high-value habitats in the landscape surroundings as well as **6**, local establishment of flower strips benefits parasitoids as well as wild bee pollinators. **7**, Wild bees in particular are often more efficient pollinators of crops than commercial honeybees. While (**a**) complex landscapes provide ecosystem services, (**b**) landscape simplification results in losses of these services, which at the same time leads to higher pest outbreaks. Consequently, (**a**) complex landscapes should benefit crop yields at the farm-level by facilitating ecosystem service

15.4 Local Adaptation and Targeted Measures Required for Ecosystem Service Maintenance

The EU Common Agricultural Policy includes environmental measures that are intended to increase both biodiversity and ecosystem functions of the EU's farmland. As an example, management practices used in diversified farming systems result in more complex and heterogeneous agricultural landscapes and thereby have the potential to generate higher levels of biodiversity at the local scale. Flower strips represent such widely used agri-environment schemes, and the benefits related to pollination have the potential to outweigh the loss of area [21]. However, EU policies mainly target farm and field levels and usually disregard the landscape

Fig. 15.3 Pollination and natural pest control are two important ecosystem services in agricultural landscapes. (**a**) While the majority of pollination service is delivered through few common species (such as the honeybee *Apis mellifera*), rare pollinators are more efficient pollinators and may play an important role under changing environmental conditions. (**b**) The configuration and composition of cropland and the surrounding landscape influences the effectivity of natural pest control, as provided by parasitoids like parasitic wasps

context. The effectiveness of these measures, however, strongly depends on the landscape structure [22]. Thus, flower strips may or may not be beneficial for a specific conservation target. For example, perennial strips with few forbs may enhance the richness of soil-dwelling arthropod predators in the field margins, whereas nectar-rich flowers in an annual field strip may attract more pollinators. Hence, a set of measures need to be implemented to enhance a diversity of important services. These measures, moreover, need to fit the biophysical and socio-economic conditions of the region in which they are to be applied.

Heterogeneity of agricultural landscapes has often been found beneficial for biodiversity; however, diversification of cropland showed strongest impacts on biodiversity in simplified landscapes [22]. Moreover, not all functional groups of species may be similarly affected by variables at the field or at the landscape scales. For example, small solitary bees forage at small ranges, whereas large bumblebees (and honeybees) on large scales [23]. Generalist predators of cereal aphids, however, benefit from simplified cereal-dominated landscapes, while specialist enemies do not [24]. In contrast, earthworms and other organisms that increase soil quality and long-term soil fertility thrive best through on-site management, such as tilling and crop rotation. Rare or endangered species and species that fulfill keystone functions in an ecosystem may need specific and targeted conservation measures in order to support their contribution to ecosystem services.

15.5 Conclusion

Neither single agri-environment measure nor single conservation action targets the range of benefits that humans derive from agricultural land. Maintaining or restoring the ability of agricultural landscapes to provide various ecosystem services requires regionally adapted schemes, which are most effective if embedded at both the farm level and the landscape level. To ensure the provisioning of many different ecosystem services in a landscape, allocating priorities for smaller units of the landscape may prove helpful in navigating potential trade-offs between ecosystem services. One well-known trade-off between different ecosystem services is yield increase through intensification, on the one hand, and increases of semi-natural habitats for pollinators and natural pest enemies on the other hand. However, it is possible to balance these trade-offs through appropriate management. The implementation of flower strips at the local scale and increasing heterogeneity at the landscape scale are promising strategies to allow spillover of functionally important biodiversity between local and landscape habitats. In combination, these measures reduce the hostility of cropland and achieve synergy effects between facilitation of pollination and increased yield. Consequently, use of agrochemicals can be minimized, which decreases detrimental impacts on, for example, important soil functions. More research is needed to identify synergies between apparently conflicting ecosystem services, and this will inform the management of multifunctional landscapes. Moreover, farmland should be recognized as social-ecological systems that are strongly influenced both by the local society and by contextual legislation that spans the continuum from local to EU policies. Eventually, a comprehensive management system for the maintenance of multifunctional landscapes needs to tackle meaningful ecological scales and match various governance levels.

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16 Provisioning Ecosystem Services at Risk: Pollination Benefits and Pollination Dependency of Cropping Systems at the Global Scale

Sven Lautenbach

Which ecosystem services are addressed? Crop pollination

What is the research question addressed? What are hotspots of crop pollination benefits?

How dependent are different regions on crop pollination? How do pollination benefits and dependency on pollination differ across land systems?

Which method has been applied? Mapping of pollination benefits and vulnerability indicators based on a proxy-based approach. GIS overlay operations

What is the main result? Several hot spots of pollination benefits exist that deserve attention to ensure that the demand for the service by crop producers can be fulfilled by the ecosystems. The spatial pattern of the dependency of the cropping systems on pollination differs from the pattern of pollination benefits

What is concluded, recommended? Regions with high dependency on crop pollination are identified. In these regions, opportunities exist for win-win situations between nature conservation (to maintain or enhance crop pollinator habitats) and agriculture (pollination-dependent crops). Land system archetype specific policy recommendations are provided for these regions

16.1 Introduction

Pollination by animals is an important service for wild plant communities [1, 2] and agricultural ecosystems [3]. A large number of crops depend upon or substantially profit from pollination by domesticated honeybees as well as by wild pollinators such as wild bees, bumblebees, butterflies, hoverflies, and in some cases vertebrates such as bats and birds [4]. Although staple crops are not dependent on pollination by animals, more than a third of crop production does depend on pollinators and about 75% of all crop species profit to varying degrees from animal pollination, including most vegetables, fruits, and spices [4]. These pollination-dependent or pollination-profiting crops are also important for a number of nutrients essential for the human diet [5]. In addition to the effects of pollination on crop production, many other ecosystem services profit from or depend on pollination, including: the provisioning of ornamental species such as orchids; fodder crops; livestock provisioning by crops; food provisioning by unmanaged systems (such as wild berries); landscape aesthetics. Other cultural services also relate to pollination as intermediate ecosystem services. The level to which these services depend on pollination, however, have not been sufficiently quantified. The focus here, therefore, is on the benefits of pollination for crop production.

There are clear indications that wild and domestic pollinators are declining [1, 6–10] due to the loss and fragmentation of (semi-)natural habitats, the increasing use of pesticides, environmental pollution, the spread of pathogens, invasive species (alternative plant species, competitors, or enemies) and climate change [11, 12]. This leads to an ecosystem service risk due to potential declines in crop production, which has an impact on both farmers (loss of income and livelihood) and consumers (higher costs, opportunity costs of substituting pollination dependent food products). The focus of the analysis here is on the ecosystem service risk from the producers' perspective. The supply of sufficient pollination services cannot necessarily be managed by the individual farmer. Spatial planning therefore plays an essential role in providing suitable landscapes for pollinators and the sustainable supply of pollination services. Information on the spatial distribution of pollination benefits is thereby essential to estimate effects of land use decisions. A reliable quantification of the loss in crop yield in economic terms by pollinator decline provides a strong argument to protect

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landscape diversity in agro-ecosystems. This requires a spatially explicit analysis of the benefits of pollination as well as a spatial-explicit assessment of the vulnerability of the agricultural production system.

16.2 Data and Methods

To estimate the part of agricultural production that depends on pollination by animals, we used country-specific data from the Food and Agriculture Organization of the United Nations [13] on production prices and production quantities for crops that depend on or profit from pollination. For the analysis of the temporal development, data from 1993–2009 were used. Information from the World Bank [14] was used to correct the production prices for inflation, choosing 2009 as reference year. Production prices were further adjusted for differences in purchasing power among countries using the Penn World Table [15]. Pollination dependencies of crops were taken from Klein et al. [4]. Pollination benefits were estimated by multiplying corrected producer prices with production quantities and pollination dependencies of the crops.

We used the global maps on crop distribution of 60 pollination dependent or pollination profiting crops from Monfreda et al. [16] on a 5′ by 5′ (approx. 10 km by 10 km at the equator) latitude–longitude grid to derive a fine-resolution representation of pollination benefits. Sub-national data were only available for the year 2000, so the spatial representation of pollination benefits is limited to that year. Data provided consists of yield information in US dollars per hectare land on which the crop is cultivated, the produced quantity as well as the percentage of the cell which is used to cultivate the crop. National rather than regional averages for producer prices and purchasing power parities had to be used, because this information was not available on sub-national levels. This can be justified, as most nations show relatively uniform prices for agricultural products. Multiplying yield with adjusted producer prices and the area used to farm the crop leads to the average yield of the crop in US dollars per hectare for the total area of the raster cells. Since this leads to a common reference area for all crops, these derived values can be summed over all crops. See Lautenbach et al. [17] for more details.

To estimate the vulnerability of the land system towards a potential loss of pollinators, the pollination benefits of all crops in a raster cell were related to the total value of crop production of that cell. The resulting percentage describes how big the loss in production, and therefore income, would be for the farmers if pollinators were lost in that raster cell. Since information about the area harvested was not always available for all regions, the extent of the maps of pollination benefits does not overlap perfectly. The resulting maps of pollination benefits and of pollination dependency were overlaid with the land system archetypes by [18] to identify the cropping land systems that depend most strongly on pollination by animals. The land system archetypes represent recurring unique combinations of land-use intensity, environmental conditions, and socioeconomic factors that incorporate both drivers and impacts of land use.

16.3 Results and Discussion

16.3.1 Temporal Trend at the Global Scale

Aggregated global pollination benefits have increased from \$203 billion US in 1993 to \$361 billion US in 2009 (Fig. 16.1). However, pollination benefits decreased relative to the gross domestic product (GDP): Pollination benefits made up 0.7% of the global gross domestic product in 1993; this decreased to 0.5% in 2009 (Fig. 16.2). This decreasing importance of pollination went along with the decreasing percentage of the gross domestic product produced in agriculture: In 1993, 4.6% of the global gross domestic product was produced in agriculture, but only 3.1% in 2009. The relationship between pollination benefits and the total value in agriculture (including livestock) stayed relative constant between 1993 and 2009, with some fluctuations in between (Fig. 16.2).

16.3.2 Spatial Distribution of Pollination Benefits

The average pollination benefit per hectare cropland provides information to develop an estimate of the potential income to be lost owing to a pollinator shortage; this information could be used to estimate how much money rational and informed decision-makers might be willing to invest to prevent a loss of pollinators. The potential loss of benefits sets the upper limit of such an investment per year. The mean pollination benefit was 14.7 US dollars/ha with a standard deviation of 76.1 US dollars/ha. The values range from 0 to 3649 US dollars/ha with a strongly right-skewed distribution.

The distribution of pollination benefits across the globe was highly uneven (Fig. 16.3), sometimes even differing widely across agricultural regions in the same country (e.g., the USA). As expected, pollination benefits differed significantly across land system archetypes (Fig. 16.4). Land system archetypes without or with low importance of crop production naturally had low pollination benefits in crop production—other unquantified services might, however, benefit from pollination in these land systems. “Irrigated cropping systems” showed the highest median pollination benefits,

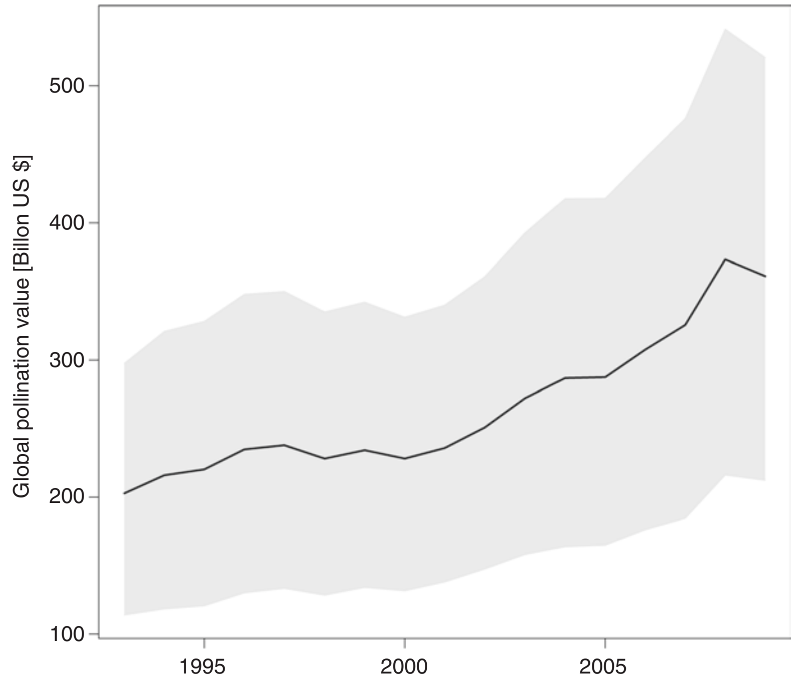

Fig. 16.1 Temporal trend of global pollination benefits. The pollination value has been expressed in US dollars that have been corrected for purchasing power parities and inflation. The grey regions indicate the uncertainty of the estimate due to the variance of the pollination dependencies of the different crops (*From* Lautenbach et al. [17]; *Creative Commons license*)

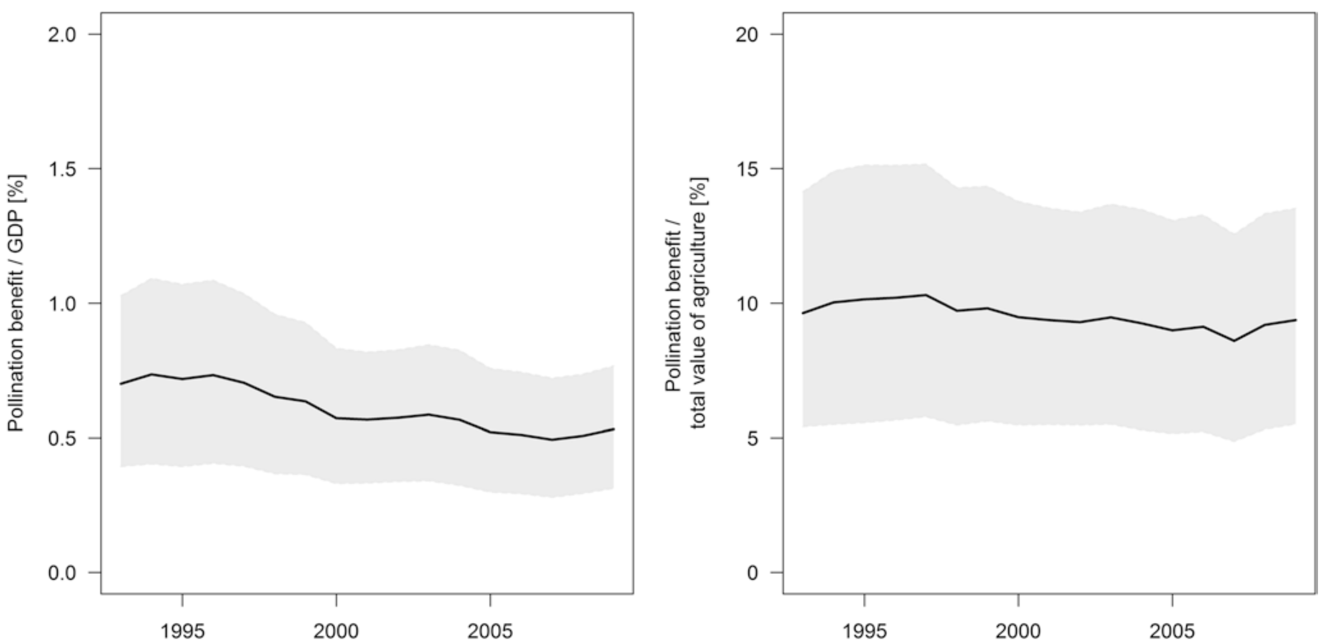

Fig. 16.2 Temporal trend of vulnerability indicators. The left panel shows the development of the part of the global gross domestic product (GDP) that is dependent on pollination while the right panel shows the part of the agricultural GDP dependent on pollination (*From* Lautenbach et al. [17]; *Creative Commons license*)

followed by "extensive cropping systems." "Intensive cropping systems" and "integrated cropping systems with rice yield gap" were characterized by much lower median pollination benefits, indicating that pollination-staple crops are dominant even if some regions are characterized by high pollination benefits.

16.3.3 Dependency of Agriculture on Pollination

Several regions are characterized by high average pollination benefits contrasted by a low dependency of agriculture on pollination (Fig. 16.5). Pollination-dependent crops make up

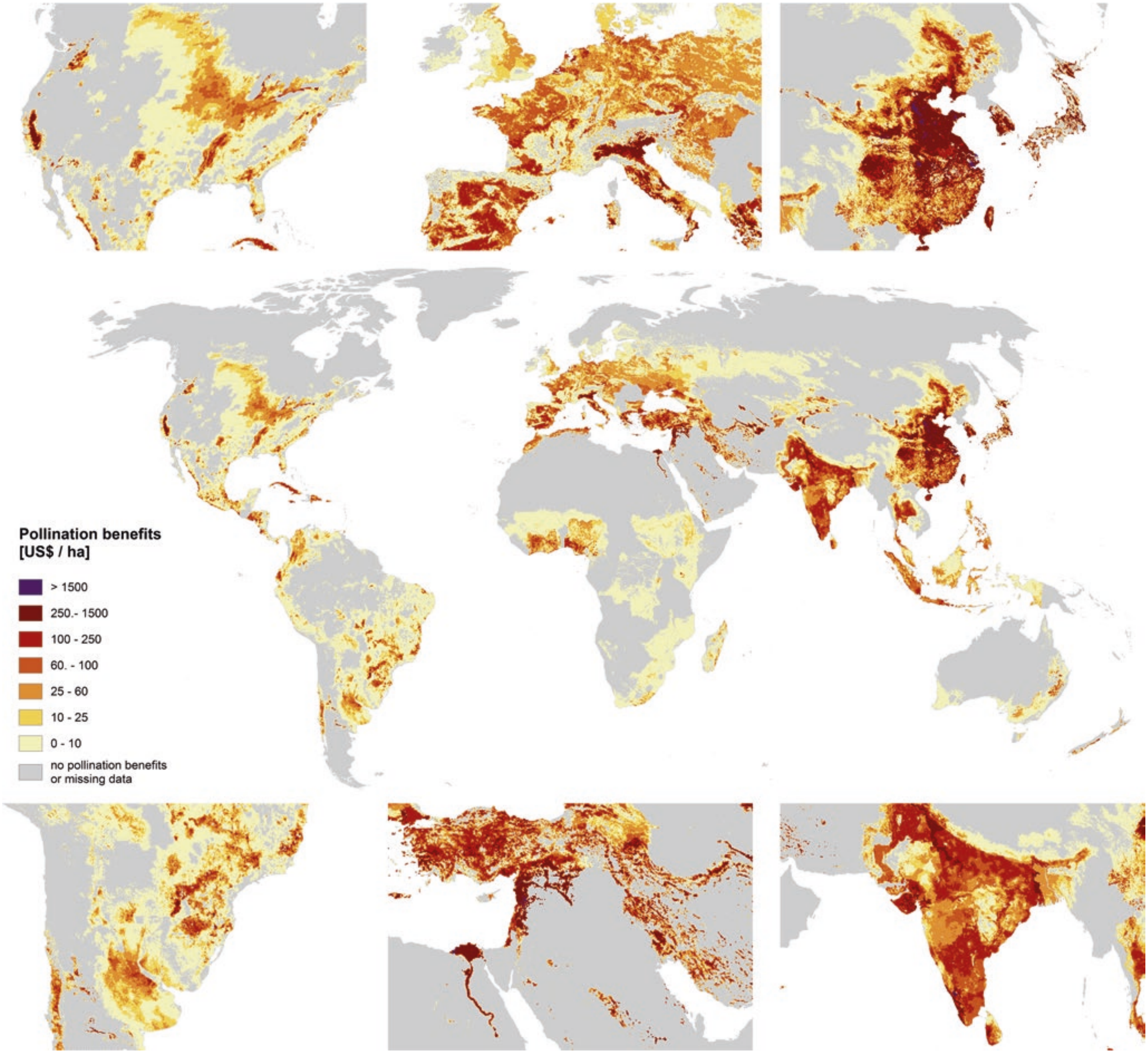

Fig. 16.3 Global map of pollination benefits. Values are given as US dollars per hectare for the year 2000. The values have been corrected for inflation (to the year 2009) as well as for purchasing power parities. Benefits are related to the total area of the raster cell (*From* Lautenbach et al. [17]; *Creative Commons license*)

only a small amount of the total agricultural production in these regions, even if the absolute value of pollination benefits is high. Examples include the agricultural systems along the Nile, in the western part of Turkey, in parts of Spain, Italy, and Germany, in parts of Chile, in the costal parts of Nigeria, in Java, in the southern and eastern parts of India, and in parts of the US states Illinois, Iowa, Indiana, Arkansas, and Mississippi. Impacts of a pollinator shortage on the local economy and local food supply in places such as these would therefore likely be not critical, even if individual farmers might face significant losses.

On the other hand, there are regions that stand out for having relatively low average pollination benefits, but an agricultural system with high pollination-dependency. Such regions include: parts of Ethiopia, the southeastern costal region of South Africa, Western New Guinea, Sicily, and parts of the southern USA. The local economies of these regions would face severe impacts from a shortage in pollination supply, which would presumably also have a significant affect on local food supply. It is therefore essential to maintain supply of pollination services by protecting or increasing wild pollinator habitat.

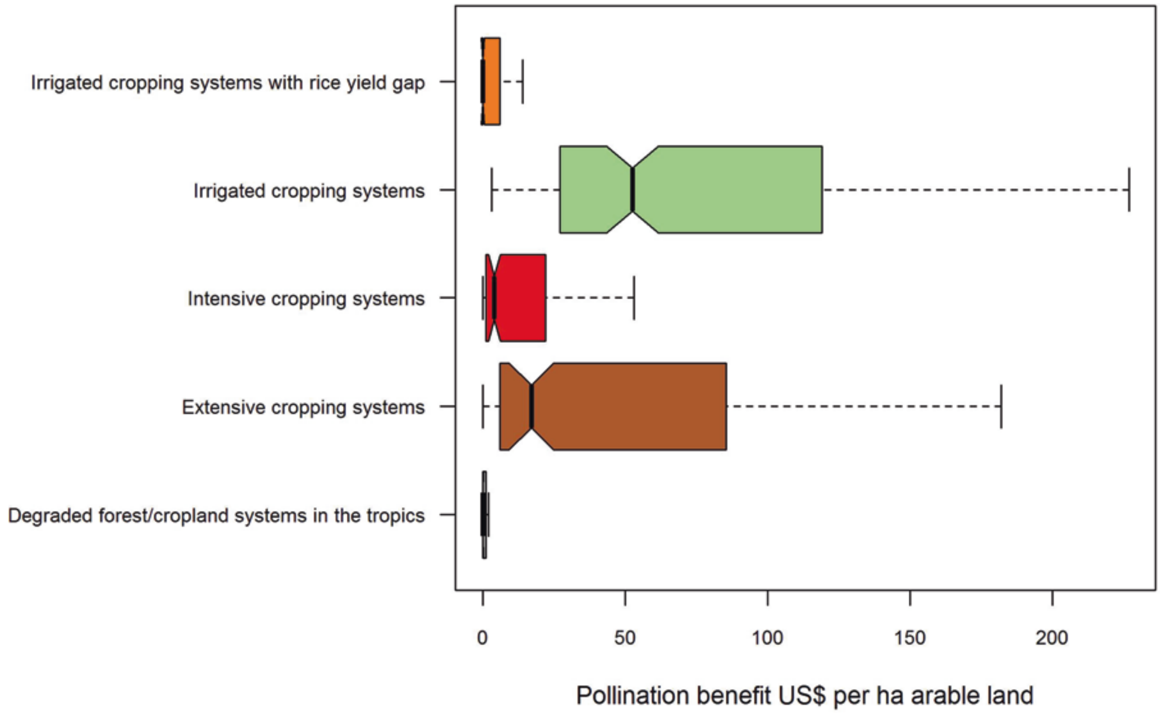

Fig. 16.4 Distribution of pollination benefits across crop land system archetypes as defined by Václavík et al. [18] Values outside of the 1.5 time the interquartile range are not shown

Several other regions are characterized by high average pollination benefits and high pollination dependency of the agriculture. These include: Cote d'Ivoire, parts of the Brazilian states Espírito Santo, Bahia, Rondônia, and Mato Grosso. Agriculture in these regions is focused on cash crops such as coffee or cacao, so a strong decrease in pollination services would hit the local economy significantly but would not affect local food supply directly. In the northeastern part of China, regions with high average pollination benefits and with high pollination-dependency of the agricultural system appear alternately with regions of low pollination-dependency. Given that spatial heterogeneity, it is unlikely that a shortage in pollination supply would have severe consequences for local food supply, even if effects on the local economy would be substantial.

Pollination-dependency differed across the crop-related land system archetypes (Fig. 16.6). However, differences were much smaller compared to pollination benefits (Fig. 16.4). Median dependency was highest in the "irrigated cropping systems," followed by the "intensive cropping systems," "degraded forest/cropland systems in the tropics," and "extensive cropping systems." Pollination-dependency in the "irrigated cropping systems with rice yield gap" was lowest.

16.4 Policy Implications

A pollination supply sufficient to allow the production of pollination-dependent crops is important from the perspective of both the local farmer and the regional economy. Food security and social justice are of high importance for high pollination-dependent regions in low-income countries. The benefits of pollination services provide an estimate of the expenditure that might be acceptable in a region to protect pollinator habitats. However, this does not indicate who should pay the costs of maintaining pollination service. High benefits do not necessarily lead to a protection of wild pollinators, as the example of the USA shows: Pollination supply is here so far managed by a pollination business that transports managed pollinators across the country to provide pollination to farmers. However, the high importance of wild pollinators for crop production identified, at least, for Europe [19, 20], and the increased risk of disease spread in pollinator populations by the current practice in the USA [12], indicate that this might not be a sustainable strategy. Measures to maintain pollination services by wild pollinators are therefore of great importance. These measures highlight the importance of the multi-functionality of landscapes. The benefits by wild pollinators are realized only if the distance

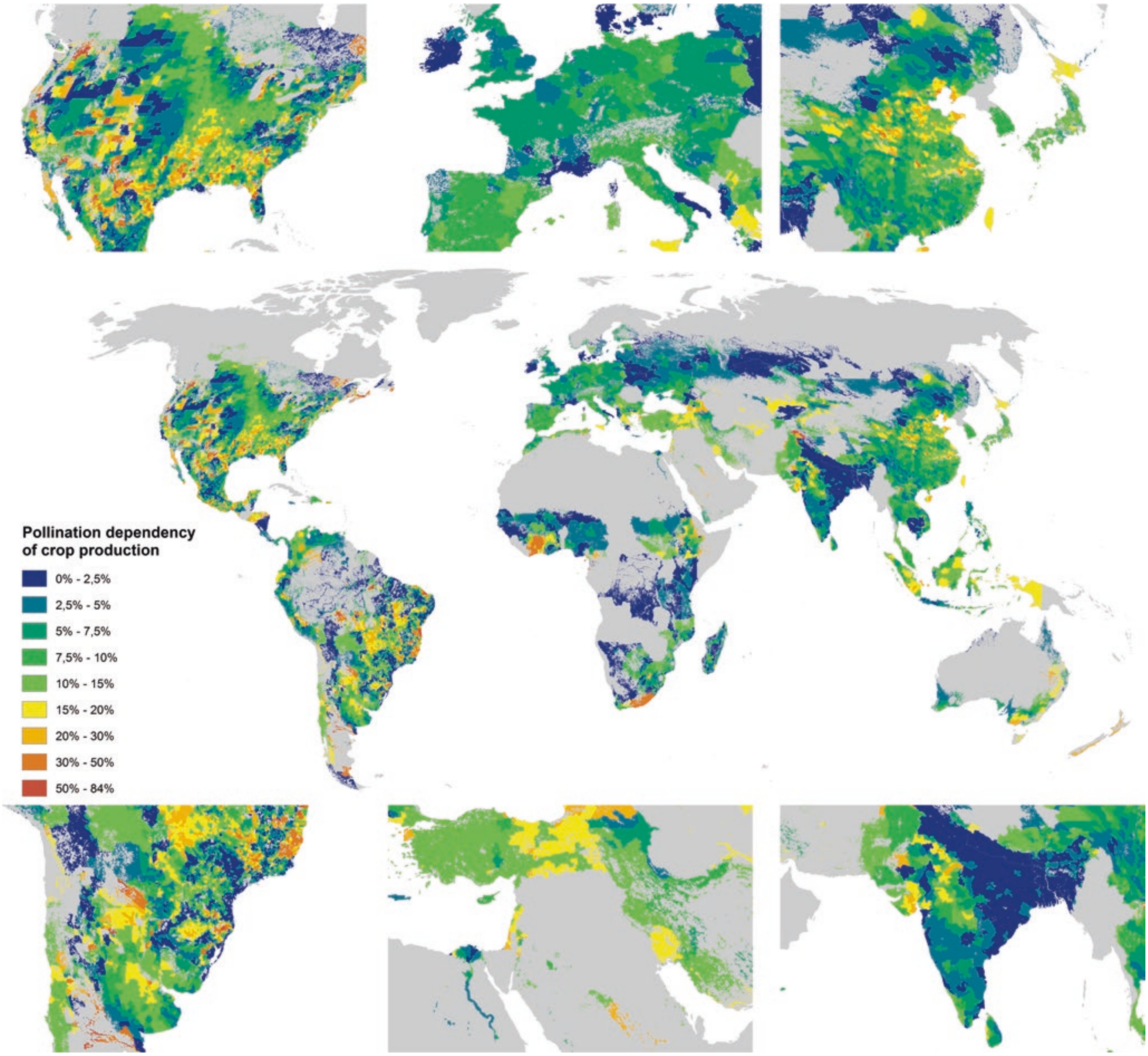

Fig. 16.5 Pollination dependency of crop production. The map shows the part of the value of crop production that depends on pollination by animals for the year 2000

between pollinator habitats and crops to be pollinated is not too large. The quantification of pollination benefits puts an important incentive on the conservation of semi-natural or natural patches or green linear elements in agricultural areas. These areas are under high pressure because their benefits are less visible than the costs of unproductive land. Any cost-benefit analysis should therefore include pollination benefits from a societal perspective, together with other services such as landscape aesthetics and scenic beauty.

Concrete decisions about how to best ensure a sufficient pollination service depend on regional circumstances, such as the importance of green linear elements to provide pollinator habitat [21]. Land system archetypes provide means to characterize potential measurements at the global scale. Measures such as organic farming practices and planting of flower strips that provide floral resources tend to have their greatest efficiency in landscapes dominated by intensive agriculture [10], offering few floral resources, as characterized by the land system archetypes "intensive cropping systems" and "irrigated cropping systems." Regions in these archetypes will also profit most from the use of managed pollinators such as honeybees, in addition to wild pollinators that have been shown to lead to highest yields [22]. Use of pesticides in these intensively managed land systems is a

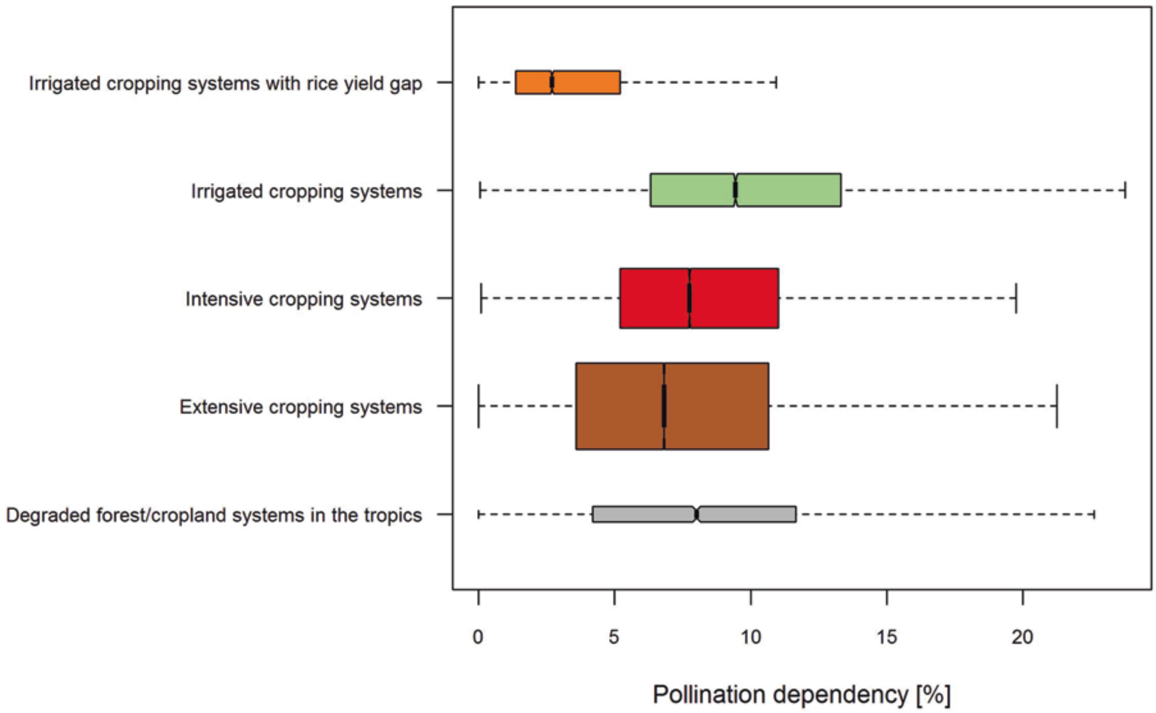

Fig. 16.6 Distribution of pollination dependency across crop land system archetypes as defined by Václavík et al. [18] Values outside of the 1.5 time the interquartile range are not shown

potential threat to wild and managed pollinators. Therefore, pollinators will profit from a reduction of pesticide use and an increase in ecological farming practices such as biological pest control. Pollinators in "degraded forest/cropland systems in the tropics" might be protected most efficiently by the conservation of remaining forest habitats or management actions to enhance degraded forest patches, together with the strengthening of diversified farming systems such as the Central American milpa systems. The use of managed bee populations in "extensive cropping systems" is presumably not very efficient—the protection of natural and semi-natural land, together with the strengthening of diversified farming systems, is the most promising approach here.

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17 Minimising Risks of Global Change by Enhancing Resilience of Pollinators in Agricultural Systems

Oliver Schweiger, Markus Franzén, Mark Frenzel, Paul Galpern, Jeremy Kerr, Alexandra Papanikolaou, and Pierre Rasmont

Which ecosystem services are addressed? Pollination

What is the research question addressed? How does climate change impact pollinators, and can land management be used to increase their resilience?

Which method has been applied? Analysing observed range shifts; species distribution models and future scenario projections; generalised linear mixed effects modelling of monitoring data

What is the main result? Current climate change has already led to range contractions of pollinators, while it is projected to have an even more severe impact in the future. However, proper land management can increase resilience of pollinator communities

What is concluded, recommended? Effects of increasing the amounts of semi-natural areas are positive and twofold: they directly increase the richness and abundance of pollinators while simultaneously making them more resilient against other threats of global change such as climate warming. However, the intended level of 7% of Ecological Focus Areas by the EU Common Agricultural Policy falls too short; at least ca. 17% are needed

17.1 Importance of Pollinators

Pollination of wild and crop plants by animal pollinators is a key ecosystem service that is important to human welfare. However, the societal benefits and dependencies of pollination vary in different times and places (*see* Chap. 16). About 90% of wild plant species depend at least partially on animal pollination [1] and about 70% of the most important global crops rely to some extent on animal pollination [2]. These crops constitute 35% of global food production, and the worldwide economic value of pollination is estimated to amount to €153 billion per year [3]. In addition, most essential nutrients in human diets, like vitamin C, are provided by plants that depend entirely or substantially on pollinators [4].

Although pollinators belong to many different animal groups, insects are usually considered to be the most important pollinators [5]. Managed pollinators, such as honeybees or some bumblebees, might be less sensitive to threats of global change. It has been shown, however, that wild bees make a critical contribution to the yield of crops and are usually more efficient than managed pollinators in agricultural landscapes [6].

17.2 Multiple Threats to Wild Pollinators

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) reports current declines of wild pollinators in abundance, occurrence, and diversity [7, 8], and such declines have been attributed to multiple drivers of change. Habitat loss and degradation along with intensive agricultural practices are among the most important factors, but climate change, spread of diseases, and alien species are also impacting pollinators [9]. Most importantly, these drivers do not act in isolation but may interact to reinforce, or alternatively to weaken, the response of wild pollinators to a particular driver depending on the severity of another one [10, 11]. Such interactive effects could also be leveraged to increase the resilience of pollinator communities. Here, we highlight how climate change can impact pollinators and how new land management practices can increase the resilience of local wild bee communities to the impacts of global warming.

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17.3 Impact of Climate Change on the Distribution of Pollinators

17.3.1 Current Climate Change

The Intergovernmental Panel on Climate Change (IPCC) reports that climate change has already caused shifts in the range of many species groups [12, 13]. For most taxa, range expansions towards the poles are a common response to warming [14], while range contractions at the equatorward range margins are rare [15]. Bumblebees, however, one of the most important pollinator groups, show the opposite pattern: Climate warming relates to severe range contractions in the south, while species generally have not expanded northwards (Fig. 17.1) [16].

These alarming results were revealed by a comprehensive cross-continental study in which we tracked long-term observations (110 years) across Europe and North America on a database of approximately 423,000 georeferenced observations of 67 bumblebee species. On this basis, we tested for climate change-related range shifts in bumblebee species across the full extents of their latitudinal and thermal limits. We found cross-continentally consistent range losses from southern range limits while, most of the species failed to track climate warming at their northern margins (Fig. 17.2).

17.3.2 Future Climate Change

For pollinators as important as bumblebees, the strong retractions at the equatorward range margins, combined with their failure to track climate warming with northwards range expansions, have implications for species distribution as climate change proceeds. For example, assessments of climate change risks (in the sense of impacting ecosystem state and condition; *see* Chap. 1) make assumptions about species' ability to track changing climates. When our findings are

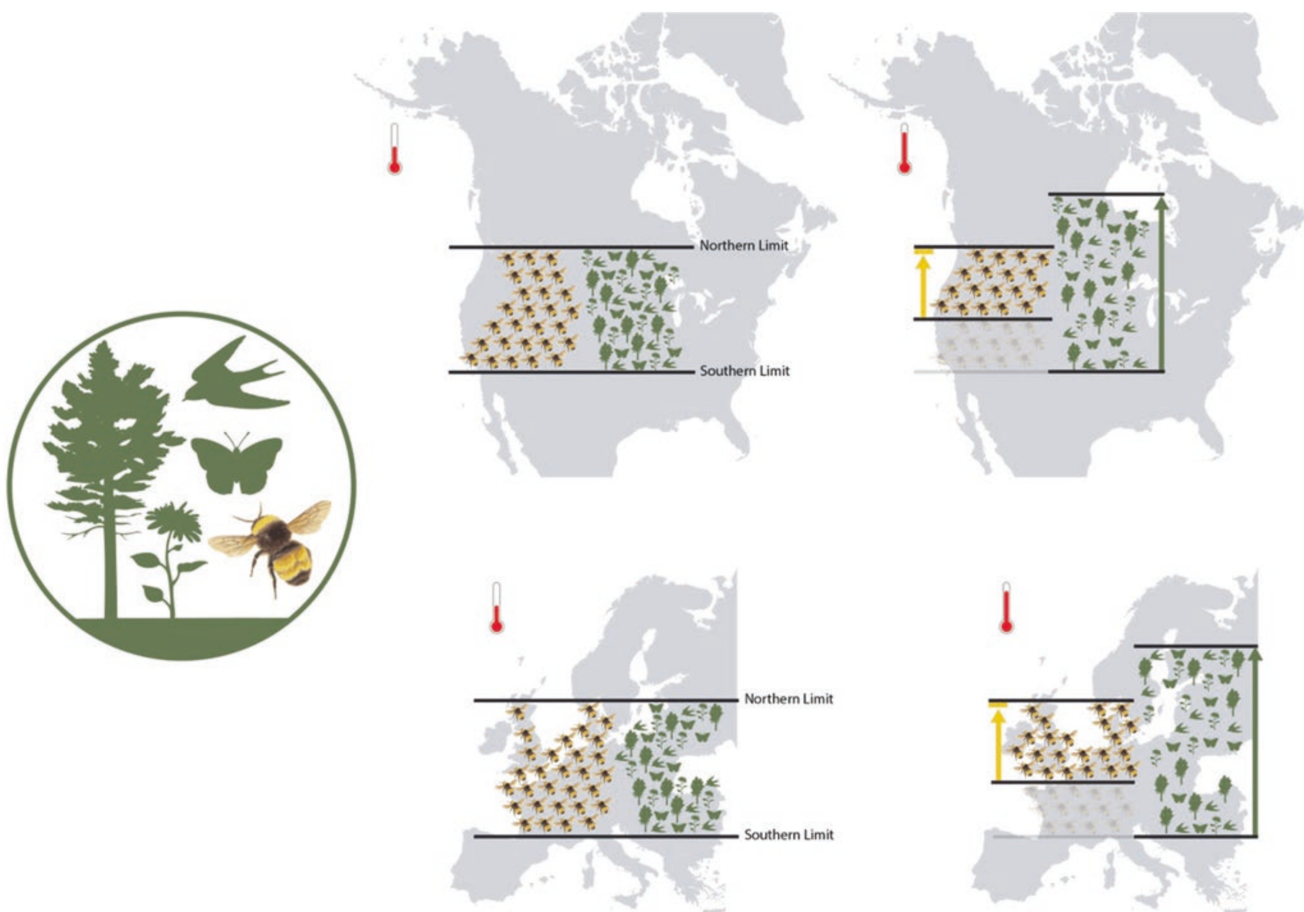

Fig. 17.1 Schematic comparison of observed responses of different species groups to climate warming in North America and Europe. While the majority of species groups (green symbols) respond to climate warming primarily with poleward range expansions and with minimal response at the equatorward margins, bumblebees (yellow symbols) react with strong range contractions at the equatorward margin but fail to expand polewards. *Image courtesy of* Ann Sanderson

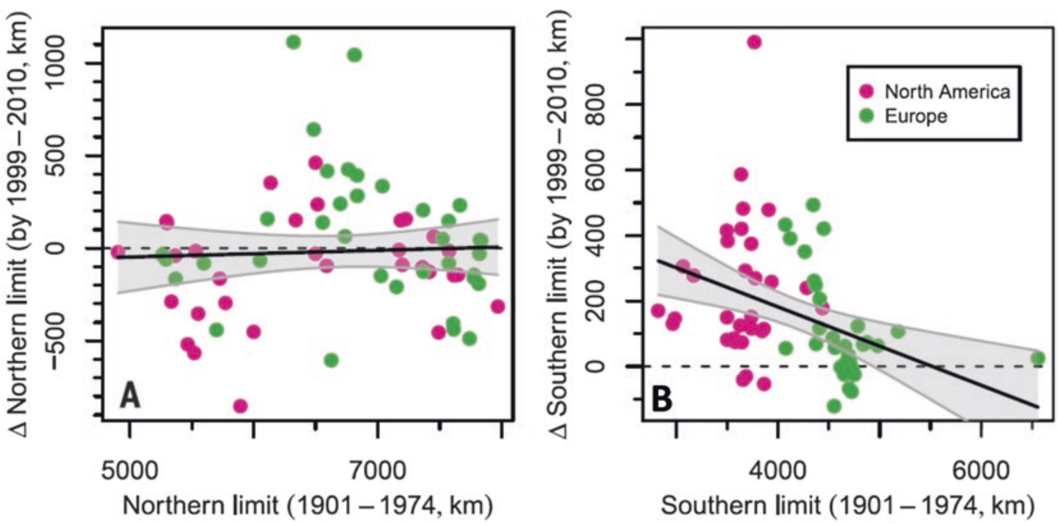

Fig. 17.2 Climate change responses of 67 bumblebee species across full latitudinal limits in Europe and North America. The y-axis shows changes in latitudinal range limits at the northern (**a**) and southern (**b**) range margin between the historical (1901–1974) and the current (1999–2010) distribution of bumblebee species. (**a**) Positive values indicate range expansions from species' historical northern limits. (**b**) Positive values indicate range losses from species' southern limits. The grey area indicates 95% confidence bands for regression models of observed changes in range limits vs. historical range limits (*From* Kerr et al. [16]; *with permission*)

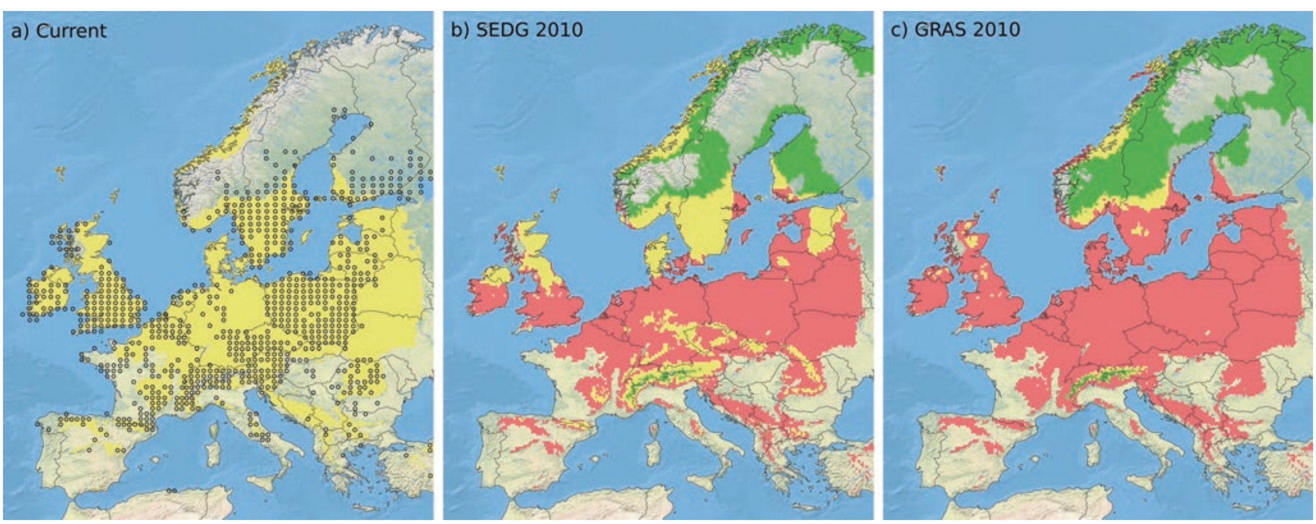

Fig. 17.3 Projected changes in suitable climatic conditions for the bumblebee *Bombus ruderarius*. (**a**) Current (open circles) and modelled (yellow area) distribution. (**b** and **c**) projected changes under 3 °C warming scenario (SEDG) and a 5.6 °C warming scenario (GRAS) for 2100. Red—losses; yellow—remaining suitable conditions; green—new areas with suitable conditions but only reachable under the assumption of full ability to track climate change (*From* Rasmont et al. [17]; *with permission*)

incorporated in such assessments, the consequences for bumblebees are severe. Based on relevant climate data and 300,435 records of all 69 European bumblebee species between 1970 and 2000, we developed species distribution models and projected the changes in suitable climatic conditions for these species under future climate change scenarios [17]. These projections relied on two alternative assumptions—full ability and no ability to track warming—leading to considerable differences when estimating future risks of climate change. Strong future retractions at the southern margins in combination with the failure to keep track with climate warming at the northern margins suggest a grim fate for many bumblebees (Fig. 17.3). Comparing full ability vs. no ability to track climate change, the proportion of bumblebee species losing more than 70% of climatically suitable area increased from 5% to 18%, 18% to 56%, or 65% to 95% under warming scenarios of 3.0 °C, 4.7 °C, and 5.6 °C, respectively, by the year 2100 (Fig. 17.4).

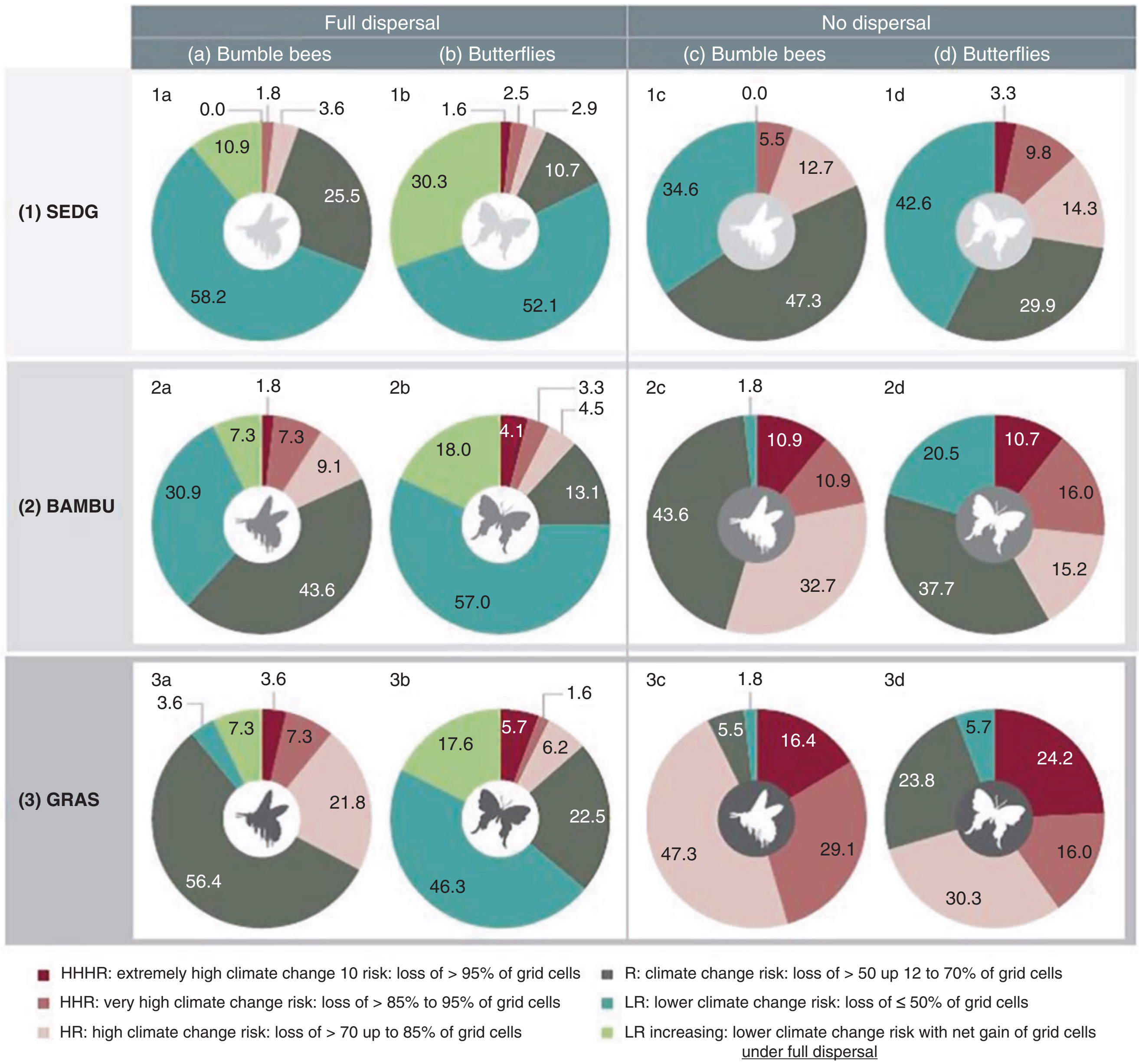

Fig. 17.4 Projected risks for European bumblebees and butterflies under different scenarios of climate change, assuming full (left) or no (right) dispersal. Risk corresponds to impacts on ecosystem state and condition following the framework of Chap. 1. Scenarios: SEDG, sustainable Europe development goal (equivalent to the IPCC B1 scenario with a mean expected temperature increase of 3.0 ° C in Europe by 2100); BAMBU, business-as-might-be-usual (equivalent to the IPCC A2 scenario with an expected temperature increase of 4.7 °C in Europe by 2100); and GRAS, growth applied strategy (equivalent to the IPCC A1FI climate change scenario with a mean expected temperature increase of 5.6 °C in Europe by 2100) (*From* IPBES [8]; *with permission*)

17.4 Land Management Can Increase Resilience of Pollinator Communities in Agricultural Landscapes

Sound management strategies are needed to minimise risks of climate change for pollinators and pollination services. To compensate for potential failures to track changing climates at the poleward range margins, managed relocation may be necessary [16], but only after careful study of risks and benefits of such actions [18].

In addition to management actions at the northern range margins, increasing resilience of pollinator populations at the southern range margins is also required. We conducted a study based on data using 95 local wild bee communities collected over three years at six intervals of two weeks in six agricultural landscapes differing in the amount of agri-

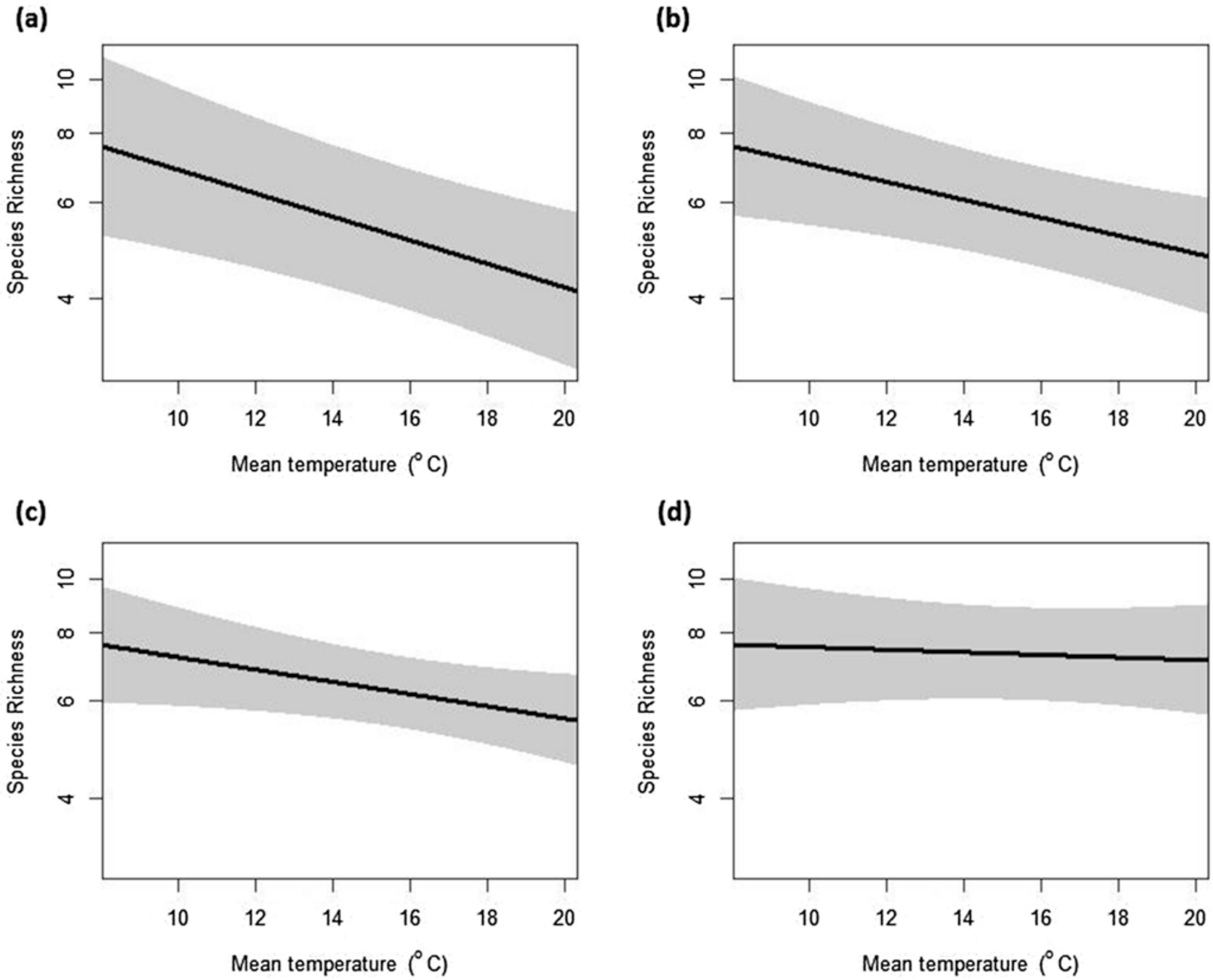

Fig. 17.5 Interactive effect of temperature and amount of semi-natural habitat on bee species richness. The effect of temperature increase on species richness is displayed for four different levels of percentage of semi-natural areas covering the entire range of the six study sites: (**a**) 2%; (**b**) 6%; (**c**) 10%; (**d**) 17%. Grey bands indicate 95% confidence intervals (*From* Papanikolaou et al. [20]; *with permission*)

cultural and semi-natural habitats. Study sites were located in Central Germany and are part of the TERENO project (Terrestrial Environmental Observatories; www.tereno.net) [19]. We found positive effects of semi-natural area and negative effects of warmer temperatures on both richness and abundance of bee species. More surprisingly, we found an interaction between temperature and the amount of semi-natural habitats in terms of species' survival prospects (Fig. 17.5) [20]. Translating these results of overly hot weather to increasing temperatures caused by climate change, this means that higher amounts of semi-natural habitats can effectively buffer negative effects of warming, and thus increase pollinator resilience amid changing climates.

17.5 Policy Implications

Given the importance of wild pollinators to the economy and human nutrition, it is essential to minimise risks confronting pollination services through measures that increase pollinator resilience. In combination with land-use intensification, climate change could drastically shrink the global distribution and abundance of pollinator species. However, increasing the amount of semi-natural habitats in agricultural landscapes might be an efficient instrument to enhance pollinator resilience against climate warming. The potential benefit of such an instrument is high, since large agricultural areas in Europe are characterised by extremely low amounts of semi-natural areas. For instance, in about 45% of agricul-

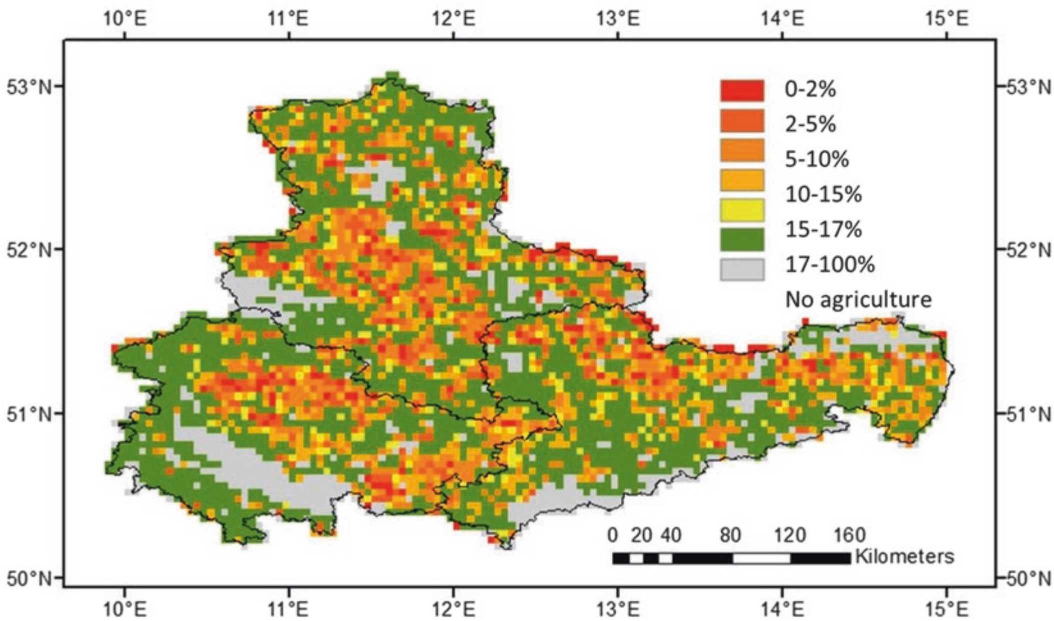

Fig. 17.6 Percentage of semi-natural area in agricultural landscapes as indicator for pollinator resilience against climate warming in Central Germany. According to findings of Papanikolaou et al. [20], a threshold of 17% of semi-natural area is assumed under which local pollinator communities are increasingly less resilient against negative effects of climate warming

tural landscapes in Central Germany, the amount of semi-natural habitat is less than 17% (Fig. 17.6), a critical threshold below which species face sharply elevated local extinction risks [20]. Although the actual numbers were assessed by a study in Central Germany, and they may vary across geographic regions, the main principle is likely to be applicable across temperate agroecosystems. The positive effects of higher amounts of semi-natural areas are twofold: they directly increase the richness and abundance of pollinators while simultaneously making them more resilient against other threats, such as global climate warming. Ensuring the resilience of pollinators under climate change is yet another reason to accelerate efforts to design agricultural landscapes for pollination services, and to implement practices that optimize the amount and distribution of semi-natural areas. In this sense, some regulations of the EU Common Agricultural Policy (CAP) and the EU strategy for Green Infrastructure point in the right direction. Article 46 of the EU Regulation 1307/2013 [21] focuses on the greening of agricultural areas by designating Ecological Focus Areas (EFA). These EFAs should cover 5% by 2015 and 7% shortly thereafter. However, the study by Papanikolaou et al. [20] indicates that this threshold falls too short for pollinators. Increasing the targets for semi-natural area to at least 17% is likely needed to increase pollinator resilience as these species confront the impacts of rapid global change.

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18 Drivers of Risks for Biodiversity and Ecosystem Services: Biogas Plants Development in Germany

Martin Dotzauer, Jaqueline Daniel-Gromke, and Daniela Thrän

Renewable, ecosystem-based energy provision through energy crops is an increasingly valued ecosystem service that benefits society. Biogas plants deliver a significant value to the German renewable electricity production of 30 TWh and 18% of the total annual renewable production. Today, there are about 8000 biogas plants (excluding biogas plants for upgrading and grid injection of biomethane) in Germany, with an installed electrical capacity of 4.5 GW [1]. These can provide synergies, but also trade-offs, with ecosystem services, and can pose risks to biodiversity, soil fertility, and the cleanliness of groundwater. These risks can be described specifically as follows. (1) Biodiversity can be threatened by demand for additional land for field crops. This may lead to intensifying agricultural production, with an increasing need for arable land at the expense of extensively farmed or near-natural areas [2]. (2) Soil fertility might suffer from intensive energy crop production owing to, for example, poor crop rotations with high shares of maize. This may lead to a higher risk of soil erosion due to the wide spaces between rows that are characteristic of maize fields [3]. (3) Groundwater quality is the main issue at risk, as it may be affected by leaching of highly mobile nutrients, mainly nitrogen, if locally bioenergy triggers an overrun in nitrogen balances [4]. Groundwater quality, however, can also be improved, if energy crop production and residuals are managed properly according to just-in-time nutrient management practices within farming cultivation schedules [5].

These explicit interactions between energy production, land use, and biosphere can be identified as the main drivers of risks to biodiversity and ecosystem services from biogas production. The main factor for the development of today's plant portfolio in the power sector in Germany is the Renewable Energy Law (EEG), which was first enacted in 2000 and has been adapted several times. Up to 2003, just a few small plants were installed, with less than 300 MW capacity (see Fig. 18.1), that were run mostly as co-fermentation plants with manure and other agricultural or agro-industrial residues.

Which ecosystem services are addressed? Agrobiodiversity by farming systems for energy crops
Groundwater production by potential nutrient leaching
Soils fertility by soil erosion

What is the research question addressed? How does biogas production push risk factors for the provision of ecosystem services? Where are interdependencies to comparable forms of agricultural production?

Which method has been applied? A spatial distribution analysis for assessing risks to ecosystem services details specific densities of livestock farming, biogas production, and the share of maize within crop rotations on farmland at a district level

What is the main result? Risk to ecosystem service caused by bioenergy, particularly biogas production, is mainly linked to energy crop production and the application of digestate. Both aspects must be considered with respect to the local conditions, foremost the amount of cattle farming in a region, since biogas plants and cow sheds act very similarly in terms of the required feedstock and incurring residues

What is concluded, recommended? To cope with the potential risks for ecosystem services associated with biogas production, it is recommended that biogas and livestock farming should be jointly considered with respect to their need for feedstocks and the local capacity for sustainable application of manure as well as digestate. This can be achieved by a joint calculation of nutrient balances for both production systems

When the EEG was amended in 2004 and 2009, additional financial bonuses for energy crops were established. These incentives led to a significant increase in annual installations of plants and capacity up to 2012, when the EEG was

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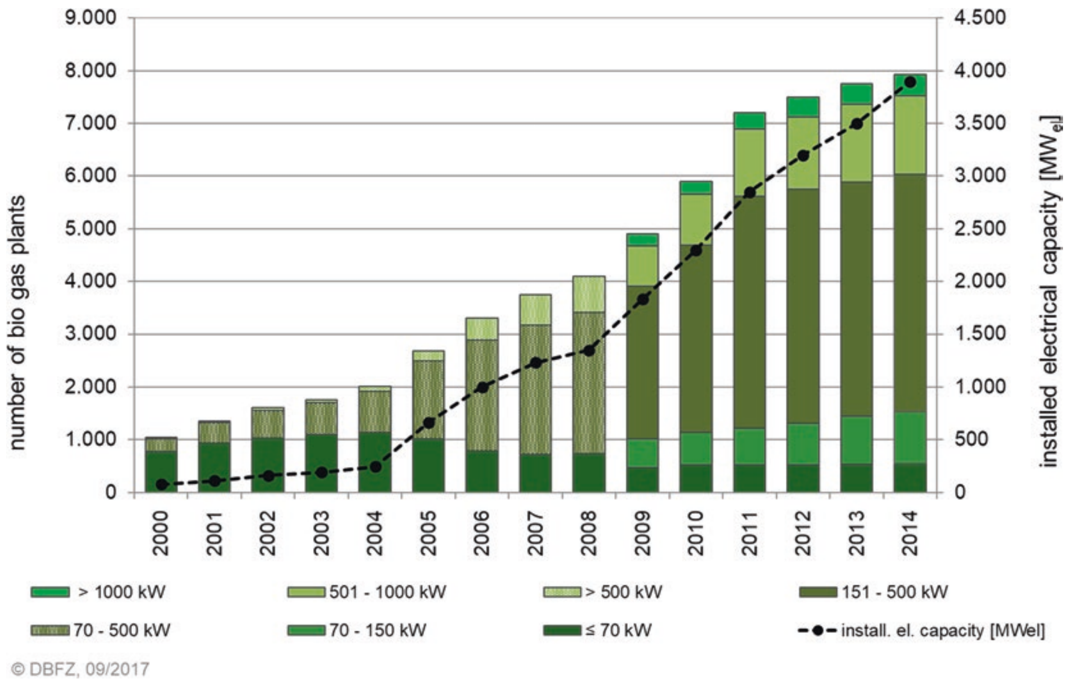

Fig. 18.1 Development of installed capacity and number of biogas plants in Germany (*Adapted from* Scheftelowitz et al. [1]; *with permission*)

revised again. In fact, today most input substrates to biogas plants consist of energy crops, in terms of their contribution to energy supply [1]. The latest amendment of the EEG, in 2014, resulted in a significant lowering of the feed-in-tariffs by, among other factors, abolishing the substrate bonus for energy crops and the biogas upgrading bonus for new plants. Against the background of the changing legal framework and conditions, increases in biogas generation capacity after 2014 will be mainly an effect of repowering, a switch to flexible plant operation, as well as new small manure plants and biogas plants in the waste sector [1]. Newly installed capacity capacities are predominantly driven by small manure plants (<75 kW_{el}) and biowaste fermentation facilities.

Biogas plants can place biodiversity and associated ecosystem services at risk owing to energy crop rotation, and application of digestate on fields risks the leaching of nutrients if the material is applied inappropriately. Leached nutrients can pollute the groundwater when nutrients exceed the legal threshold for drinking water. Energy crop cultivation in Germany is like regular agricultural production of food and feed, and thus puts pressure on biodiversity by the use of pesticides and herbicides in monoculture plant stocks, which leaves just a few ecological niches for the remaining species. The plants themselves can also lead to ecological disturbance; for example, if accidental leakage occurs, digestate could enter waterways and lead to an expanded impact [6].

In Germany in 2015, 11.8% of total arable land (1.4 million hectare of 11.8 million hectare) [7]) was used for the production of biogas substrates. These 1.4 million hectare were used for cultivation of different energy crops, of which 0.9 million hectare are planted with maize, to satisfy the need of biogas plants [8]. Maize, therefore, as the dominant energy crop, covers 7.6% of total arable land. The production of the energy crop production for biogas by itself functions as a driver of biodiversity risk, but the distribution of the production is also relevant, as the spatial distribution of biogas plants doesn't follow a homogeneous pattern. In fact, a lot of installed capacity is concentrated in some hotspot areas (Fig. 18.2) that also correspond to structures of the whole utilization chain, i.e., energy crop production, access roads, biogas plants, and the application of slurry, which can become problematic if there are local concentrations.

The operation of biogas plants, which primarily use energy crops, is comparable to that of cattle farming for milk production. These operations use similar feedstock and produce slurry that has comparable effects on ecosystem services, with the exception that livestock breeding may also include the use of pharmaceuticals and imported concentrated feed [9]. In a rough estimation, one livestock unit (LU) is approximately equivalent to 1 kW of installed rated biogas capacity in terms of demand for land (input and output materials). Therefore, the following observation focuses on the spatial distribution of livestock farming and maize production, as these two types of production are the principal drivers for silage maize cultivation for feedstock for both biogas and animal production [10].

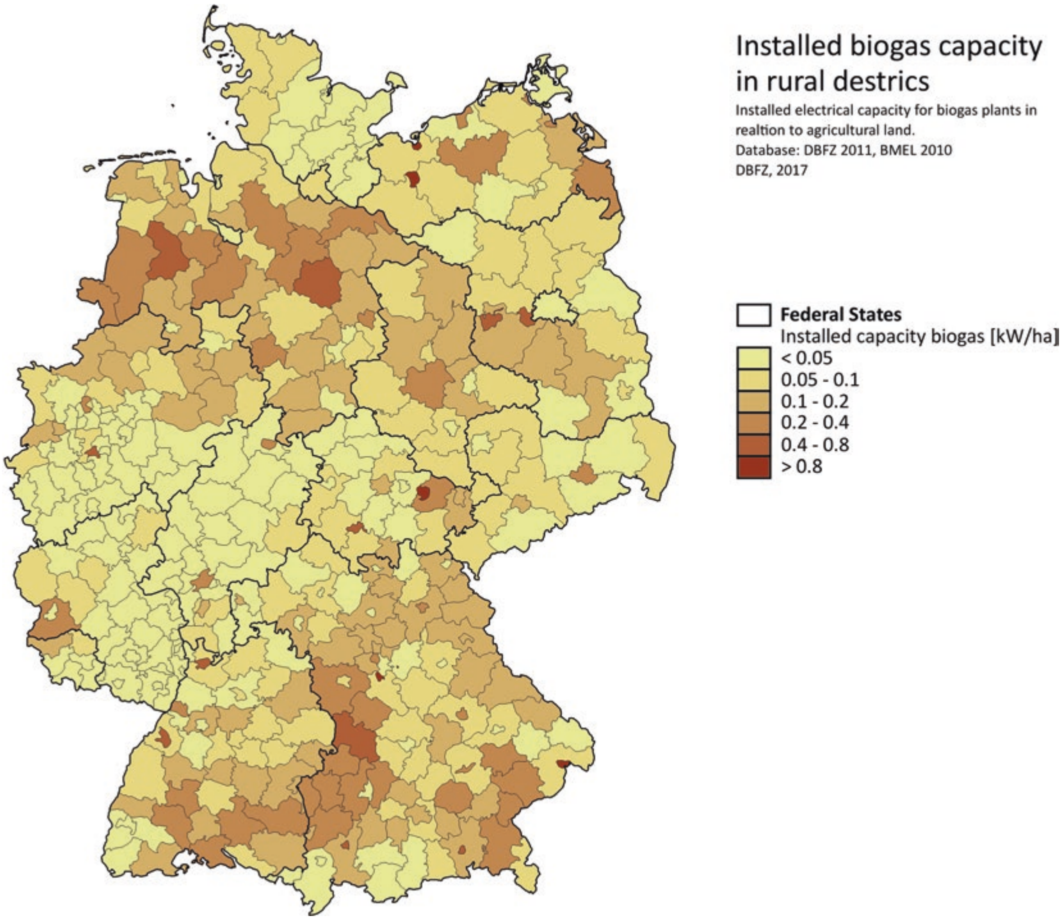

Fig. 18.2 Specific power density of biogas plants in Germany in 2011 in relation to the amount of cultivable land per rural district

The overall concentration of livestock farming in Germany shows an inhomogeneous spatial distribution (Fig. 18.3) with a concentration in the northwest and some southeastern regions of Germany. Except for several small local spots in the eastern states, the highest concentration can be found in the states of Schleswig Holstein, parts of Lower Saxony, Rhineland-Palatinate, Baden-Württemberg, and Bavaria. Depending on natural landscape conditions, limits for maximum livestock units per hectare are recommended to secure the ability of agricultural ecosystems to deliver an essential input for feedstock and an intake capacity for residues, e.g., nutrients. Considering these requirements based on the German Federal Soil Protection Act and Ordinance (Bundes-Bodenschutzgesetz [11]) as well as the fundamental Council Directive 91/676/EEC (protection of waters against pollution caused by nitrates from agricultural sources), basic rules for good agricultural practices are defined [12]. Although it is not reasonable to determine a single limit for all regions, high concentrations of livestock density can lead to environmental impacts, especially by nutrient pollution [13].

Maize as a major feedstock for biogas as well as animal production can generate an impact on biodiversity if its regional share in crop rotations becomes dominant. Its ability to be grown without crop rotation over a long period also determines the potential threat to biodiversity [14]. In summary, the distribution patterns of biogas plants and livestock production show that certain areas are characterized by high stocking densities and high crop ratios for the related feedstock, such as maize (Fig. 18.4). The concentration of biogas plants in these regions intensifies the existing problems by raising the need for further crop production and simultaneously the amount of slurry, namely manure and digestate, that may affect soils, ground water and biodiversity. To counter that risk, a frequently discussed solution could be to handle digestate like manure in terms of its nutritional content, especially nitrogen [15]. For farmers who participate in direct payments of the European agricultural aid, annual nutritional balance of arable land is mandatory. By including digestate as a highly relevant nitrogen source in these balance calculations, very high values of balance surplus of nitrogen in hotspot regions could be avoided.

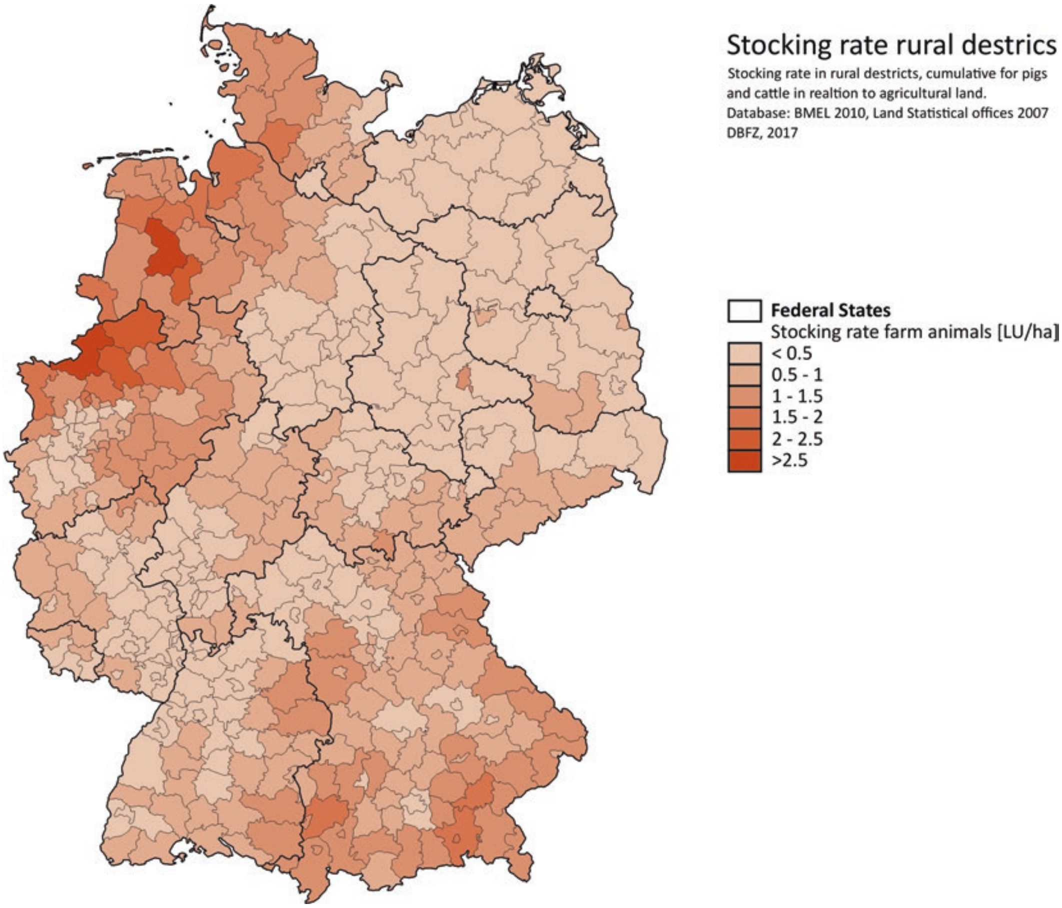

Fig. 18.3 Specific density of livestock units in relation to the amount of cultivable land per rural district

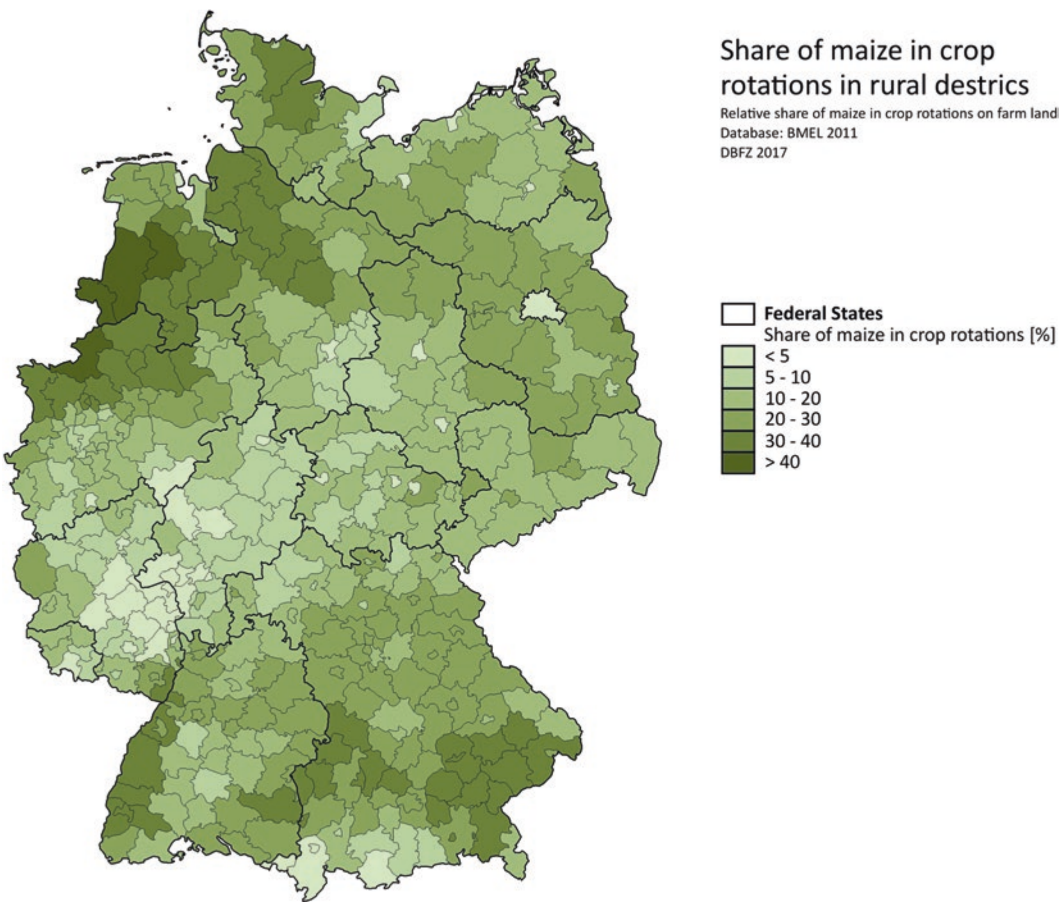

Fig. 18.4 Share of maize on crop rotation on arable land per rural district

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19 European Energy Governance Landscapes: Energy-Related Pressures on Ecosystem Services

Sebastian Strunz, Erik Gawel, and Alexandra Purkus

19.1 Trade-Offs Between Energy Provision and (Other) Ecosystem Services

Energy provision affects ecosystems and their services for humans. This applies to *all* forms of energy, whether fossil resource-based or renewable, ecosystem-based or not. Energy infrastructures are important stressors of ecosystems (e.g., by cutting into habitats—first order risk) and may increase the riskiness of ecosystem services provision (second order risk). Yet overall, the magnitude of potential second order risks from energy provision remains understudied, and concerted research efforts to fill this gap are necessary [1]. That said, each form of energy entails specific risks: a system built on nuclear and wind yields other risks than one built on hydro, biomass, and solar energy. Some forms of energy, such as bioenergy, constitute a provisioning ecosystem service by themselves—and hence potential negative effects from biomass production on regulating and supporting ecosystem services are actually trade-offs between different services within a given ecosystem. By comparison, other renewables are not classified as ecosystem services, as they depend primarily on abiotic parameters. Fossil energy provision induces climate change and other externalities, thereby straining the global provision of provisioning and regulating services in a variety of local ecosystems.

In other words, extent and geographical distribution of both first and second order risks depend on the given technological portfolio and its spatial allocation. These parameters are driven by energy policy decisions, which, in turn, depend (amongst other factors) on the society's risk preferences. The latter may differ both between and within countries. For instance, some perceive nuclear energy as a risk worth taking, while others consider it as an untenable risk (see the "Eurobarometer," a survey of technology perceptions [2]). In consequence, current low-carbon transitions may lead to geographically very heterogeneous distributions of policies and accordingly differing distributions of ecosystem service risks.

Which ecosystem services are addressed? Bioenergy as provisioning ecosystem service

Risks for regulating and cultural services that stem from different technologies (e.g., wind power affecting recreational landscapes)

What is the research question addressed? What is the spatial distribution of nuclear energy, renewables (as a general category), and bioenergy around Europe—in terms of both policies and technology shares?

Which method has been applied? Mapping of energy policies and technology shares

What is the main result? Policy approaches, technology shares and, in consequence, ecosystem service risks vary across the continent

What is concluded, recommended? There exists no generally optimal response strategy on the European level. Rather, the appropriate mix of responses (avoidance, trade-off management, adaptation, and transformation) will differ according to the respective national risk preferences

Against this background, this chapter sketches the variety of energy policy approaches within the European Union and the resulting heterogeneity of energy transitions. What is, specifically, the spatial distribution of nuclear energy, renewables (as a general category), and bioenergy around Europe—in terms of both policies and technology shares? In other words, this chapter maps the energy-related drivers of increased risks, but not the ecosystem service risks themselves.

19.2 Nuclear Power

First, the use of nuclear power yields important risks for human safety as well as biodiversity and ecosystem services [3]. After the 1986 catastrophe in Chernobyl, detrimental

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impacts on biodiversity included morphological, physiological, and genetic disorders as well as reduced population sizes across major taxonomic groups [4]. The 2011 reactor meltdown in Fukushima yielded similar effects [5]. Biodiversity attributes, in turn, are most often positively correlated with the sustained provision of ecosystem services, which derive from a variety of interactions in complex systems [6]. Thus, nuclear power constitutes a first order risk. Beyond these indirect effects on ecosystem functioning, however, nuclear accidents yield direct risks for provisional ecosystem services. This may involve contamination of agricultural land or highly elevated radiation levels in game, mushrooms, and wild berries. While there is no established scientific evidence on the magnitude and the duration of these effects, this may be due to a "don't ask, don't tell" bias in the literature ([7], p. 8). Hence, both systematization of existing work (partly gray literature) and a coherent protocol for further research on the risks of nuclear power for provisioning ecosystem services are called for.

As evident from Fig. 19.1, the use of nuclear power and the associated risks are unevenly distributed across Europe. Both policies (Fig. 19.1b) and the share of nuclear in overall electricity consumption (Fig. 19.1a) vary across the continent. A range of countries neither currently relies on nuclear power nor intends to do so in the future (e.g., Italy). Some countries still use nuclear power but have committed themselves to phase it out (e.g., Germany, Belgium, Switzerland). At the same time, other countries maintain a large fleet of nuclear reactors and will continue to do so for a foreseeable time, even if the share of nuclear in overall electricity consumption shall decrease (e.g., France). Furthermore, some countries are planning to increase their nuclear fleet. For instance, the UK government recently succeeded in having the EU Commission condone its plan to build a new nuclear reactor at Hinkley Point, which is to be financed via a "nuclear feed-in premium" (a so-called Contract for Difference which entails that electricity generators receive the difference between a set "strike price" and average market prices). While Poland does not yet have any nuclear power plants, it aims at building a reactor. Overall, therefore, policy approaches towards nuclear as well as the importance of nuclear for energy provision, measured by its share in electricity consumption, are highly diverse. This is one important factor in the EU Member States' reluctance to hand anymore decision-making power regarding energy matters to the EU Commission [8].

19.3 Renewables

Second, due to their lower energy density, renewable energy sources need more land area than fossil resource-based energy, so the transition to renewables entails extensive land-use change. Impacts vary greatly between technologies (the

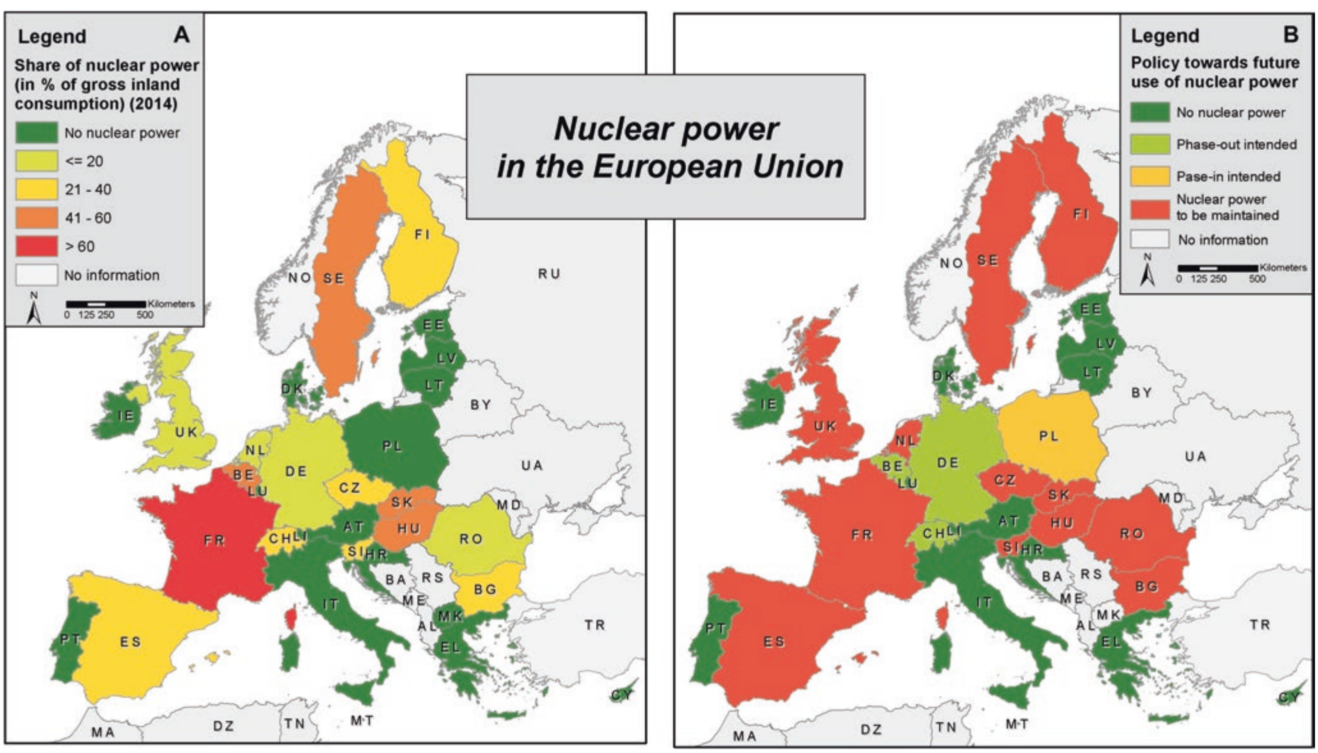

Fig. 19.1 Share of nuclear power in electricity consumption (2014, panel **a**) and policy approaches towards the future use of nuclear power (panel **b**) [18]

specific case of biomass is discussed in more detail in the next section): for example, rooftop photovoltaic (PV) modules occupy no new land, while solar PV parks on the ground are only compatible with limited biodiversity. Thus, while some of this land-use change is certainly to be welcomed (e.g., no more open-pit coal mines), the deployment of renewables may also diminish the provision of some ecosystem services. This may be due to either direct land-use trade-offs—such as large-scale PV installations replacing agriculture as a form of provisioning ecosystem service—or more indirect effects from the increased pressure of renewables on ecosystem functioning (e.g., a wind park may be a hazard to birds and bats unless siting and operation decisions are adapted to the ecological circumstances). Knowledge gaps regarding the specific magnitude of ecosystem service risks [1, 8] notwithstanding, it should be acknowledged that the increasing deployment of renewables may entail both first and second order risks to ecosystems and their services. These risks materialize locally following the specific technologies used and their geographical location. The latter, in turn, is influenced by the policies that are employed to promote renewables.

Against this background, Fig. 19.2a maps the distribution of support policies for renewables within the EU. Traditionally, the two main approaches for supporting renewables were feed-in tariffs and quota schemes: the former directly remunerates each kWh of electricity produced from renewables, while the latter requires energy providers to include a specified quota of renewable energy in their portfolios. Feed-in tariffs are generally thought to benefit small-scale installations (such as PV panels on individual homes), while quota schemes benefit large-scale installations (such as wind parks). While feed-in-tariffs were clearly the most prominent support instrument during the 2000s [9], there is a recent trend towards hybrid support instruments, which are based on combinations of feed-in tariffs, feed-in premiums (a mark-up on the market price for each kWh of renewable electricity) and/or tenders where specified amounts of renewable electricity provision are ‘auctioned off’ (the idea being that the bidding procedure incentivizes cost reductions and yields cost-efficient remuneration of renewables). Tender schemes currently receive particular attention as the EU Commission strongly pushes for them as default support instrument so as to further decrease the cost of the deployment of renewables. As can be seen from Fig. 19.2, these efforts already inform national approaches towards renewables.

The issue of support instruments and their geographical distribution is important for ecosystem service risks because decentralized, small-scale approaches may better reflect local knowledge on how energy installations may interfere with local ecosystem services [10]. Conversely, if renewable installations were exclusively allocated through an EU-wide tendering scheme, this would lead to centralized, large-scale production of renewable energy (e.g., photovoltaic installations in Southern Europe), which might transpose into hotspots of ecosystem service risks. In other words, a one-size-fits-all support for renewables would be counterproductive. Yet there are constant critiques leveled against national support instruments for renewables: these critiques narrowly focus on the generation costs of energy and disregard potential ecosystem service risks from centralized, large-scale production of renewables [11]. In sum, therefore, the heterogeneous distribution of policy instruments, expansion targets, and renewables’ share in gross final energy consumption (see Fig. 19.2b, c) reflects the heterogeneity of risk preferences and locally grounded knowledge.

19.4 Biomass

Third, among renewable energy sources, the use of biomass as a provisioning ecosystem service entails specific sustainability challenges. An increase in bioenergy demand increases pressures on agricultural and forestry ecosystems, and measures such as agricultural intensification or an expansion of the agriculturally used area may negatively affect other ecosystem services and lead to a decline in ecosystem resilience [12]. Figure 19.3 maps the relevance of biomass use for energy consumption in the EU member states (encompassing the use of solid biofuels, biogas, charcoal, liquid biofuels, but not including municipal wastes whose use does not increase ecosystem services risks). To illustrate the increase in land use pressures and associated first and second order risks, the expansion of biomass use over time is depicted for selected member states. For the EU as a whole, gross inland consumption of biomass more than doubled between 2000 and 2014, from 237,755.4 TJ in 2000 to 501,779.9 TJ in 2014 [13].

Meanwhile, impacts on biodiversity, soils, water quality and availability depend on the bioenergy pathway in question (e.g., wood-based, energy crop-based, waste and residual-based) as well as on local and regional circumstances [14, 15]. Depending on prior land uses, biomass cultivation for energetic purposes can negatively, but also positively, impact the provision of ecosystem services (e.g., if energy crop cultivation leads to an increase in agricultural biodiversity compared to feed or food crop monocultures). At the same time, land-use changes asso-

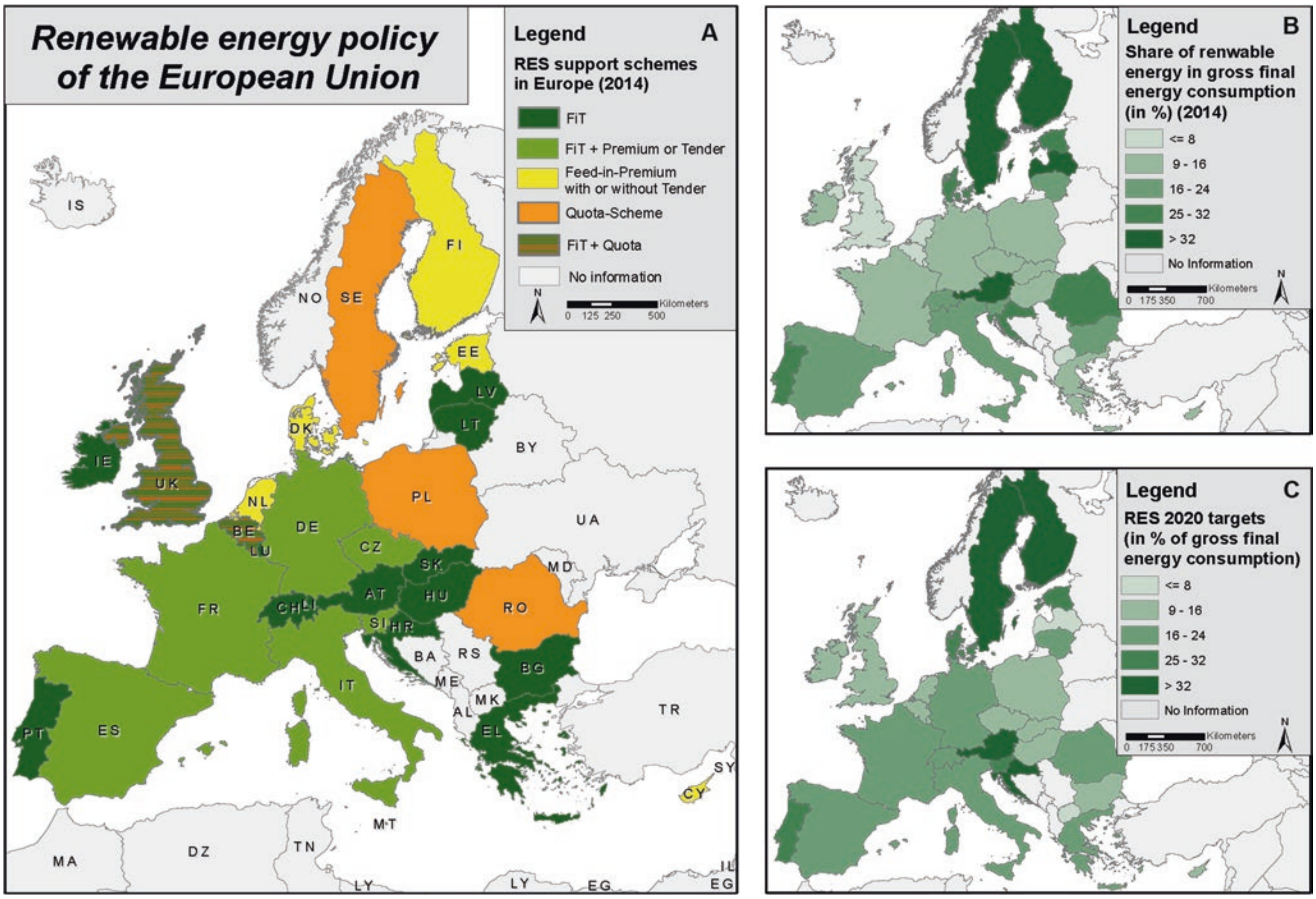

Fig. 19.2 Support schemes for electricity from renewable energy sources (RES) in 2014: Feed-in tariffs (FiT), Premium schemes, Tenders, Quota schemes and their various combinations (panel **a**) [13]. RES shares in gross final energy consumption (2014, panel **b**) and RES targets for 2020 (panel **c**) [18]

ciated with biomass production have an important impact on the greenhouse gas balances of bioenergy pathways [16, 17]. This emphasizes the importance of designing institutional framework conditions that safeguard against negative impacts of an increased biomass demand on ecosystem services (e.g., through stringent environmental legislation), and provide incentives for fostering environmental co-benefits (e.g., through sustainability criteria in demand-sided deployment support instruments, or as part of agri-environment measures).

19.5 Summary

The transition to low-carbon energy systems also changes energy-related pressures on ecosystems. Yet, both first and second order risks will not be affected in a uniform way because land use patterns within a sustainable energy system differ according to the specific technologies employed and their geographical distribution. Whether nuclear power should play a part in future low-carbon energy systems is a matter of particular contention. This contribution demonstrates the rather heterogeneous picture of transition pathways currently emerging as a result in Europe: policy approaches and, in consequence, ecosystem service risks vary across the continent. While this chapter does not map specific ecosystem services in detail, it is complementary to those analyses of this volume that quantify specific impacts, since understanding the drivers is an indispensable component of any comprehensive risk assessment. The overall implication of this contribution is that there can be no generally optimal response strategy on the European level. Rather, the appropriate mix of responses (avoidance, trade-off management, adaptation, and transformation) will differ according to the respective national risk preferences.

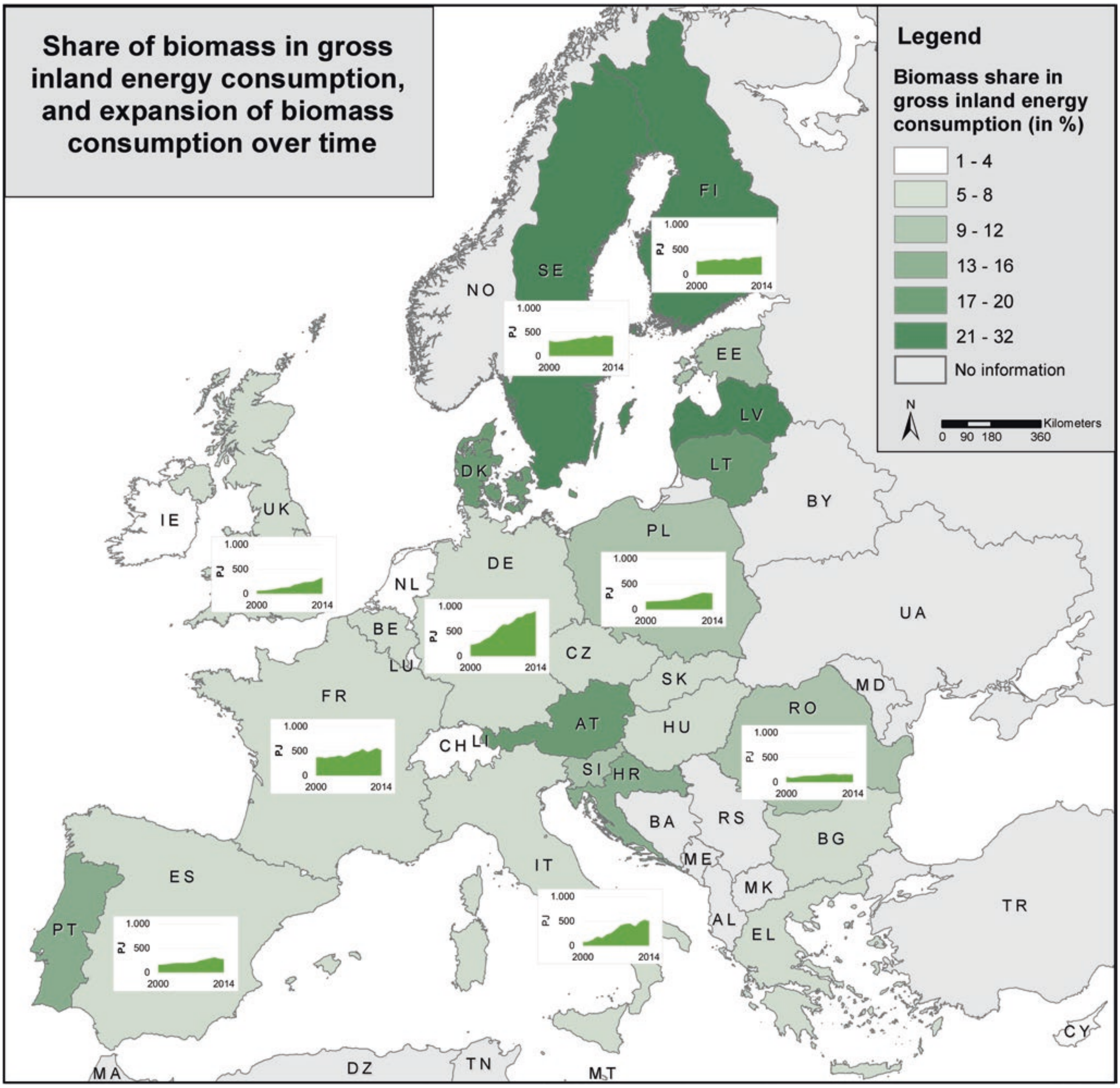

Fig. 19.3 Share of Biomass in gross inland energy consumption (2014): "Biomass" encompasses solid biofuels, biogas, charcoal, liquid biofuels; municipal wastes are excluded. The expansion of biomass consumption (measured in Petajoule) is presented for exemplary countries only. Switzerland: data refers to gross energy consumption [13]

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20 Wind Power Deployment as a Stressor for Ecosystem Services: A Comparative Case Study from Germany and Sweden

Thomas Lauf, Kristina Ek, Erik Gawel, Paul Lehmann, and Patrik Söderholm

20.1 Introduction

The ecological and social impacts of wind power plants affect the provision of ecosystem services in many ways, mainly with respect to cultural services as well as habitat provision (for overviews, see [1, 2]). First, the mere existence of wind power plants affects the character of the landscape. Hence, the most concerned ecosystem services are related to inspiration, sense of place, and identity and cultural heritage as these are associated with landscapes. For example, many people tend to attach values to scenic views (unspoiled by wind turbines) which they have grown up with and that form part of their definition of homeland. Second, when producing power, the turning rotor-blades build up a vertical barrier for animals, which causes habitat losses (e.g., interruption of pathways for migratory birds) as well as bird and bat mortality due to collision. These impacts relate to lifecycle maintenance, habitat, and gene pool protection. Bat mortality further affects the pest and disease control because insectivorous bats contribute to the disruption of population cycles of agricultural pests. Tabassum et al. [2] and Hastik et al. [1] show that the specific impact of renewable energy sources deployment on different types of ecosystem services strongly depends on the spatial context and the technology under consideration. In sum, negative spatial externalities are site-specific (i.e., damage depends on the characteristics of the site) and include an additional distance-related dimension (i.e., damage also depends on the distance to human settlements). These challenges need to be considered when a proper institutional framework for the deployment of wind power is to be designed to mitigate trade-offs between different ecosystem services. A suitable policy mix should not rely on just one criterion (choosing sites with the best wind conditions), but rather acknowledge ecological as well as societal spatial externalities. Understanding how the current institutional framework affects the spatial allocation of wind turbines and the related impacts on ecosystem services is a first step towards this insight.

Which ecosystem services are addressed? Regulating (incl. habitat provision, wild plants and animals), and cultural ecosystem services (incl. aesthetical experience, recreation)

What is the research question addressed? Which institutional and non-institutional factors drive the spatial allocation of wind power deployment and related ecosystem services impacts?

Which method has been applied? Statistical evaluation of German and Swedish data

What is the main result? Both non-institutional and institutional factors may have a significant impact on the distribution of wind turbines. Their relative importance varies with geographical and regulatory contexts

What is concluded, recommended? Spatially invariant renewable energy sources support schemes can be combined with regional priority areas for wind deployment to effectively control the spatial distribution of wind power deployment and related ecosystem services impacts

20.2 Drivers of Wind Power Deployment

The deployment of renewable energy sources (RES) till 2020 is a major means to attain the European Union's (EU) energy and climate targets. Wind power plays a major role here across all Member States, even though deployment levels vary significantly across EU Member States and regions.

To increase the deployment of wind power, most Member States have adopted support schemes subsidizing RES power generation. These schemes may thus be presumed to be an important institutional driver behind the scale and spatial allocation of wind power deployment and related ecosystem services impacts. Additional institutional determinants

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include spatial planning and nature conservation regulation. These can be understood as means to better consider ecosystem services impacts of RES deployment by either excluding ecologically particularly sensitive areas or establishing priority areas for wind power generation.

In addition, the deployment of wind power is also driven by non-institutional characteristics, such as differences in geographic patterns (e.g., average wind potential or population density/availability of space) and economic parameters (GDP/capita, unemployment rate, land prices). Such aspects provide an indication of benefits and (opportunity) costs of wind turbine installations.

To what extent the different institutional and non-institutional variables actually drive the spatial allocation of wind power is unclear ex ante and can only be assessed by an empirical investigation. Understanding the empirical importance of the different types of drivers is decisive to subsequently identify means to control the spatial allocation of wind power deployment and its ecosystem services impacts.

20.3 Empirical Approach to Assess Drivers of Wind Power Deployment

Recent empirical studies of the United States [3], Sweden [4], and Germany [1, 5] conclude that both institutional and non-institutional drivers play an important role in explaining differences in the deployment of wind power, despite using different empirical approaches. However, so far, only country-specific studies exist. These tell little about why some factors play a role in one context but not in another. Therefore, we apply a multi-country evaluation framework to better understand differences in the relevance of institutional and non-institutional drivers between countries. We focus on the cases of Germany and Sweden to compate which factors have driven the spatial allocation of wind turbines. For this purpose, we analyze the installation of wind power capacity per sqare kilometer in the years 2008–2012, on the district level for Germany, and on the municipality level for Sweden. When explaining the spatial allocation of this capacity statistically, we control for the impact of the share of protected areas due to nature conservancy-related restrictions (to consider indirect information about the availability of ecologically sensitive areas within a specific region regarding, e.g., habitat provision), as well as population density (which can be seen has proxy for a potential social impact of new wind power installations, e.g., with respect to cultural ecosystem services). Figure 20.1 provides an overview of these variables for the district level of Germany and the municipality level of Sweden.

20.4 Empirical Results: What Drives the Spatial Allocation of Wind Power?

Figure 20.1 illustrates significant differences in the spatial distribution of wind power deployment between Germany and Sweden. It also identifies various differences concerning geographic properties, population density, regulation design, and the renewable energy sources support scheme. Due to the significantly smaller wind power deployment rates in Sweden and a population density that is on average nearly five times lower than in Germany, land availability is certainly more constrained in Germany. An empirical analysis using an econometric Tobit model with Cragg specification [6] reveals that smaller wind power investments at less windy locations are more likely under the German feed-in tariffs, which depend on a reference yield mechanism, compared to a Swedish quota mechanism in which the remuneration does not depend on the locational wind power condition and is in general lower. In terms of the variable "population density", the findings are similar for both countries. The variable, does not seem to influence the probability of having wind power expansions during the period from 2008 to 2012, but various conflicts of interest due to high population density shares may arise and are more likely to constrain the magnitudes of wind power deployment. These results are consistent with previous empirical findings [1, 4]. However, our study does not allow a clear statement concerning the variable "protected areas". Despite having the expected negative impact on wind power deployment rates, the result has not been significant for the German case study. This observation notwithstanding, our analysis indicates that space constraints (imposed by the density of population or protested areas) are clearly more relevant for the spatial allocation of wind power deployment in Germany than in Sweden. This implies that the attainment of the current wind power deployment targets is more likely to produce trade-offs in terms of ecosystem services provision in Germany than in Sweden.

Land use policy is another important aspect of wind deployment. Our statistical model shows that the designation of specific priority areas for new wind power installation has a significant impact on wind power capacity installed within a region. When establishing these priority areas, regional planning authorities typically incorporate and balance the various impacts of wind power deployment on different types of ecosystem services. Our analysis suggests that this type of instrument may in fact be very effective in concentrating wind turbines in selected areas if it is sufficiently binding for wind power investments, which holds true for Germany but not for Sweden. As long as the underlying planning process provides for a proper consideration of ecological and social risks, this policy approach may contribute to mitigating trade-offs between different types of ecosystem services.

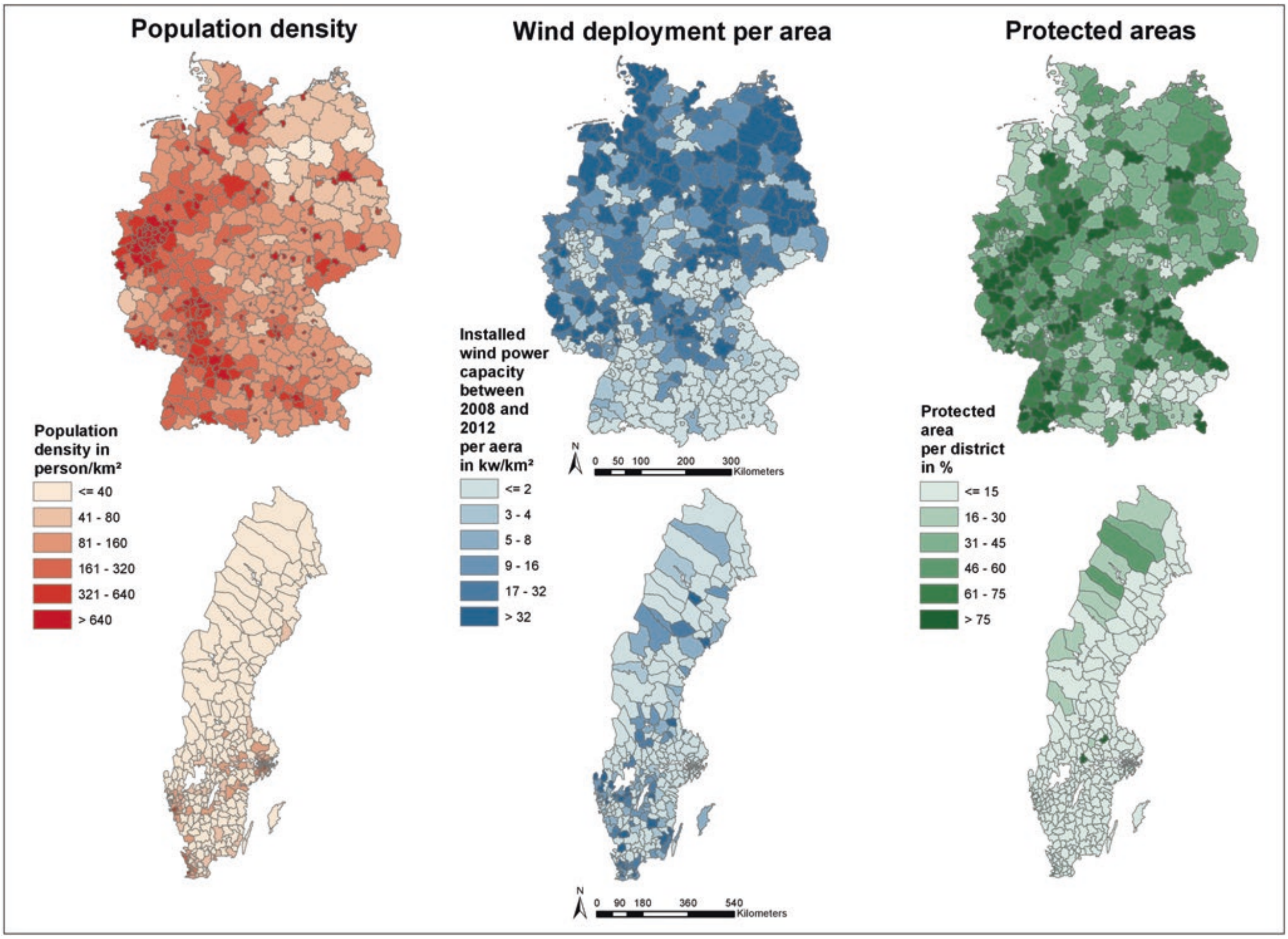

Fig. 20.1 The figure shows, per regional administrative unit for Germany and Sweden, the wind deployment capacity per area installed between 2008 and 2012 and two exemplary social and ecological impact categories as proxies for habitat and cultural ecosystem services, i.e., population density and the overall protected areas

20.5 Conclusions

This analysis is complementary to other risk-oriented analyses, since a solid understanding of spatially differentiated drivers is a necessary component of any comprehensive risk assessment. The analysis confirms that not only are geographic and economic aspects relevant, but the institutional setting also matters, like the implemented support scheme or the practiced land-use policy, when wind power deployment is analysed as stressor for ecosystem services. This allows for designing institutional drivers of wind power deployment in a way that driver-related risks of ecosystem services in "energy landscapes" might be mitigated. One important lesson is that spatially invariant renewable energy sources support schemes can be combined with regional priority areas for wind deployment to effectively control the spatial distribution of wind power deployment and related ecosystem services impacts.

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Selected Trade-Offs and Risks Associated with Land Use Transitions in Central Germany

21

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21.1 Introduction

How can future uncertainties and risks be addressed if systems (e.g., humans interacting with their biophysical environment) are too complex to predict neither their behaviour nor social or environmental impacts? In our case, at a regional scale focusing on land-use change in Central Germany, these coupled systems are usually influenced by a multitude of factors, originating from different scales and domains (e.g., social, technological, environmental, economic, and political) [1, 2]. One of the most frequently used methods in environmental studies, to address at least some potential future developments and some of the consequences and risks, is to use scenarios [3], often developed in participatory approaches to include stakeholder perspectives and increase their credibility and relevance.

The objectives of this study were 1) to identify the biggest future risks and uncertainties for the study region Central Germany, and to develop storylines along the largest uncertainties, how the region may change until 2050 [4]; and 2) to use environmental models to simulate land-use changes and impacts on selected ecosystem services (ES) related to agricultural production.

Which ecosystem services are addressed? Provisioning services in terms of food, feed, and bioenergy crop production.

What is the research question addressed? What are the biggest future risks and uncertainties for the study region Central Germany perceived by stakeholders and scientists, and which land use changes and impacts on selected ecosystem services related to agricultural production can be expected?

Which method has been applied? Participatory scenario development and integrated modelling with the SITE model.

What is the main result? Climate change may have beneficial or adverse effects on crop yield levels, depending on crop type and level of climate change. Crop production is additionally influenced by regional preferences influencing crop land extent (e.g., afforestation), crop management (e.g., organic production), and crop types for food or bioenergy production.

What is concluded, recommended? As climate change, land availability, and land management all influence agriculture, integrated studies like this are needed to assess future crop production. However, sustainability objectives may prefer other than the most productive agricultural pathways providing additional benefits such as regulating or cultural services.

21.2 Methods

21.2.1 Scenario Development

In this study, we organised a participatory scenario process, involving representatives of different societal groups, public and private bodies and organisations, as well as scientists with different backgrounds. Surveys were conducted among scientists and practitioners to assess their views on future risks and drivers of change. The largest discrepancies in stakeholder perceptions were related to the velocity or magnitude of changes, and the question whether the region would take a more citizen- and environmental-friendly pathway or rather focus mainly on regional economic development (Fig. 21.1). The main risks/uncertainties identified by stakeholders and scientists were used to define two key-uncertainty axes, and to develop four scenarios. Assumptions about expected changes were quantified partly in the stakeholder process, partly by the authors to enable the application of simulation models to address land-use change and selected ecosystem services.

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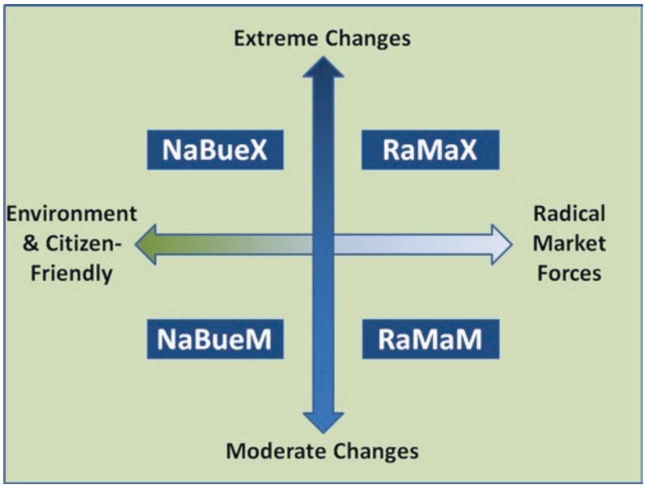


Fig. 21.1 Key-uncertainties and regional scenarios. NaBue scenarios: citizen- and environmental-friendly pathway, land use focusing on bioenergy, organic production, and reforestation; RaMa scenarios: focus on economic development, land use focusing on conventional production and bioenergy crops due to increased fossil fuel prices, while reforestation plays a minor role; rates of change until 2050: M: moderate change; X: extreme change; see Table 21.1 for selected factors driving changes in the four scenarios

21.2.2 Study Region and Simulation Model

For the modelling purpose, the entire area of Central Germany (55,000 km^2) was divided into a 500 m raster grid, each grid cell covered by one land-use/land cover type that was linked with additional socio-economic and biophysical data. Several thematic maps, such as CORINE and others, and statistical information [5] were used as underlying maps to control the land-use model [6, 7]. The upper panel of Fig. 21.2 presents the current land cover and land-use distribution in Central Germany.

The land-use model SITE analyses biophysical and socio-economic suitability factors for each pixel and each land-use/land cover type. Based on this multi-criteria analysis and additional factors such as protected areas, SITE translates the annual "demands" of a scenario (number of people, agricultural products, etc.) into land demands and allocates land cover types. In this study, we distinguished four urban, eight agricultural, three forest, and five other land cover classes such as water, mining, or transportation. Based on annual allocation steps, crop yields and biomass production have been calculated. Based on monthly data of the period 1991–2010 we analysed the impact of weather conditions on the simulated crop yield levels of seven major crops grown in Central Germany, established highly significant linear regression equations between rainfall and/or temperature sums in the growing season and the levels of crop yields, and applied the equations to the scenario period until 2050.

21.3 Results

21.3.1 Scenarios

Four scenarios were developed in a participatory process with regional stakeholders and scientists. The NaBue scenarios assume citizen- and environmental-friendly changes, while the RaMa scenarios assume a strong focus on economic development and less interest in environmental issues or organic production. Two pathways were assumed for both the NaBue and RaMa scenarios, characterized either by moderate or by extreme changes (see [4] for details).

In the surveys and in the workshop and discussions during the scenario process, it turned out that scientists tended to assume high uncertainties and risks from large-scale driving factors such as changes of climate or the (global) economy, while non-scientific stakeholders tended to focus on regional factors such as population decline and changes in regional lifestyles or preferences, e.g., concerning bioenergy production, organic agriculture, or the regional economy. Both large-scale and regional drivers were addressed in the simulation model.

21.3.2 Risks and Opportunities Associated to Climate Change

Weather conditions were known to differ between subregions and were also expected to differ in the future. Thus, the analysis was carried out for each federal state—Saxony, Saxony-Anhalt and Thuringia—and each crop separately, assuming crop management to be continued at the same level of intensity as today. Crop yields shown in Fig. 21.3a (left) under conditions of moderate climate change were calculated to remain at a similar level to today (±0), or slightly benefit (+5%), except spring- and winter-barley, for which slightly lower yield levels resulted. The simulations presented in Fig. 21.3b (right), under conditions of strong climate change, show declining yields of major crops grown in the region towards 2050 (rye, winter barley, rapeseed, wheat, spring barley). Contrastingly, maize and sugar beet seem to benefit from future warmer climate conditions. Consequently, for the latter two the future risk of yield losses is lower than today, while for the other five the risk of yield losses is expected to increase. The expected changes presented as averages over 10 years may seem small, but in several single years they can reach +20% to −30% under strong climate change and mostly between +20% and −20% under moderate climate change, compared to current yield levels.

Table 21.1 Factors driving land use change and their influence on agricultural production[a]

Factor driving change	NaBueM		NaBueX		RaMaM		RaMaX	
	Factor	Risk AP	Factor	Risk AP	Factor	Risk AP	Factor	Risk AP
Reforestation	↑↑	↑↑	↑↑↑	↑↑↑	→	→	→	→
Expansion of urban land	↑	→	↑	→	↑↑	↑	↑↑↑	↑
Organic agriculture	↑	↑↑	↑↑	↑↑↑	→	→	↓	→
Bioenergy production	↑	→	↑↑↑	↑	↑	→	↑↑↑	↑
Change in agricultural productivity	↑	↑	↑↑	↑↑	↑↑	↑↑	↑↑↑	↑↑↑

[a]This table presents simulation results based on relative changes of five different factors (Factor), as well as their risks for agricultural production (Risk AP); ↑ increase; → no change/effect; ↓ decrease; multiple arrows indicate stronger changes or effects

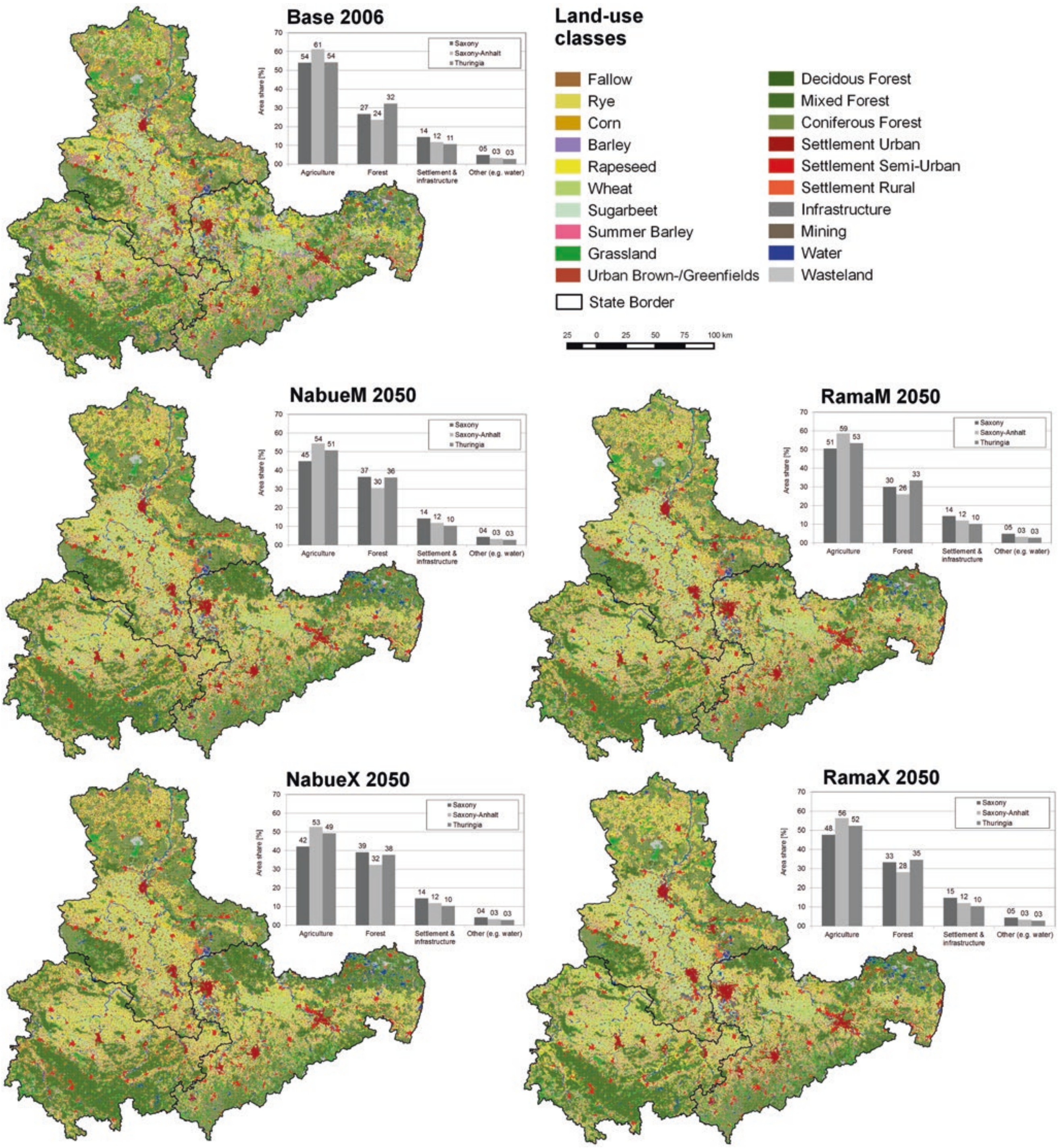
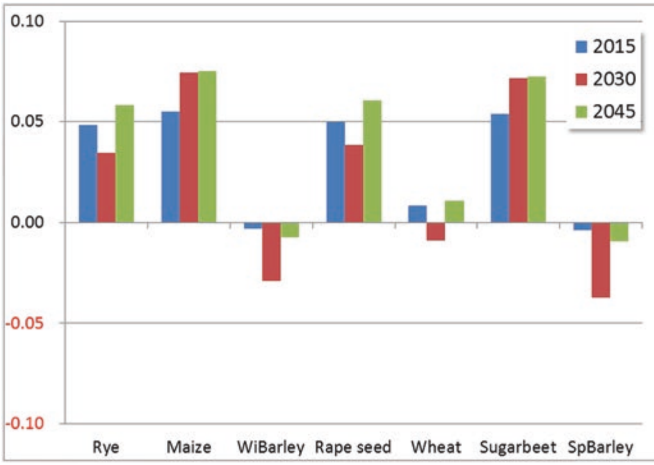


Fig. 21.2 Current and potential future land use and land cover. The study region Central Germany comprises three federal states (Saxony, Saxony-Anhalt, Thuringia). Upper panel: current land use and land cover. Central and lower panels: simulated maps of potential land use and land cover in the year 2050 of the scenarios NaBueM, NaBueX, RaMaM and RaMaX. Bar charts indicate percent land cover for four aggregated classes in the three federal states of the study region. Source: Own calculations based on simulations with the SITE model



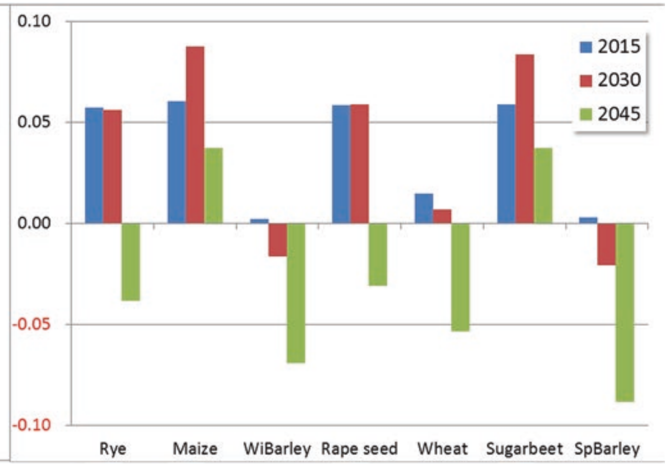


Fig. 21.3 Impacts of climate change on crop yields in Central Germany. Crop yields are presented as changes relative to current yield levels and represent averages of 10 years. (**a**) (left), under conditions of moderate climate change (IPCC SRES B1 scenario). (**b**) (right), under conditions of strong climate change (IPCC SRES A2 scenario). Source: Own calculations based on simulations with the SITE model

21.3.3 Risks Associated to Regional Factors of Change

Here we present the risks for agricultural production, based on the land-use changes simulated for the different scenarios (Table 21.1). Some of the links between changes in drivers and risks or trade-offs for agricultural production are almost trivial, as in the case of land conversions consuming agricultural land, such as reforestation expanding to lower slopes of the Harz or the Thuringian Forest mountains in the west (Fig. 21.2, e.g., map and bar charts in lower panel of the NaBueX scenario). Other links are less obvious. In the case of increasing organic agriculture (crop production), we assumed a fraction of approximately 2/3 of the conventional productivity based on GENESIS online database [5] and similar sources. In the environmental and sustainability-oriented NaBue scenarios 14–25% of the farmland was converted to organic production.

Shifts towards higher production can be expected from increased use of bioenergy crops, because the crop mix is dominated by the high-yielding crop maize, the latter additionally benefiting from climate change.

Total agricultural production until 2050 differed widely in the four scenarios, ranging from moderate decline in NaBueM to considerable increases in RaMaX. Thus, the simulation of the combination of different large-scale (e.g., climate) and regional drivers of such as land use and land management preferences led to partly unexpected net effects on agricultural production and showed some of the trade-offs to be expected between the different drivers and rates of change assumed by stakeholders and scientists in the regional scenarios (Fig. 21.4).

21.4 Discussion

Major risks, uncertainties, and preferences identified by scientists and stakeholders were integrated into scenario storylines and drivers of change. Additionally, historic trends of land cover changes observed in (Central) Germany, such as decreasing agricultural areas, urbanisation, and growing forest areas [8, 9] were continued in the scenarios. The widely differing changes in the scenarios until 2050 in land use and land management and their impacts on ecosystem services are reflecting the range of assumptions and stakeholder preferences (Fig. 21.2 central and lower panel maps).

21.4.1 Consequences of Land Cover and Land-Use Change

The demands for commodities of agricultural sub-sectors such as bioenergy to which the assumptions were translated, drove land-use changes and agricultural production. It is noteworthy that in this regional scenario process, stakeholders uniformly assumed increasing bioenergy crop production. In the environmental and sustainability-oriented NaBue scenarios, the amount of farmland converted to organic production was partly surpassing the ambitious 20% goal set by the first red-green German government. The combined shifts in land cover, land use, and agricultural management led to trade-offs in ecosystem service provision, i.e., food (lower, due to land losses and partly shifts to organic production), bioenergy (higher, benefiting from land gains, preferences and partly climate), and forest related ecosystem services (higher, benefiting from land gains driven by reforestation preferences).

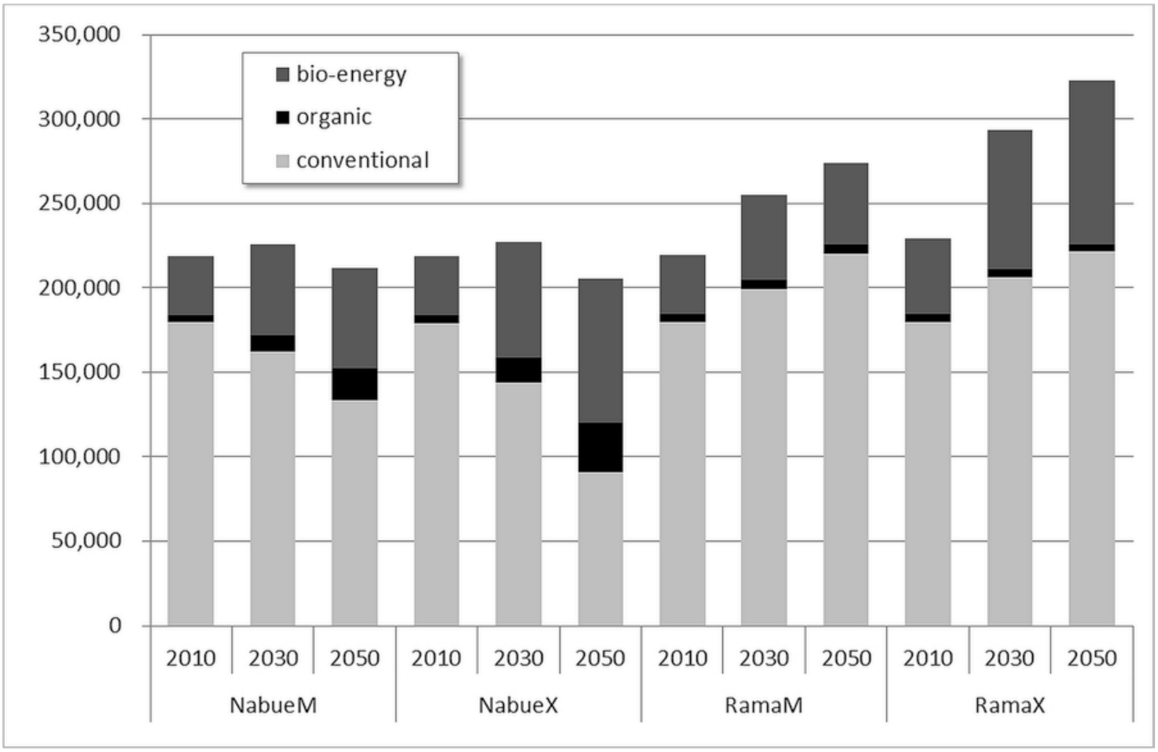


Fig. 21.4 Agricultural production pathways until 2050. To enable aggregation across different crops, results are presented in units of energy (Tera Joule) for three different periods until 2050. Source: Own calculations based on simulations with the SITE model

The large land-use changes calculated for the agricultural sector towards 1) organic food and 2) biofuel production, and, in the 3) forestry sector, towards reforestation, reflect the perception of risks, preferences, and the state of knowledge at the time of scenario development in 2009–2011, especially in the NaBue scenarios. Today, 6 years later, perceptions have changed and political, public, and scientific debates are more critical and less supportive of biofuel-related benefits, but more focused on the trade-offs in ecosystem service provision related to them. However, legacies of the bioenergy pathway supporting national and EU-level targets and regulations [10, 11] persist and accumulate, e.g., the numbers of biogas plants in Germany, or subsidised E10 fuel (containing 10% biofuel) being sold in every German gas station. Thus, it may happen that the scenario estimates of up to 30% of conventional cropland for bioenergy may underestimate rather than overestimate future demands, which are founded on the long-lasting consequences of built infrastructure as well as current policies and regulations and ambitious EU targets for 2030 (27% renewable energy share) and 2050 (80% reduction of European GHG emissions).

21.4.2 Combined Impacts of Biophysical and Socio-Economic Factors on Crop Production

In accordance with large-scale yield trends identified for the recent past and climate change affected futures [12] for mid- and high-latitude crops, our model simulated mostly moderate gains and losses for the seven crops analysed, except peaks in extreme years reaching gains or losses up to 30%. However, our results also suggest that long-term trends until 2050, especially under the more severe climate change, are not only negative for most crop yields, except sugar beet and maize, but can also be expected to vary stronger than today [13]. Consequently, the risks of climate-related crop failures are expected to increase, slightly less for biofuels, because the crop mix (cereals, maize, rapeseed) partly benefits from climate change.

In our study, we addressed three major influences on agricultural production: 1) climate change impacts on crop yields; 2) land-use change, driven by shifts in regional preferences concerning conventional, organic, and bio-energy production; and 3) land management in terms of conventional and organic production and crop selection for food and bioenergy. While climate impacts on all scenarios are similar until 2030, stronger negative climatic impacts towards 2050 are only visible in the RaMaX scenario, dampening production increases in the period 2030–2050. However, major differences calculated for agricultural production between the RaMa and the NaBue scenarios originate in the strong contrasts of the areas under organic production, which remain at low-to-very-low levels in the RaMa scenarios, but strongly increase in the NaBue scenarios. Thus, the combination of land loss, climate effects, and large-scale lower-yielding organic production bares the risk of yielding production levels lower in 2050 than today, as calculated for the NaBueX scenario.

Despite its lower productivity and consequently larger demand for land, organic agriculture is seen as one of the pathways contributing to more sustainable production worldwide. The main reasons are that organic agriculture is expected to lower regional and global risks and negative impacts for ecosystems and ecosystem services—e.g., regulating ecosystem services associated with high application rates of fertilizers and agro-chemicals in conventional agriculture—and that it contributes to the paradigm shift in resource use suggested by Seppelt et al. [14].

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New EU-Level Scenarios on the Future of Ecosystem Services

22

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22.1 Introduction

This study was elaborated in OpenNESS, a European Union FP7 research project (http://www.openness-project.eu) focusing on the operationalization of natural capital and ecosystem services. The new European Union level OpenNESS scenarios have been developed to fill a thematic gap in existing broad-scale environmental scenarios to assess the uncertainties and risks of different drivers of change for natural capital and ecosystem service provision. The scenarios are aiming at applicability for science and policy-making at different scales, including the European level [1] and regional and local scales, e.g., in the OpenNESS case studies. The conceptual framework and methods for integrative scenario development mainly followed Priess and Hauck [2]. Similar to the SRES scenarios of the IPCC or the GEO4/5 scenarios of the United Nations Environment Programme [3, 4], drivers and uncertainties identified by the primary users and the scenario team were organized along axes of key-uncertainties, focusing on the key objective of OpenNESS, which is the operationalization of ecosystem services. So far mainly the scenarios of the Millennium Ecosystem Assessment have been addressing ecosystem services explicitly [5]. However, in the Millennium Ecosystem Assessment scenarios there was a strong focus on supply and demand of provisioning services and the sustained provision of ecosystem services was assumed for all scenarios. In these aspects, the OpenNESS scenarios are going beyond the Millennium Ecosystem Assessment scenarios, making broader assumptions in covering different types of ecosystem services, and different pathways of ecosystem services provision, including risks of ecosystem service losses.

In this study we focus on uncertainties and risks related to land-use change, exemplified via simulation results for a provisioning and a regulating service.

Which ecosystem services are addressed? One provisioning service and a regulating service, namely food production and carbon sequestration.

What is the research question addressed? What are uncertainties and risks related to land-use change, exemplified via impacts on a provisioning and a regulating service?

Which method has been applied? Participatory scenario development and integrated modelling with the CLIMSAVE modelling framework.

What is the main result? A large set of different drivers of change, including social, technological, ecological, economic, and policy drivers was needed to reveal the potentially big differences in terms of positive and negative impacts on future land use and the provision of different ecosystem services in Europe. Trade with land-intensive commodities contributes to lower or increased pressures on ecosystem services.

What is concluded, recommended? Drivers within Europe as well as trade with land-intensive commodities and the policies steering them contribute to lower or increased pressures and risks of ecosystem services loss, e.g., on agricultural land in Europe and the countries of trading partners.

22.2 Methods

The four scenarios were developed in an iterative process involving the scenario team, intended users, and EU-level policymakers, to ensure quality, consistency, and applicability [6].

Figure 22.1 reflects the key uncertainties which pathways in Europe might take in terms of developing more integrated cross-sectoral policies, and whether governance would develop towards more responsibilities at European or at

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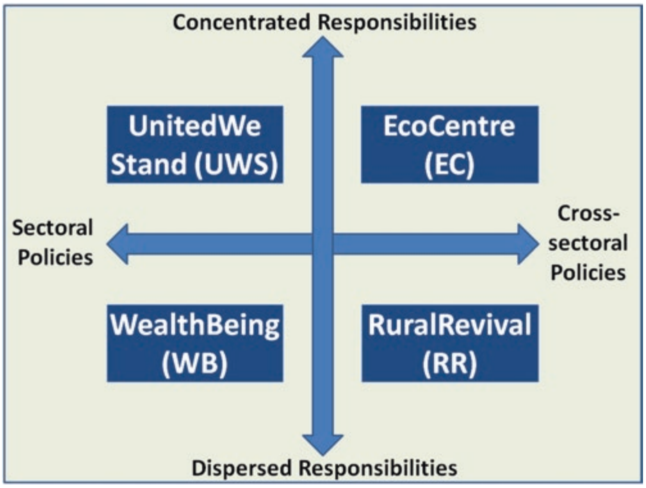

Fig. 22.1 The OpenNESS scenarios. The scenarios populate the quadrants of the two key-uncertainty axes—vertical policy integration between the EU and the member states and horizontal policy integration between sectors of society

national levels. Based on the uncertainty axes and a recent review of scenario drivers [7], in the next steps the scenario team, supported by modelling experts, quantified the drivers of change, again using an iterative process. Drivers included changes in population and lifestyle (food and settlement preferences), economic (including trade) and technical developments, and policies addressing, e.g., agriculture, water use, or nature protection (Fig. 22.1). The quantified driver assumptions have been used to parameterize EU-level models.

Simulations were carried out with the CLIMSAVE integrated assessment platform, a modelling framework to assess impacts of climate and other drivers of change on land use and ecosystem services at the European scale [8]. CLIMSAVE covers the countries of the European Union plus Switzerland and Norway, and is presented as grid-cells of 10′ by 10′. The allocation of crops and forests was based on bioclimatic and terrain properties. Additionally, forests using tree species adapted to climate change store more carbon per unit area (all scenarios but UnitedWeStand). Simulations were run for a 2010 baseline year and for 2050. All scenarios have been calculated using the same climate scenario (SRES A1 using the CSMK3 climate model with medium climate sensitivity), which means that risks related to climate change are identical for all four OpenNESS scenarios, which enabled us to more explicitly address the impacts caused by, e.g., differences in other drivers (see Sect. 22.3.2).

22.3 Results

22.3.1 Scenarios

The four scenarios, WealthBeing, UnitedWeStand, EcoCentre, RuralRevival, cover the European Union. They were all structured in the same fashion. The storylines start with assumptions about events triggering the pathway of each scenario, followed by assumptions about midterm (until 2030) and long-term developments (until 2050). Table 22.1 provides an overview of the key-assumptions.

22.3.2 Simulating Land-Use Change and the Risks for Ecosystem Services

The CLIMSAVE model was parameterized with the list of quantified drivers developed for the OpenNESS scenarios. Starting from the current distribution of land use and land cover (Fig. 22.2), simulations of the combinations of driving forces caused land-use changes up to +65% (UnitedWeStand: forest) or −37% (UnitedWeStand: grassland) (Table 22.2). The most extreme changes highlighted here for scenario UnitedWeStand can mainly be attributed to the highest increase in irrigation efficiency (+58%) and crop yields (+50%), an almost constant human population in Europe (+1%), moderate increases in meat demands (+10%) and increasing food imports (+10%).

The land-use and land-cover changes caused by the different combinations of drivers had very pronounced impacts on the provision of ecosystem services, increasing or decreasing the risks of lower levels of ecosystem services. Here we focus on the provisioning service "food" and the regulating service "carbon sequestration" provided by managed forestry (Fig. 22.3). The maps show, for example, a considerable consolidation of food provision in central Northern France, Northern Italy, and Central Europe under the WealthBeing scenario, matched by a reduction in forest area and carbon sequestration. This reflects the pressures put on the agricultural system by WealthBeing's increasing population (+10%) and its reduced reliance on international imports (−20%). Conversely, UnitedWeStand, a scenario that shares similar levels of agricultural innovation to WealthBeing, but has only limited population growth (1%) and increases food imports relative to today's levels (+10%), shows less concentrated food provision and more carbon sequestration associated with managed forests across Northern and Central Europe. The other scenarios,

Table 22.1 Key-assumptions used in the OpenNESS scenarios

Policy area	WealthBeing	UnitedWeStand	EcoCentre	RuralRevival
General tendencies				
	Large political and economic differences between member states but also globally; sectoral EU policies, national legislation strengthened; deregulation of markets.	Joint EU policy approaches, sectoral policies; economically, EU and the world are developing at a comparable moderate pace.	Cross-sectoral EU policy integration; EU leads mainstreaming of ecosystem services and changes towards eco-friendly life style, other countries follow.	Large differences between member states; cross-sectoral integration; economically EU falls behind the rest of the world.
Midterm developments until 2030				
Political, societal, and economic change	Economic success, and growth of export sectors; unequal distribution among member states; high demands and prices for resources and energy; reduced social and environmental standards.	Prosperity of all member states and citizens; Euro-centric visions and policies; strong belief in technical solutions for environmental problems; substantial investments in education and social policies; neglect of environmental concerns.	EU-wide campaign for environmental education and awareness raising; reduced consumption; environmental justice; participatory (environmental) decision-making; 'Genuine Progress Indicator' introduced to account for environ-mental and social factors.	High popularity of green, idealistic citizen and 'back to nature' and 'simple life' movements; cooperative and less wealth-oriented policies; local manufacturing; EU institutions dwindle; high outmigration.
Urban, rural and grey infrastructure development	Rural infrastructure and settlements neglected; strong urbanisation and urban sprawl.	Strong development of industries and infrastructures; urbanisation and urban sprawl.	Urban green development and gardening; Open-source mentality; strongly increasing efficiency resource use.	Rural areas regain socio-economic importance; different types of work and sustainable life-styles develop.
Land use and environmental conservation	Intensification of agriculture and forestry; high demand for renewable energy and materials; Consumerism as leading lifestyle; alliances between agrarian and industrial lobbies weaken environmental policies.	Consumerism as leading lifestyle; importance of regulating ecosystem services decreasing, due to technical solutions; increase in greenhouse gas emissions, land use change and exploitation of mineral resources; decreasing environ-mental concerns.	Agricultural production is converted into organic farming or sustainable integrated farming; pressure on land resources; environmental conservation with the idea of "rewilding." Protected areas increase.	Cooperatives and farmers diversify production; lower land-use intensity and mechanisation; EU imports of agricultural commodities; Protected areas increase, their role is debated hotly.
Long-term developments until 2050				
	Degradation of agricultural and aquatic systems due to high demand for ecosystem services; prices for all land intensive commodities continue to rise.	Degradation of ecosystems pushes technical solutions to their limits; transition from fossil fuels to renewables; growing demand for provisioning services from outside EU.	Unsuccessful trials of participatory EU policies, due to high bureaucracy and low efficiency; EU continues to be a strong actor but also facilitates regional developments.	General focus on sustainable management strategies; large multinational companies and agro-industries either adapt or move out of EU; less organised regions are left behind; revival of traditional, well adapted varieties of crops, vegetables, fruits, old livestock races.

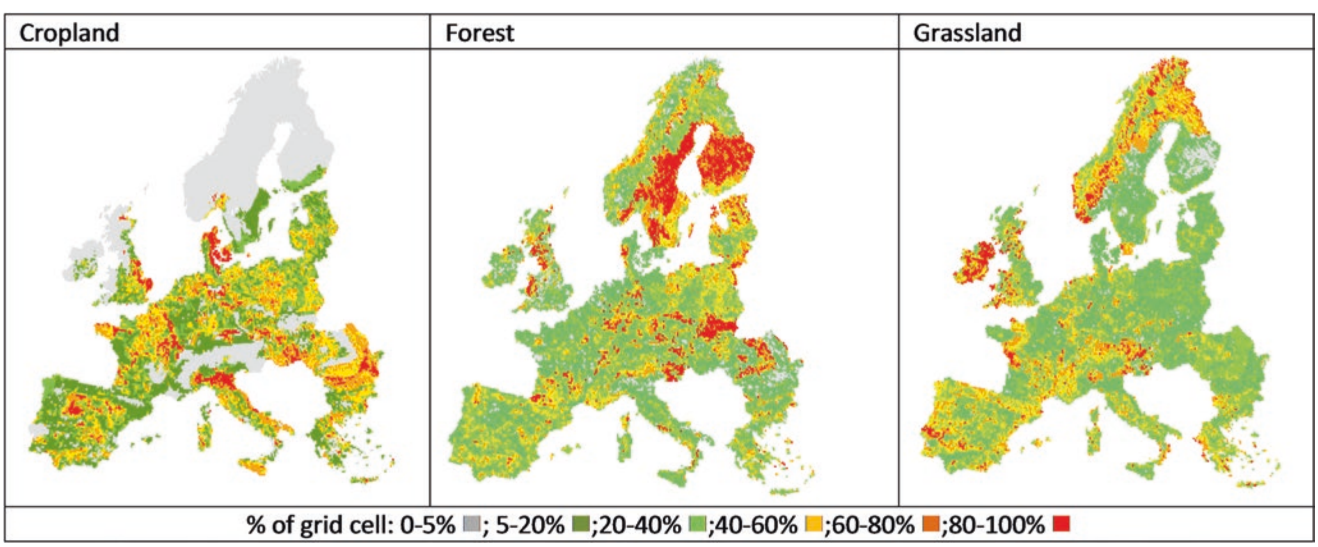

Fig. 22.2 Current land use and land cover (year 2010) as represented in the CLIMSAVE model

Table 22.2 Land use change in Europe for 2050[a]

Land use	Scenario			
	WealthBeing	UnitedWeStand	EcoCentre	RuralRevival
Cropland	−1	−39	+8	+6
Forest	+13	+65	+10	+11
Grassland	−17	−37	−18	−17

[a]Changes are presented as relative differences (%) between the baseline in 2010 and 2050

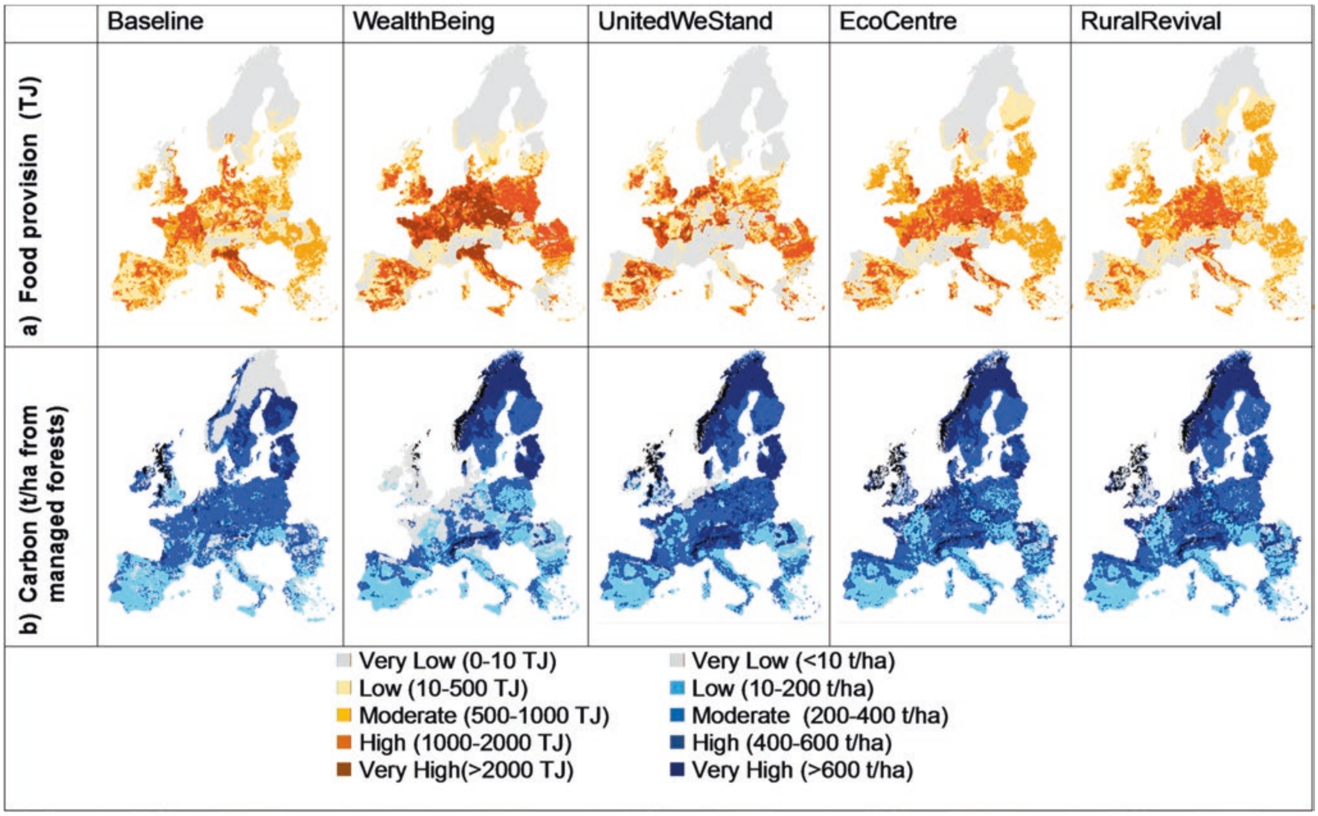

Fig. 22.3 Examples for changes in ecosystem services provision in Europe under four different scenarios. (**a**) Food provision; (**b**) carbon sequestration

RuralRevival and EcoCentre also show clear differences in spatial patterns, driven by different levels in innovations, different dietary preferences, and declining populations. In RuralRevival, where agricultural yields are 10% lower than today's and no improvements in irrigation take place, climate change is more of a pressure and food provision pushes further North through Estonia and Finland with knock-on impacts on carbon sequestration in these areas.

22.4 Discussion

22.4.1 Scenario Development

The development process benefitted substantially from the iterative cycles the scenarios went through, in terms of knowledge integration from local and EU-level stakeholders and enabling the developer team to involve additional experts (e.g., concerning global trade assumptions) to increase the quality and consistency of scenario assumptions [6]. While integrating intended users from the case study level worked well, originally a higher level of stakeholder participation was envisaged at the European level, at which just one intensive workshop was possible. Based on the storylines, the case study survey and the scenario review [7] a set of drivers of change was established and their expected changes were quantified in an iterative quality assurance procedure, applied to repeatedly cross-check assumptions.

22.4.2 Scenario Applications

The qualitative scenario storylines and the quantified drivers were intended to serve as boundary conditions in several of the 27 case studies, and to parameterize models, e.g., at EU level (see next section) or in the case studies. The team supported some of the case study "frontrunners" in downscaling and contextualizing the scenarios to regional/local levels (see Zurek and Henrichs [9], Kaljonen et al. [10], and Metzger et al. [11] for downscaling methods). The different worlds, or pathways into the future, elaborated in the storylines were either considered relevant "as is," or were complemented by regionally relevant drivers and processes. It was appreciated that current experiences/problems such as decreasing levels of biodiversity or ecosystem services provision were also reflected in some of the scenarios. As the Millennium Ecosystem Assessment scenarios [5] were limited to assuming sustained ecosystem services provision of provisioning services, they would have been less compatible with the broader views preferred in this study, e.g., about pressures on land resources and increasing or decreasing levels of ecosystem services provision, at both European and regional levels.

22.4.3 Simulation of the Scenarios

The scenarios were simulated with the Integrated Assessment Model CLIMSAVE. A primary advantage of Integrated Assessment Models is that they allow a quantitative basis for sense-checking the impacts of societal and climatic changes within the scenario worlds [8]. Integrated Assessment Models can provide quantitative information to address questions raised within scenarios. For example, is it possible to meet demand for food in a world where the population is increasing, dietary preference for meat is increasing, and trade policies aim at exporting less? What does this mean for other sectors/policies such as forestry (e.g., climate-adapted tree species) or biodiversity (e.g., protected areas)? Simulation results can be interpreted alongside the scenarios to identify where aspects of qualitative storylines work—and where there may be physical, social, or environmental limits as to what is possible. Figure 22.3 and Table 22.2 clearly show that the combinations of drivers in the scenarios lead to partly strong shifts in land use and land cover in terms of space and quality, creating both positive and negative impacts on the ecosystem services associated with them. This type of assessment is particularly important in areas such as ecosystem services, where cross-sectoral impacts are so significant. Model integration facilitates identification of where synergies are possible and where the trade-offs may be; thus the exploration of chances and risks associated with changes such as the unexpected increases of croplands in the EcoCentre and RuralRevival scenarios, or the north shift of agriculture in the RuralRevival scenario. In addition, scenarios in combination with Integrated Assessment Models are valuable as a means to provide different types of outputs that can be used within scenario processes with stakeholders to facilitate their imagination and understanding of the scenario worlds by visualizing results in the form of maps and tables, which are especially useful to address unexpected results, such as strong increases or decreases or shifts of land uses as discussed above for the EcoCentre and RuralRevival scenarios.

22.5 Conclusions

The simulated results of the scenarios in terms of land-use change and selected ecosystem services represent the interplay of a large set of different drivers of change, including social, technological, ecological, economic, and policy drivers. In terms of ecosystem services provision, no clear winners and losers could be identified, but rather different mixes of trade-offs and synergies. Based on the storylines, one could assume that a scenario such as EcoCentre, with a slightly decreasing human population and less resource-demanding lifestyles, would be a winning team in terms of sustainable ecosystem services provision. It turned out, however, that reduced foci on technology development and application and low crop yields require more land for agricultural production than today, while more people with higher demands, but a strong focus on technological development as assumed in UnitedWeStand, would require considerably less space, but would need the support of more food imports than today and in the EcoCentre scenario. Thus, not only drivers within Europe, but also trade with land-intensive commodities and the policies steering them, contribute to lower or increased pressures and risks of ecosystem services loss, e.g., on agricultural land in Europe and the countries of trading partners. These results reflect the broad views of the qualitative storylines on potential futures of ecosystem services and natural capital in Europe. These broad views enabled the team and other users to assess not only some consequences of desired, but also of less desired pathways, as well as partly unexpected impacts such as those in the EcoCentre and RuralRevival scenarios that came about as a result of the interactions of the multitude of driving forces pointing towards risks of trade-offs and ecosystem service losses.

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23 The Rural-to-Urban Gradient and Ecosystem Services

Dagmar Haase

23.1 Introduction

Urbanisation is a significant form of land take that has various impacts on the pattern, functionality, and dynamics of natural landscapes, and thus also on the ecosystem services provided [1]. Such effects become particularly obvious if observed and analysed along an urban-to-rural gradient. This article presents a case study on long-term land-use and impervious cover change along the urban-to-rural gradient in the city of Leipzig, Germany. Any broad range of empirical studies and modelling exercises can show that soil sealing towards the rural periphery imposes the risk of a diminishment through a complete decline of ecosystem services provisioning in an urban region. At the same time, urban land perforation, decline, and temporary brownfields create new open spaces after demolition and de-sealing in inner parts of the city, and interim uses such as community gardens, as well as large remnants of nature such as rivers, riparian zones, and wetlands represent ecosystem services provisioning units of great importance that lower the risk of an ecosystem "desert" along and most importantly in the centre of the rural-to-urban gradient [2]. The gradient concept offers a promising approach to, first, integrate historical data into current land-use change impact assessment and, second, to uncover effects of iterative and simultaneous phases of urban growth and decline, including sprawl and compaction [1].

Which ecosystem services are addressed? A multitude of ecosystem services, including provisioning (including food production, gardening), regulating (including water regulation, flood protection), and cultural ES (including recreation).

What is the research question addressed? A case study on long-term land-use and impervious cover change along the urban-to-rural gradient in the city of Leipzig, Germany, and its impacts and risks on the provision of ecosystem services.

Which method has been applied? GIS, statistics, field mapping, modelling.

What is the major result? Soil sealing towards the rural periphery imposes the risk of a diminishment through complete decline of ecosystem services provisioning in an urban region. Urban land perforation, decline, and temporary brownfields create new open spaces after demolition and de-sealing in inner parts of the city, interim uses such as community gardens as well as large remnants of nature such as rivers, riparian zones and wetlands represent ecosystem services provisioning units of great importance that lower the risk of an ecosystem "desert" along and in the city centre.

What is concluded, recommended? The gradient concept offers a promising analysis approach for assessing ecosystem services at risk under long-term spatial urban land-use change.

23.2 Setting the Scene

Urbanisation is arguably the most significant form of land-use and cover change because it has considerable effects on the pattern, dynamics, and functionality of landscapes and ecosystems, including the services they provide [3]. The process of urbanisation can be observed along the urban-to-rural gradient, that is, the ideal typical transect that links "the urban" (built/sealed) and "the rural" (open, vegetated), which displays a typical configuration in terms of population and built-up density, impervious cover, and demographic structure next to (with decreasing tendency) living habits and lifestyles [1, 4]. Along this gradient, an increasing amount of land consumption, i.e., the transformation of vegetated into built surface, has been reported on by a multitude of authors based on empirical research and the analysis of statistical data [5–8]. At present, the transformation of the urban-to-rural gradient is detected to a great extent by remote sensing methods [9].

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23.3 Effects on Ecosystem Services Along the Rural-to-Urban Gradient

Numerous studies have shown that land consumption is a real risk for the human-environment-complex in various regards [10] as it could affect ecosystem services [11–13] and, consequently, reduce the ability of nature to fulfil human requirements [13]. Many of the negative effects of land consumption along the rural-to-urban gradient can be attributed to the sealing of soils [10]. The transformation of open or arable land to impervious cover can thus be taken as a key variable when it comes to mapping and evaluating land-use change and its impacts along the urban-to-rural gradient. Individual ecosystem services that are impaired by the spread of impervious cover include the production of food, the regulation of energy and matter flows by soil particles and vegetation, freshwater supply, the provision of recreational space, habitats for species, and natural aesthetic values (Fig. 23.1) [2].

In the following, single ecosystem services are discussed in terms of the impacts on them along a rural-to-urban gradient in urban regions: Potentials and risks are shown for provisioning services such as food production in cities, and regulation services such as surface water retention, air temperature regulation, and pollutant filtration.

23.4 The Example of the Water Regulation and Flood Risk Mitigation

There is a rural-to-urban gradient of surface-water-runoff regulation as shown in Fig. 23.2 for the city of Leipzig, Germany [10]. At a total water balance of 560 mm per year, we find surface-runoff of >250–300 mm up to 400 mm in the central parts of the city. Also in the outer parts, where large newly-built commercial areas (holding companies such as Porsche, BMW, German Post, and Amazon to list a few) and the delivery companies of the Airport Leipzig-Halle have been built, surface water runoff reaches up to 450 mm or 80% of the total annual water balance [10, 14]. Surface runoff regulation in the central floodplains is high (not highest due to small filtration paths and high groundwater levels in wetlands); this refers also to large urban parks. Thus, we cannot state a clear rural-to-urban gradient when it comes to rainwater-induced flood retention and risk mitigation, but the clear risk that the surface water regulation capacity of the city area is being diminished is particularly relevant when it comes to heavy local precipitation events.

23.5 The Example of Pure Air Supply

The picture is different for air purification and the ecosystem service of pure air supply [11]. Here, Fig. 23.3 shows a clear rural-to-urban gradient with maximum values of >2.5 t/km^2*a of PM_{10} in the central parts of the city due to high traffic volumes over the whole day and at night time [14]. As Leipzig is a compact city, traffic concentrates in the centre and decreases with increasing distance to the periphery (Fig. 23.3). In addition to the pollution sources for particulate matter in the central parts of the cities, the risk for the urban population is high as a) most people live in the inner parts and thus are directly affected in their daily life; and b) other stressors such as noise and heat "pollution" also concentrate in the city centre of compact cities [14, 15]. However,

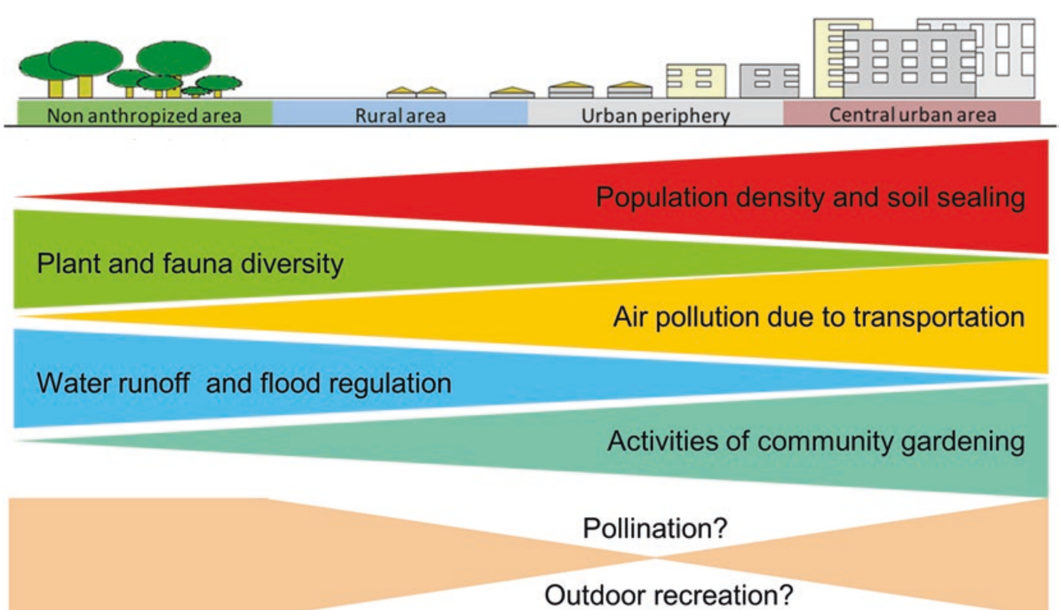

Fig. 23.1 Dynamics of ecosystem services provisioning along the rural-to-urban gradient

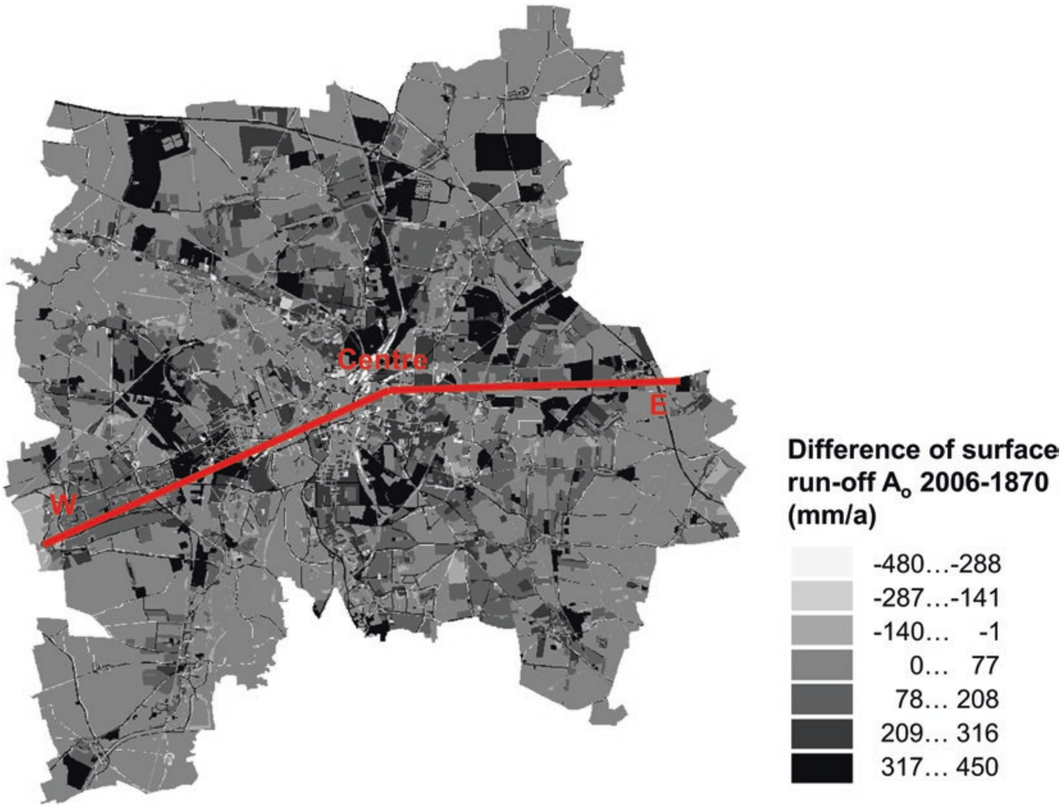

Fig. 23.2 Soil sealing and surface water flow retention capacity along the rural-to-urban gradient in Leipzig, West (W)-Centre and Centre-East (E)

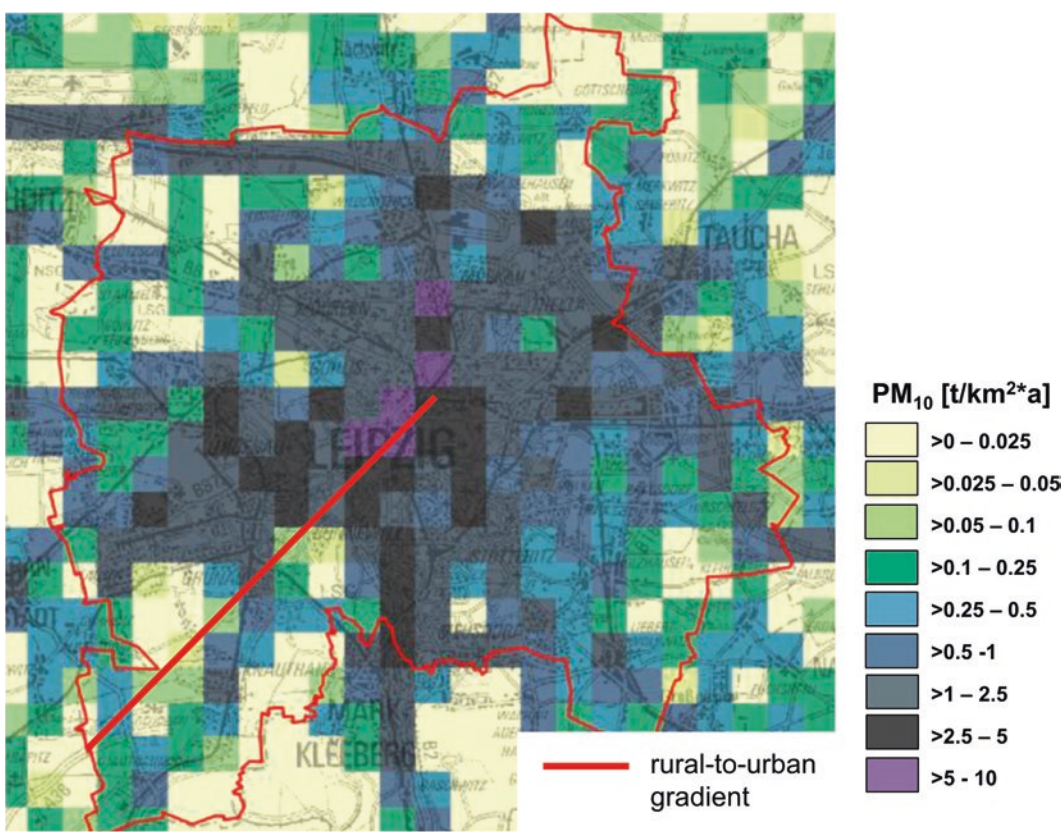

Fig. 23.3 Particle emissions (PM_{10}) from transport (Lagrangian particle dispersion model LASAT [14]; *with permission*)

Fig. 23.4 Community gardening and local food production in the inner parts of Leipzig and Berlin

Fig. 23.3 also provides evidence that green spaces exhibit much lower particle emissions (up to <0.25 t/km^2*a) to be seen in the pixels where the floodplain forests are situated and where large parks are located. Thus, a combination of measures of traffic reduction and more (large and tree-rich) green infrastructure to filter particulate matter and buffer noise are urgently needed to reduce the health risk for the residential population.

23.6 The Example of Urban Food Production and the "Edible City"

As Fig. 23.1 already suggests, there is a reversed rural-to-urban gradient when it comes to the local food production ecosystem service in cities, studied in the cities of Leipzig and Berlin. Most of the community gardens are situated in the central parts of the city, predominantly as interim or non-permanent uses [10]. Here, local food is produced in a bottom-up and participatory way. The amount of food is not "commercial" in the sense of selling own-grown food to earn money, but the productivity of the gardens has shown that they could contribute to neighbourhood food supply and, what is even more, to social cohesion and education about nature and dealing with nature [11] (see also Fig. 23.4). Thus, the "edible city" shows a clear urban-to-rural gradient and a great potential to counteract "urban risks" of social segregation, fragmentation, and isolation (of children, low-income households, migrants, unemployed, etc.). But habitat connectivity and pollination could also be improved by community gardens, as they provide nicely structured vegetated spaces, including old fruit trees that are key for pollination.

23.7 The Example of Heat Mitigation

Last but not least, a clear rural-to-urban gradient can be found when we look at the risk of urban heat and the ecosystem service of heat regulation by green (vegetation) and blue (waters) infrastructure of the city. Leipzig as a compact city exhibits a clear urban heat island with high evening (surface) temperatures in the city centre in summer (Fig. 23.5) [11].

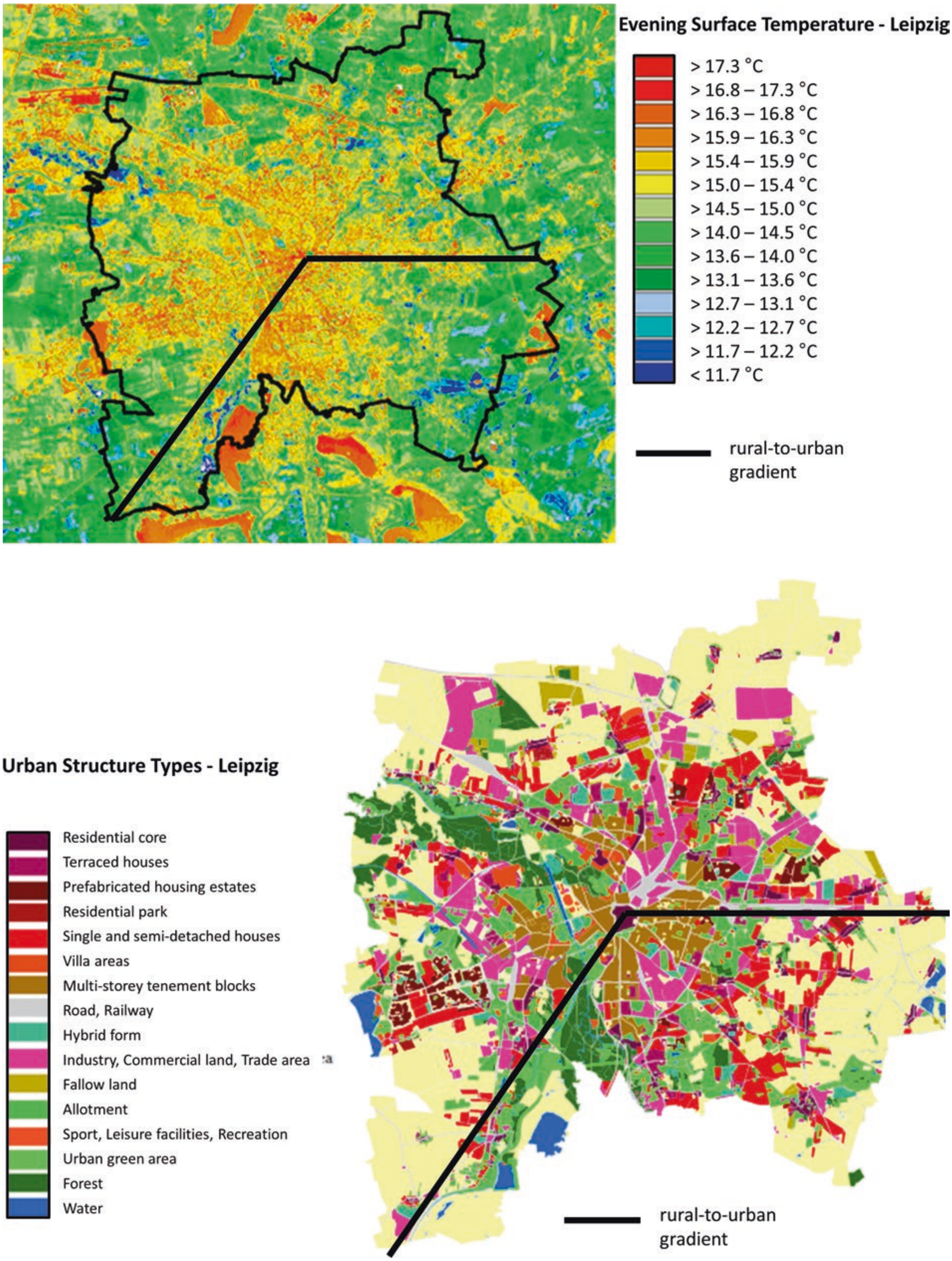

Fig. 23.5 (**a**, **b**) Rural-to-urban gradient (black line) of the evening land surface temperature in Leipzig showing a clear temperature increase from the outer, less surfaced, to the inner, highly sealed, parts of the city in relation to the land-use structure (lower part of the figure, based on Weber [14]; *with permission*)

A regression analysis showed a clear heat reduction at green spaces (parks, cemeteries, urban floodplain forest) in the morning and at night time [11]. For lawns and meadows, this heat regulation could not be found, so peripheral agricultural fields and shallow inland waters also appear warmer than inner-urban green spaces (Fig. 23.5).

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24 How to Reconcile the Ecosystem Service of Regulating the Microclimate with Urban Planning Projects on Brownfields? The Case Study Bayerischer Bahnhof in Leipzig, Germany

Florian Koch, Uwe Schlink, Lars Bilke, and Carolin Helbig

24.1 Brownfield Re-Use: Are Ecosystem Services at Risk?

The reuse of inner-city brownfields as urban planning strategy gains importance in the light of growing urban population figures and the need to think about more sustainable forms of urban development, as, for example, stated in UN HABITAT's New Urban Agenda [1]. Brownfields are sites that have been affected by the former uses of the sites and surrounding land; they are derelict and underused; they may have real or perceived contamination problems; they are mainly in developed urban areas; and they require intervention to bring them back to beneficial use [2]. In urban areas, characterized by high population density, mixed uses, and developed infrastructure systems, vacant lots such as brownfields represent potential areas for development. Areas that are located in the core of the urban area and surrounded by built structures (i.e., the inner-city) are of especially high interest for urban planning. Through their reuse (also labeled as "redevelopment" or "revitalization"), the amount of land-use change from agricultural land to suburban areas, or from forests to urbanized land, can be reduced. This densification of existing urban structures aims at reducing negative environmental effects of urban sprawl [3]. German planning law therefore champions inner development before outer development ("Innenentwicklung vor Außenentwicklung"), codified in §1a (2) German building law code [4]: New building projects should be realized preferably on sites that are located within the urban pattern. While it is acknowledged that brownfield revitalization can limit urban sprawl and therefore contribute to a more sustainable urban development, its effects on urban ecosystem services require further analyses.

Urban areas in general provide a variety of ecosystem services including local climate regulation, recreation and biodiversity potential, food supply, above-ground carbon storage, reductions in air pollution, and enhanced public ecological knowledge. They also have direct health benefits such as reducing the prevalence of early childhood asthmas [5].

Which ecosystem services are addressed? Regulating the microclimate.

What is the research question addressed? What happens to the Ecosystem Service of regulating the microclimate when new construction is realized on an inner-city brownfield site?

Which method has been applied? Microclimate modelling and visualization.

Review of urban design concepts.

What is the main result? The re-use of the brownfield area does not necessarily limit the regulation of the microclimate. Due to the interplay between the new park-like greenspace and the location and design of the planned construction, positive as well as negative effects on the urban microclimate appear.

What is concluded, recommended? Adequate urban design structures in combination with green areas can help to maintain the regulation effects of former brownfield sites even after new construction is realized.

Most of these ecosystem services can also be found on brownfields; examples include the ecosystem service of local climate regulation [6], providing ecological habitats, managing urban stormwater, and the provision of food in urban gardening projects on brownfields [7].

Little is known, however, about the effect of the *reuse* of brownfields on ecosystem services. Is urbanization (here understood as the construction of new buildings on vacant land) in general an anthropogenic second order risk (see introductory Chap. 1) for ecosystem services, acknowledging that urban densification processes can pose a threat to urban green space [8]? This question is related to the *paradox of the compact city* [9]: Compact and dense urban structures have the most negative balance concerning various

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urban ecosystem services, but are at the same time favored by urban planners for their energy efficiency, their potential to create innovation and social cohesion, and to implement efficient public transport [10].

In this chapter we aim to analyze this paradox empirically. We use, as examples, the brownfield and the surrounding neighborhoods of Bayerischer Bahnhof and the ecosystem service of regulating the local climate. This chapter summarizes some of the results of our work on compact and cool cities and the use of micro-climate modeling [11]. Our objective is to compare the current situation with a potential revitalization scenario. It is beyond the scope of this article to assess the effects of urban land-use changes on further ecosystem services, even though we are in line with Baro et al. in acknowledging the value of such more comprehensive studies for urban planning [12].

24.2 Methods

Thermal comfort and microclimate are strongly modified by the urban structure. To study the impact of land-use changes, ENVI-met [13] calculations were made, which simulate the interactions between different urban surfaces (asphalt, brick, sand, concrete, granite, grass, hedges, forest) and the atmospheric boundary layer. ENVI-met is a three-dimensional non-hydrostatic model together with a vegetation model, an atmosphere model including radiative transfer model, and a one-dimensional soil model [14]. The user of ENVI-met must implement the geometry of the study area into the area-input file which, finally, represents buildings, plants, and different surfaces (for example, see Fig. 24.1). In addition, there is a file with start and boundary values for the iterative solution of the partial differential equations. In ENVI-met it is

Fig. 24.1 Area input data illustrating the current land-use situation around the Bayerischer Bahnhof for modelling with ENVI-met. Note that this region is dominated by a large brownfield with uncultivated vegetation (meadows, lines of trees; plotted in green color) surrounded by residential blocks (in grey color)

possible to compare the current situation with the situation after revitalization. ENVI-met simulation results are analyzed with two- and three-dimensional scientific software (Leonardo for ENVI-met and ParaView [15], respectively). Multi-variate three-dimensional result data can be explored with a self-developed ENVI-met ParaView-plugin [16], which helps to explore complex wind-flow behavior using a Virtual Reality display system [17, 18]. Simulations of ENVI-met might be limited in their relevance to a real weather situation, as the software runs with boundary values, but is not dynamically linked to weather changes in the surroundings of the simulated area.

The simulations on the regulations of the local climate have been done for 21 July 2015, which was a representative day for hot summer weather in Leipzig (maximum temperatures exceeding 30 °C, minimum temperatures near 20 °C). Two different land-use scenarios have been implemented: The first scenario represented the current situation, (mainly vacant area with succession vegetation; see Fig. 24.1 for the representation of input data in ENVI-met). The second scenario implemented the planned urban design concept. The modelling was done for the three wind directions, east, west, and south, and for daytime (1 p.m.) and nighttime (11 p.m.). The calculated spatial distribution of air temperature 1.5 m above ground was considered in the following comparative discussion.

24.3 Case Study Bayerischer Bahnhof, Leipzig

Leipzig, located in Eastern Germany, encountered a long period of population decline and deindustrialization after the political turnaround in 1989. As a consequence, the amount of abandoned areas increased and brownfields were spread all over the city. At the start of the 2000s, Leipzig counted around 3000 brownfields with a total of 900 ha, ranging from smaller to big industrial and commercial lots [19]. During this time, urban planning instruments were implemented in the frame of the Stadtumbau-Ost national funding scheme to maintain the inner-city structures despite a declining population: low-density housing (e.g., townhouses), renaturation of brownfields, and renovation of vacant housing. Leipzig has become a forerunner city concerning the handling of brownfields: innovative interim use strategies or the creation of urban forests were developed in order to deal with an increasing number of brownfields. Since the first decade of the 2000s, the demographic development of Leipzig has changed and population is increasing at a fast pace. This growth needs new spaces and makes interim uses for brownfields less important. New neighborhoods for more than 10,000 inhabitants are planned in inner-city locations. The municipal masterplan states that the provision of new expansion areas is foremost to be found within the urban pattern [20]. Brownfields need to be redeveloped to reduce urban sprawl and to re-densify inner-city areas. On several inner-city brownfields reuse projects are foreseen mainly as a combination between residential and commercial uses.

One of these areas is Bayerischer Bahnhof, a former railway area with around 40 ha which hasn't been used since the beginning of the 2000s and is one of the biggest brownfields in Leipzig (see Fig. 24.2a, b). The main part of the area consists of succession vegetation, which emerged unplanned because of a decade of nonuse. The vegetation consists mainly of ruderal vegetation [21]. The City of Leipzig, however, also planted trees with a height of 5–10 m on the area as part of an interim-use strategy called "Urban forest".

The area is hardly accessible as fences were constructed to prevent entrance to the area. A recent study has pointed out that Bayerischer Bahnhof is primarily used by dog-owners to walk their dogs [21]. Increasing population and its attractive location, close to the city center (see Fig. 24.3), raised interest in this area. An urban design competition took place in 2010 and the winning concept by Wessendorf and Atelier Loidl foresees construction of new residential and commercial buildings. Only a minor (yet considerable) part of the area is dedicated for construction. The major part of the area is planned to be a park-like greenspace. The area has been bought by a private developer who intends to revitalize the area according to the guidelines of the winning design concept (Fig. 24.4).

Until now it is not clear when the concept will be implemented due to land property aspects and planning law issues. But while some changes concerning details of the urban design may occur, it is not likely that the whole concept will be changed.

24.4 Results

The modelling revealed that changes in the current situation of the brownfield have effects not only on the area itself, but also influence nearby neighborhoods (*see* Koch et al. [11] for a more detailed description of the results). Comparing micrometeorological simulations for the current land-use situation (Fig. 24.1) with simulations based on the design concept (Fig. 24.4), we assess the revitalization impact on microclimate in an ex ante approach. Atmospheric conditions will become modified in the following ways:

(a) During daytime in a south-wind direction, the new design would lead to partly increasing temperatures on the site. Mainly in the southern part of the new constructions, places with higher temperature will appear due to the new houses, which act as obstacles and prevent cooling air from entering (Fig. 24.5). Nevertheless, the new construction will also lead to cooling effects. This happens on the site itself. In addition, the neighboring northern part will be cooler (up to 0.6 K lower) due to changing wind patterns. If we compare this situation

Fig. 24.2 (**a**, **b**) Current state of the brownfield area around Bayerischer Bahnhof (Photos: F. Koch)

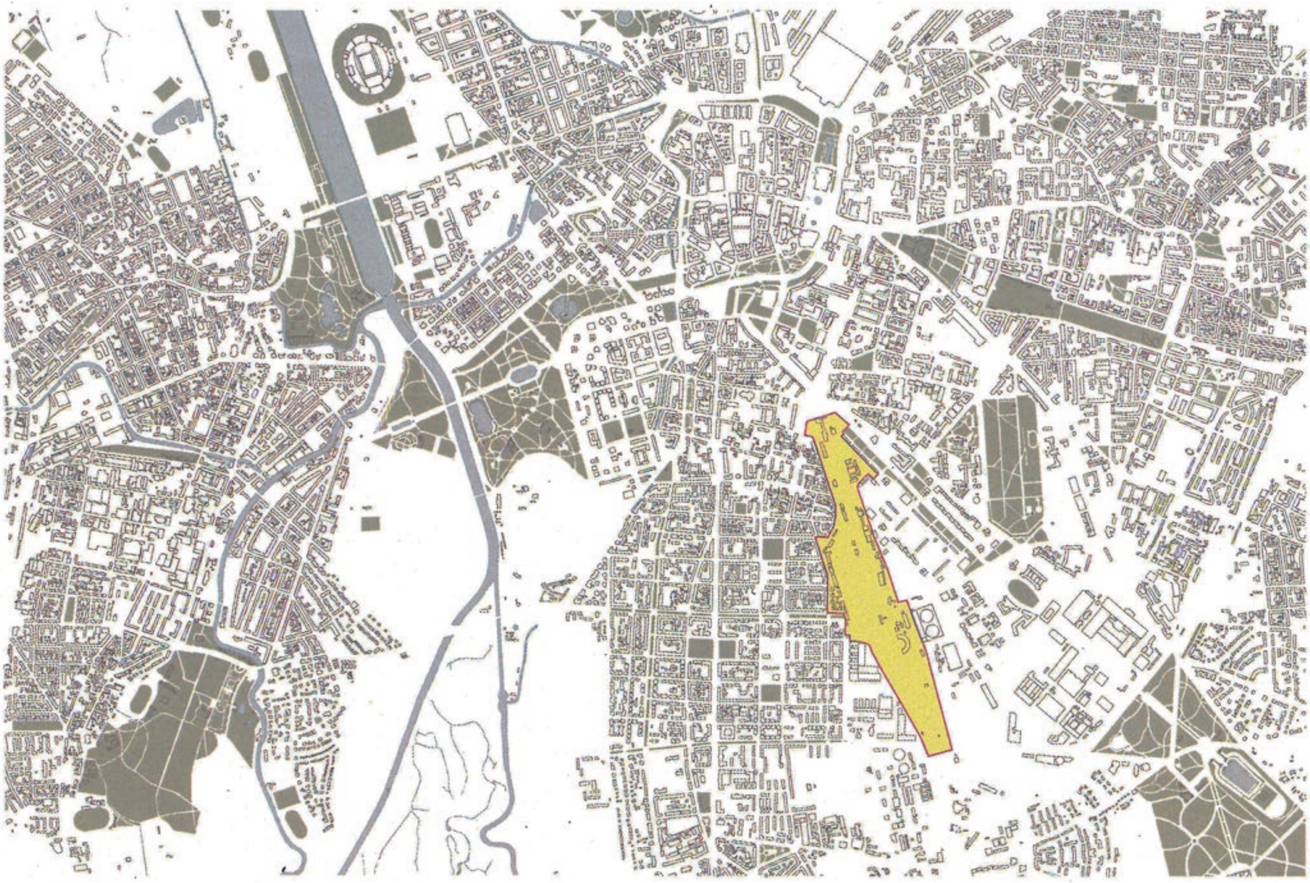

Fig. 24.3 The inner city of Leipzig with the Bayerischer Bahnhof area indicated in yellow [25]

with other wind direction scenarios, similar effects appear. For example, in an east-wind scenario, the cooling effects will be mainly in the western part of the brownfield (Fig. 24.6). The newly constructed blocks cause changing wind streams (Fig. 24.7) which result in cooling effects for the existing buildings and lead to a temperature reduction of 0.5–1.0 K. Analogously, west-wind causes cooling effects mainly in the eastern part of the area (not shown here).

(b) During the night, the new buildings will mostly increase air temperatures by up to 0.5 K. This pattern is fundamentally different from the cooling induced by the new buildings during daytime. For example, at 11 p.m. in a south-wind scenario, the area itself as well as the bordering northern part encounter temperatures increased by up to 1 K (not shown here). A similar situation appears in the east- and west-wind scenario: East-wind (Fig. 24.8) causes higher temperatures in the western part of the area and the bordering neighborhood, and west-wind leads to increasing temperatures in the eastern part of the area and the building blocks located at the eastern fringe of the area.

(c) The local wind conditions change fundamentally (Fig. 24.9). The newly constructed houses make the winds form jets along the streets and canyons between the new houses, and this is continued to the surrounding quarters and affects the comfort of their inhabitants.

These results show the twofold impact of the revitalization of the brownfield on the local climate: During the daytime, a cooling effect can be identified, not only on the brownfield itself but also for the bordering neighborhoods. During nighttime, the situation changes, and temperature increases are visible on the area and the surrounding neighborhoods.

24.5 Conclusion and Outlook

Our research shows that the reuse of the brownfield does not necessarily limit the regulation of the microclimate. Due to the interplay between the new park-like greenspace and the location and design of the planned construction, positive as well as negative effects on climate regulation will appear (depending on whether daytime or nighttime is being considered). These results are comparable with other local-climate related research on the brownfields: For the brownfield Tempelhofer Feld in Berlin, research has demonstrated that

Fig. 24.4 Winning concept of Jörg Wessendorf and Atelier Loidl for the planning competition Bayerischer Bahnhof [19]

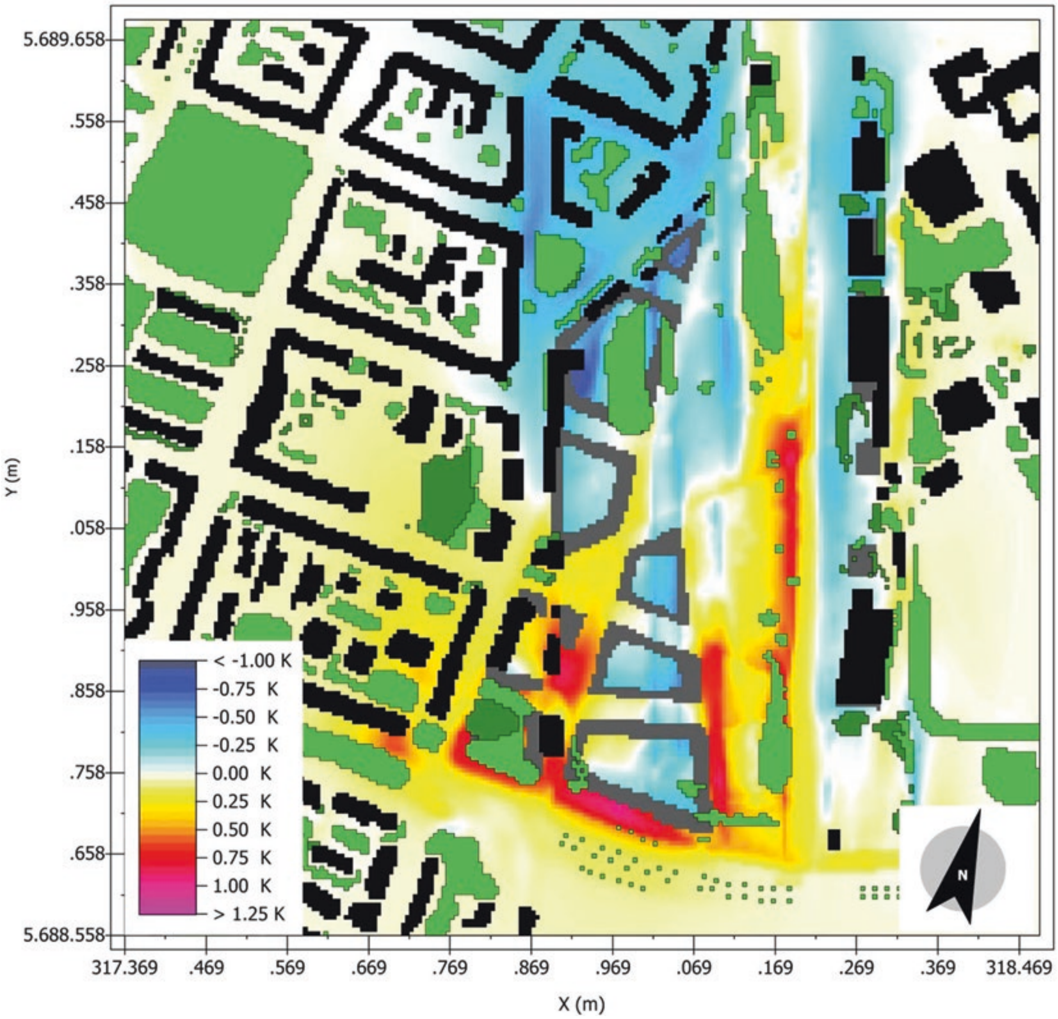

Fig. 24.5 Differences of daytime air temperatures (1.5 m above ground, 1 p.m.) between revitalized and current land-use scenarios. UTM coordinates (zone N32), vegetation (green color), existing buildings (black) and new buildings (grey) are plotted. Simulation is valid for 27 July 2015, 13:00 and wind from South (193°). Wind speed 4 m/s at 10 m above ground; roughness length 0.1 m; initial temperature of atmosphere 20 °C, 38% relative humidity 2 m above ground, specific humidity 5.5 g/kg at 2500 m above ground, initial soil temperature 20 °C, albedo walls (0.3) roofs (0.4), building inside temperature 22 °C, heat transmission walls (1.94 W/m^2 K) roofs (6.0 W/m^2 K), data update every 30 s [11]

new construction on vacant land only marginally influences thermal comfort [22]. Studies on Manchester, Freiburg, and Beijing show the impact of different land-use and vegetation scenarios on the microclimate [23–26].

The legal requirements of the German planning law to prioritize reuse of inner-city brownfields in ways that decreases urban sprawl does not automatically bring an end to the microclimate regulation effect of brownfields. Adequate urban design structures in combination with green areas can help to maintain the cooling effects of the site even after revitalization. The ex ante approach we applied can be helpful in elaborating urban design concepts for brownfields and for planning in dense urban structures in general. It shows, prior to starting a revitalization project, whether and how the ecosystem service of microclimate regulation can be reconciled with new construction. Through this approach, adjustments to the design concept can easily be implemented before the construction phase. The ex ante approach helps with decision-making processes on the different societal responses to ecosystem services risks (avoidance, trade-off-management, adaptation and transformation; for details see conceptual framework of the Atlas of Ecosystem Services (Chap. 1)). Our approach can be enlarged by analyzing the effects of creative design ideas such as vertical and roof-top green spaces on the local microclimate and the potential to overcome the paradox of the compact city structures.

While the value of brownfields for the provisioning of different ecosystem services has been highlighted in prior case studies, and research has been conducted on how transforming

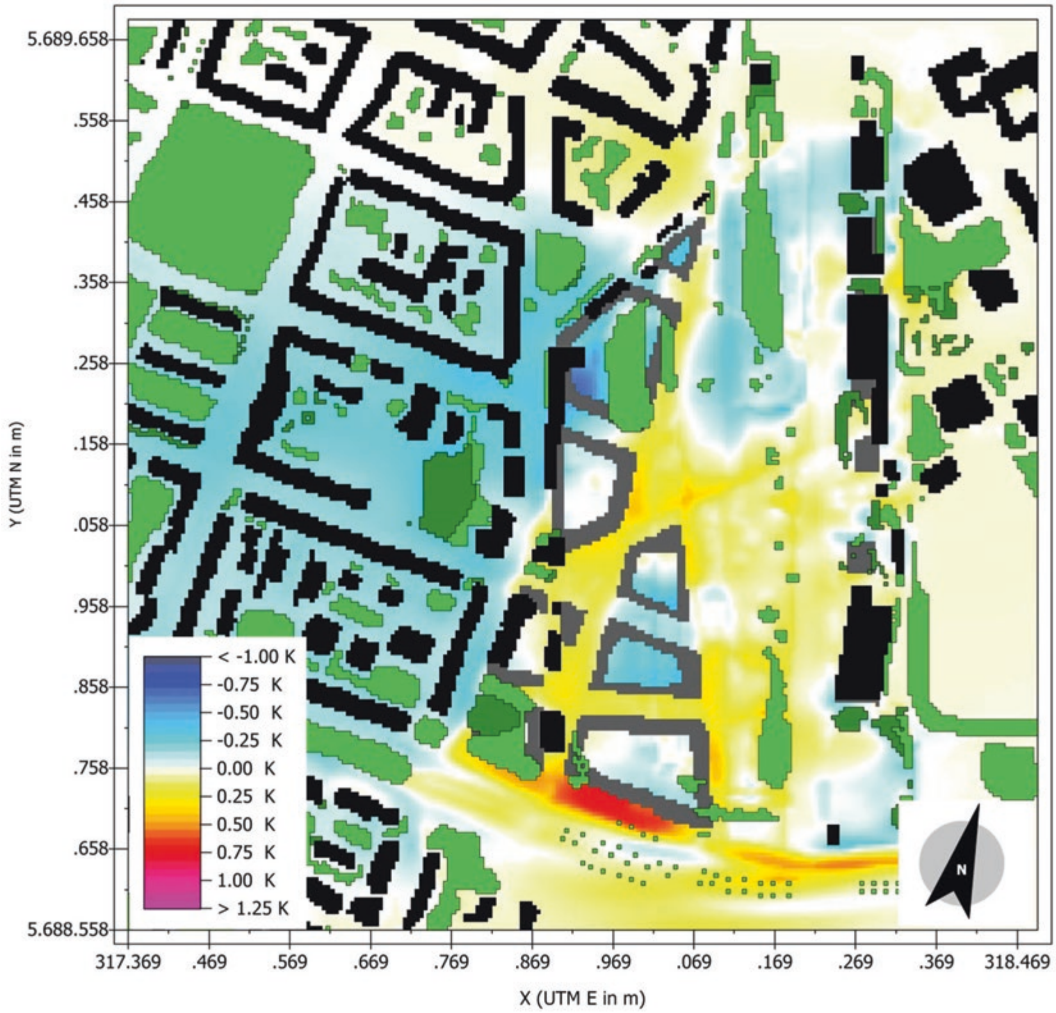

Fig. 24.6 Differences of daytime air temperatures (1.5 m above ground 1 pm) between revitalized and current land-use scenarios. Same parameters as in Fig. 24.5, but for wind from the east (103°) [11]

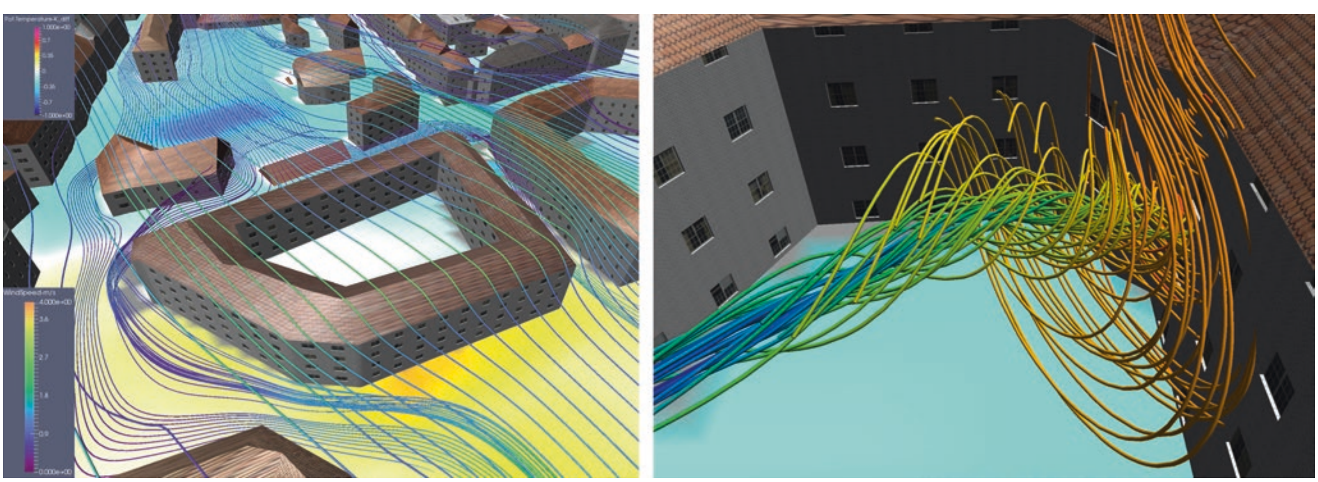

Fig. 24.7 Warmer air is lifted to higher levels by new building facades; expanded air originating from wind tunnels cools down areas behind new buildings, colors on the surface: differences of air temperature between scenarios at 1.5 m height, streamline colors: wind velocity (left). Turbulences (although with small velocities <0.1 m/s) can occur leeward in inner courtyards (right) [11]

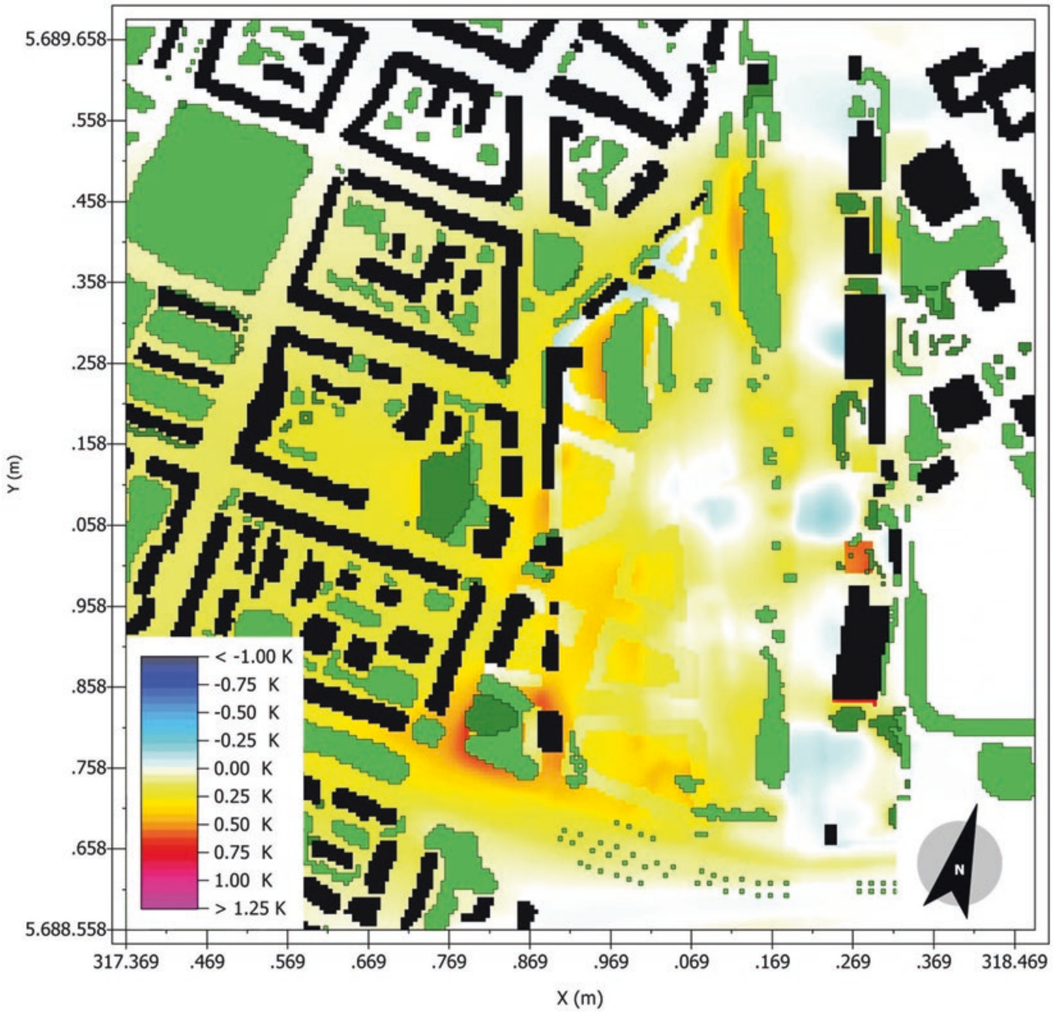

Fig. 24.8 Differences of nighttime air temperatures (1.5 m above ground, 11 p.m.) between revitalized and current land-use scenarios. Same parameters as in Fig. 24.5, but for wind from the east (103°) [11]

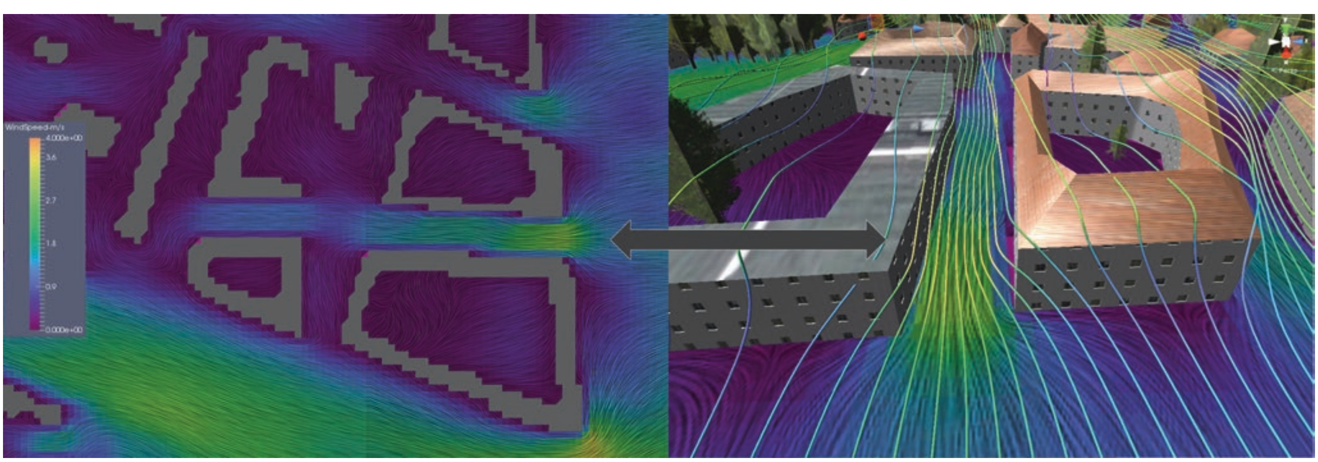

Fig. 24.9 Wind tunnel effects in the revitalized region. Wind tunnel patterns are clearly identifiable when two building blocks are perpendicular to the wind direction (east-wind in this case). Advanced visualisation algorithms such as line integral convolution (left) and 3D streamlines (right) help to identify areas of interest in complex data sets. Colors: wind velocities at 1.5 m height (left) and at the streamline locations (right) [11]

brownfields to green spaces affects ecosystem services, we are only at the beginning when it comes to understanding the relationship between ecosystem services risks and ongoing urbanization. Issues such as how cultural ecosystem services are influenced by reusing brownfields, what effects on biodiversity may appear in revitalization processes, as well as the impact of new urban planning projects on ecosystem disservices, need to be analyzed. This requires an understanding of constant land-use, demographic, and building environment changes as key elements of cities, rather than taking built and social structures as given values.

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25 Urban Green Infrastructure in Support of Ecosystem Services in a Highly Dynamic South American City: A Multi-Scale Assessment of Santiago de Chile

Ellen Banzhaf, Francisco de la Barrera, and Sonia Reyes-Paecke

25.1 Green Infrastructure as Pillar for Ecosystem Services in Fast Growing Cities

Urban green infrastructure contributes to the health and quality of life for human beings [1]. Green infrastructure covers all kinds of vegetated urban spaces, with diverse forms and structures performing multiple functions [2]. Characterised by an interconnected network of natural and semi-natural areas with other environmental features, it is one of the major suppliers of urban ecosystem services in terms of carbon sequestration, climate mitigation, and various cultural services. Public green spaces are one of the most important elements to secure urban ecosystem services and are considered to be public goods that allow free access to all citizens and represent pockets of nature for all residents. Vegetation cover, however, is a broad underlying concept with a diverse structural pattern comprising root penetration, ramification, foliation, and including different types such as grasses, herbs, shrubs, and trees [3, 4].

The aim of this chapter is to illustrate pressures on and needs for green infrastructure in the frame of urban growth and climate change using Santiago de Chile as our showcase study. We demonstrate the significance of urban green infrastructure for ecosystem services and their implicit provision at different spatial and administrative scales, from the entire metropolis to single municipalities and, finally, for selected neighborhoods along the Andean foothills.

Due to their incessant sprawl, South American metropolises put serious pressure on regional ecosystems. Such highly dynamic urban growth patterns trigger changes in land use and land cover. The inferred loss in natural ecosystems, in turn, leads to a deficiency of key ecosystem services, with a direct impact on human well-being. Ecosystem services are most needed in densely populated areas such as cities.

With an area of 850 km^2 and almost seven million inhabitants, Santiago is, relative to other South American cities, a medium-sized metropolis. It experiences the development of rapid urbanization which is significant in terms of 1) the expansion of the built-up area into agricultural land with fertile soils and into the Andean foothills; and 2) the increase in urban population with the peculiarity of socio-spatial differentiation that takes place at large scale. In contrast to European cities, this socio-economic situation is rather typical for South America.

Which ecosystem services are addressed? Urban ecosystem services: cooling effects of shading plants, climate mitigation, regulating run-off, improving air quality, carbon sequestration, and various cultural services such as recreation and sense of place, aesthetics.

What is the research question addressed? How does the socio-spatial differentiation influence urban green infrastructure and its supporting ecosystem services in a metropolitan area?

Which method has been applied? Literature review, remote sensing techniques, field work, interviews, and observation.

What is the main result? Urban ecosystem services need to be differentiated according to the societal impacts at various levels. Green infrastructure dominated by native species has a higher resilience to climate disturbances, which is important in the context of climate change.

What is concluded, recommended? Ecosystem services provided by urban green infrastructure have been poorly planned by authorities. Results recommend stakeholders to maintain and manage small- and medium-sized green spaces sustainably as well as to ensure preservation and further cultivation of local green infrastructure as urban forests and gardens with climate-adapted native species. In this respect, the regional government should ensure the preservation of well-conserved rural landscapes and the development and maintenance of green spaces.

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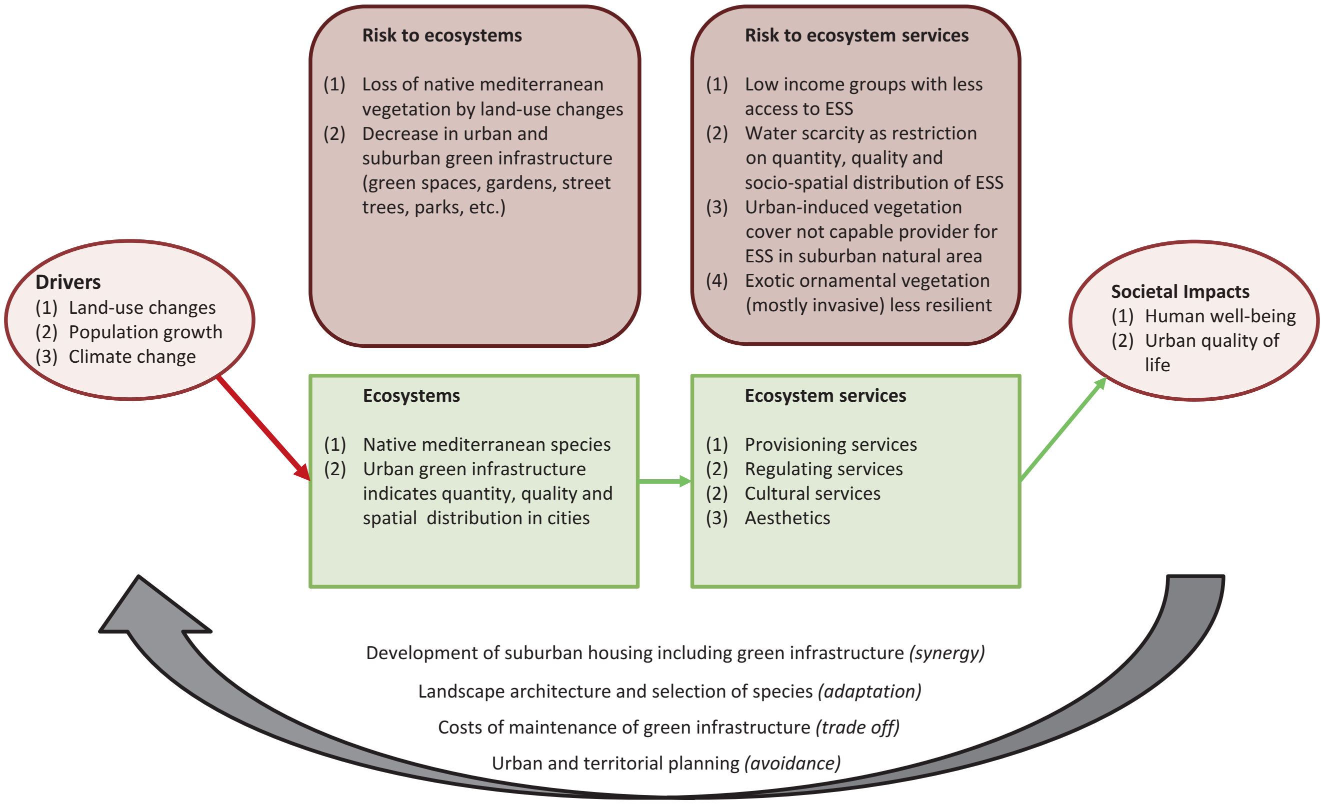

Fig. 25.1 Framework for assessing risk to ecosystems and ecosystem services in a South American metropolitan area

Santiago is characterized by a Mediterranean climate with annual winter rainfall of around 330 mm and a long, dry, and warm summer. Climate change projections indicate a 40% decrease in annual rainfall by 2050 and increasing average temperatures in the hottest months [5]. Hence, urban growth and climate change are major drivers influencing green infrastructure and related ecosystem services (Fig. 25.1) [6]. Concurrently, Santiago is in the middle of a hotspot of biodiversity determined by Mediterranean ecosystems. Globally, these ecosystems occupy around 5% of terrestrial surface, but contain almost 25% of the earth's biodiversity (Fig. 25.2). Being well adapted to frequent droughts and high solar radiation (Fig. 25.3), they contrast the urban vegetation dominated by exotic plants and requiring irrigation most of the year (Fig. 25.4) [7].

25.2 Ecosystems and Their Services at Risk

For Santiago and other South American cities, the main risk to natural ecosystems is the strong process of urbanization, which goes hand in hand with the loss of native vegetation due to urban sprawl. Related land-use changes consequently lead to deterioration of pristine sites and harm wildlife habitats. Beyond that, densification processes lead to the fragmenting of green infrastructure and places the urban ecosystem at risk by diminishing green spaces. Similarly, large urban parks and ecological preserves are frequently fragmented by transport and energy infrastructure.

Local risks to ecosystem services, such as provisioning, regulating, and cultural services as well as aesthetics, which differ strongly due to socio-spatial differentiation, are of utmost concern. Dependent on the socio-spatial neighborhood, public green spaces can be composed of bare soil and suffer from a lack of vegetation. At the same time, such green spaces have a high-use intensity because deprived neighborhoods do not possess private gardens or other privately owned green spaces (Figs. 25.1 and 25.5) [3, 8]. Apart from that, the urban-induced species are not as capable as native species of fulfilling ecosystem service provision in suburban natural areas [4]. Introduced species are mostly exotic, serve ornamental purposes, and are less resilient.

Solutions to the problem of how to evaluate the societal impacts need to be considered at various levels as illustrated in Fig. 25.1. A synergy in urban sustainable development would include green infrastructure. When adapting to the regional ecosystem, landscape architects would do best to select native species with less need for irrigation. A trade-off could be the costs of maintenance of green infrastructure for the benefit of human well-being. Avoidance of risks is in the hands of urban and territorial planning, which must consider environmental aspects.

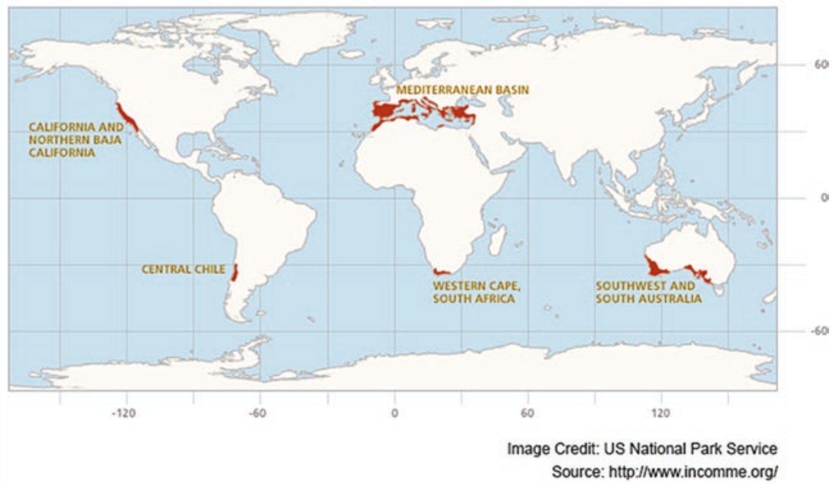

Fig. 25.2 Location of Mediterranean-climate ecosystems and Santiago de Chile in South America (left); Global coverage of Mediterranean-climate ecosystems (right). (*Used with permission of the* U.S. National Park Service.)

25.3 Urban Growth and Its Effects on Green Infrastructure at Metropolitan Scale

Remote sensing studies show an expansion of the built-up area by 110% (460 km^2), and a loss of 42% of green infrastructure (41 km^2) in the last decades (Fig. 25.6) [9].

The intense urban expansion has not been accompanied by the provision of public green spaces for residents of the new developments [10]. Although suburban municipalities may have a higher share of green infrastructure and a lower proportion of impervious surfaces than the urban core area, most vegetation cover is located on private lots, resulting in few poorly maintained public green spaces with little shading [3]. The loss of vegetation cover has had a strong impact on ecosystem services and has increased vulnerability of urban areas to disastrous weather events, especially flooding and landslides, the latter mainly along the Andean foothills.

25.4 Green Infrastructure Mirroring the Socio-Spatial Differentiation at Municipal Scale

In Santiago's municipalities, green infrastructure is positively correlated with residents' income level. Figure 25.5 illustrates the different proportions of green infrastructure within three municipalities depending on the income ratios of the inhabitants. Vitacura, one of the municipalities with the highest family income and low population density (40 inhabitants/ha) possesses the highest green infrastructure (40% of built-up area), mainly on private properties. La Florida is a middle-income municipality, with high population density (109 inhabitants/ha), and its total vegetation covers 26% of the built-up area. Cerro Navia, a low-income municipality, has the highest population density (166 inhabitants/ha) and lowest green infrastructure (15% of built-up area) of these three municipalities [3].

Fig. 25.3 Natural landscapes dominated by (**a**) forests and scrublands (**b**) (Photos by F. De la Barrera)

Fig. 25.4 (**a**, **b**), Urban parks dominated by lawn and exotic trees without shrubs (Photos by S. Reyes-Paecke)

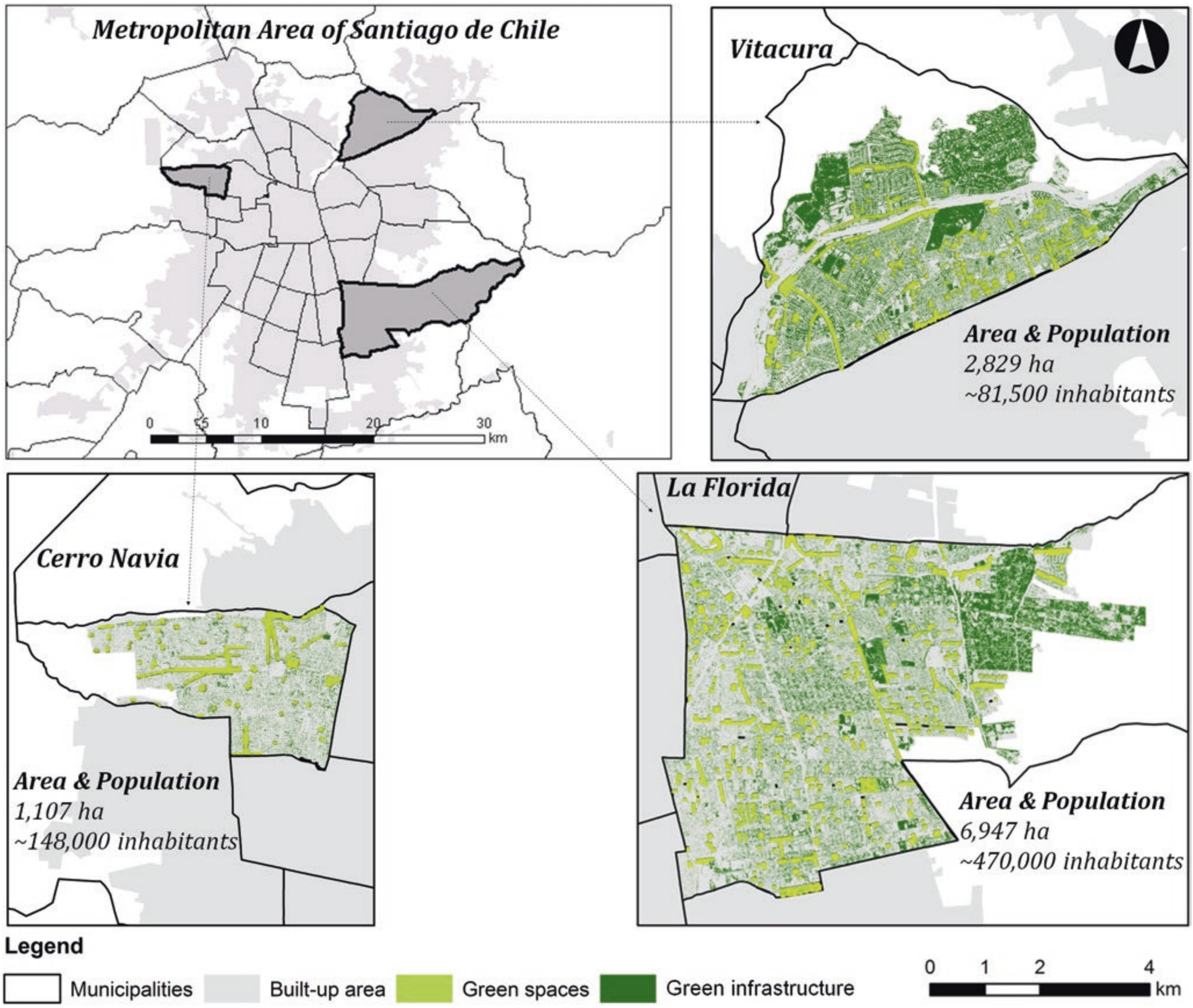

Fig. 25.5 Location and vegetation cover in three socio-economically differentiated municipalities (based on [3]). Data for built-up area and green infrastructure are calculated based on QuickBird satellite images; data for urban green spaces were recorded in own mapping; administrative data from National Statistics Institute of Chile (INE)

25.5 Green Infrastructure in Suburban Areas at Neighborhood Scale

Different rural ecosystems coexist where Santiago abuts the Andes Mountains. These include well-conserved ecosystems predominantly composed of native and dense scrublands and forests that are very well-adapted to prolonged dry seasons. In contrast, these also include degraded ecosystems that are composed of a mix of exotic plants and native vegetation, and which are much less dense as a consequence of perturbations. Urbanization can modify both types with contrasting consequences, creating green infrastructure with a novel structure of vegetation pattern (Fig. 25.7) [10].

In this regard, La Dehesa is a new suburban area located in a previously degraded ecosystem in the northeast of Santiago. It has experienced a low density of expensive housing, diversely furnished by green infrastructure with intensively irrigated exotic vegetation. Quite simply, ecosystem services are well-provided, such as cooling effects of shading plants, regulating run-off, and improved air quality at the local scale, which even secures high benefits of ecosystem services beyond the local neighborhood [4]. In contrast, urban development is planned in a privately-owned land (El Panul) with a well-conserved ecosystem and native vegetation in the east of Santiago. This has garnered attention because of its cultural ecosystem services (e.g., recreation and sense of place), and forces a reconsideration of its further development.

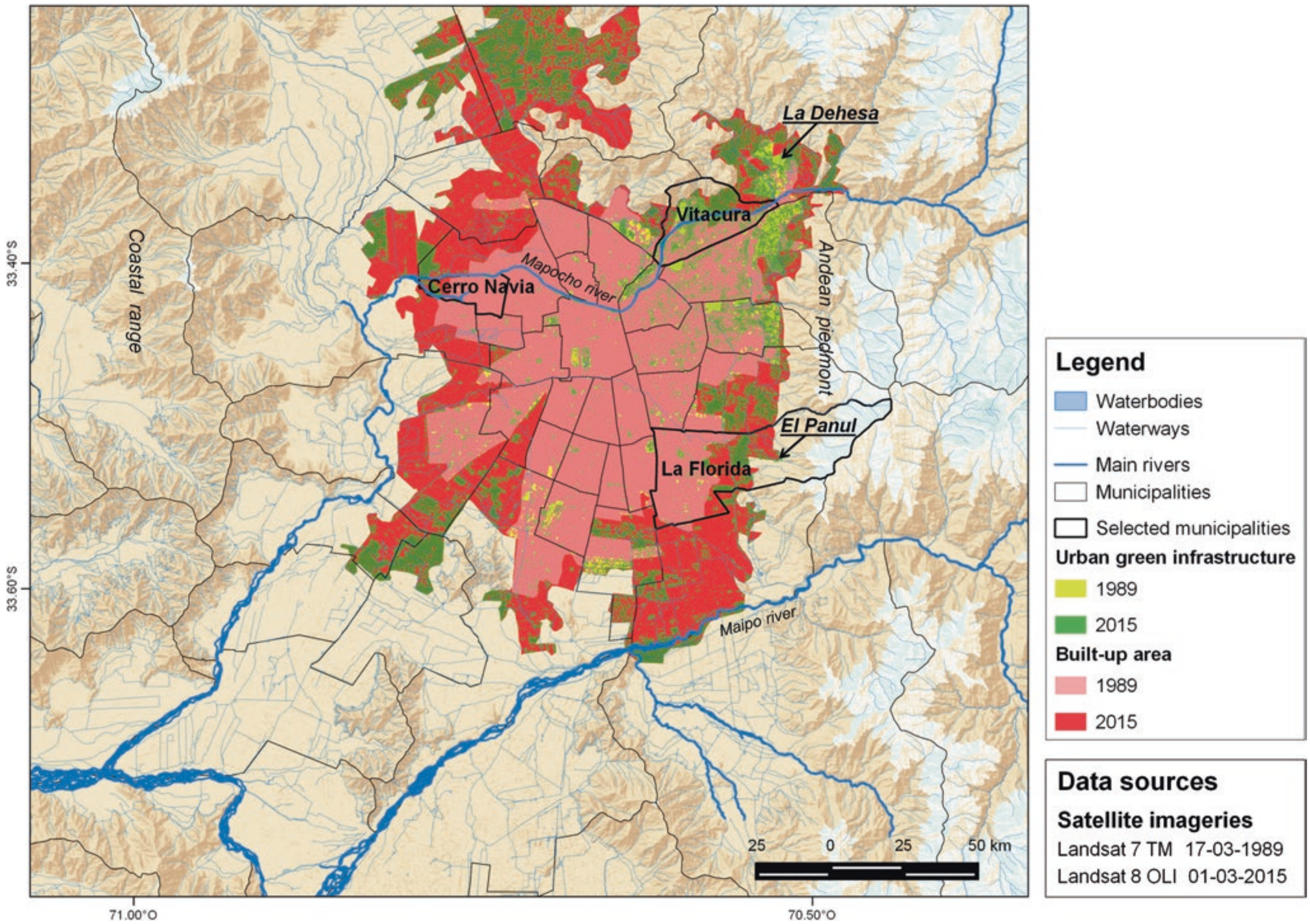

Fig. 25.6 Metropolitan area of Santiago: location and land-use changes from 1989 to 2015. Data for built-up area and urban green infrastructure are calculated based on Landsat satellite images; topographic base is derived from ASTER GDEM satellite images; administrative data from National Statistics Institute of Chile (INE); data for waters from Military Geographical Institute, Chile

This comparison suggests that the mere presence of vegetation (whether native or exotic) is positive for the supply of regulating ecosystem services. Beyond, residents benefit from public green spaces as cultural ecosystem services. Furthermore, green infrastructure dominated by native species more resilient to climate disturbances, which is important in the context of climate change.

25.6 Conclusions

In Santiago de Chile, ecosystem services provided by urban green infrastructure have been poorly planned by authorities. Consequently, ecosystem services are at risk because they strongly underlie processes of neo-liberal market mechanisms in which private agents transform rural environments into residential neighborhoods with very little policy-steering mechanisms. The existence of public green infrastructure can partially mitigate this decompensation and ensure the provision of ecosystem services. Results recommend that stakeholders maintain and manage the small- and medium-sized green spaces sustainably. Stakeholders should also ensure the preservation and further cultivation of local green infrastructure such as urban forests and gardens with climate-adapted native species. In this respect, the regional government should ensure the preservation of well-conserved rural landscapes and the development and maintenance of green spaces. From policy perspective, we advise policymakers to steer urban green infrastructure at different institutional levels, thereby ensuring a long-term provisioning of ecosystem services across local and regional scales. It is important to increase not only urban green infrastructure, but also the proportion of native species. Besides their conservation value and from an environmental health perspective, native species would contribute largely to cost savings attributed to the control of physical damages and prevention of diseases owing to the existence of green infrastructure.

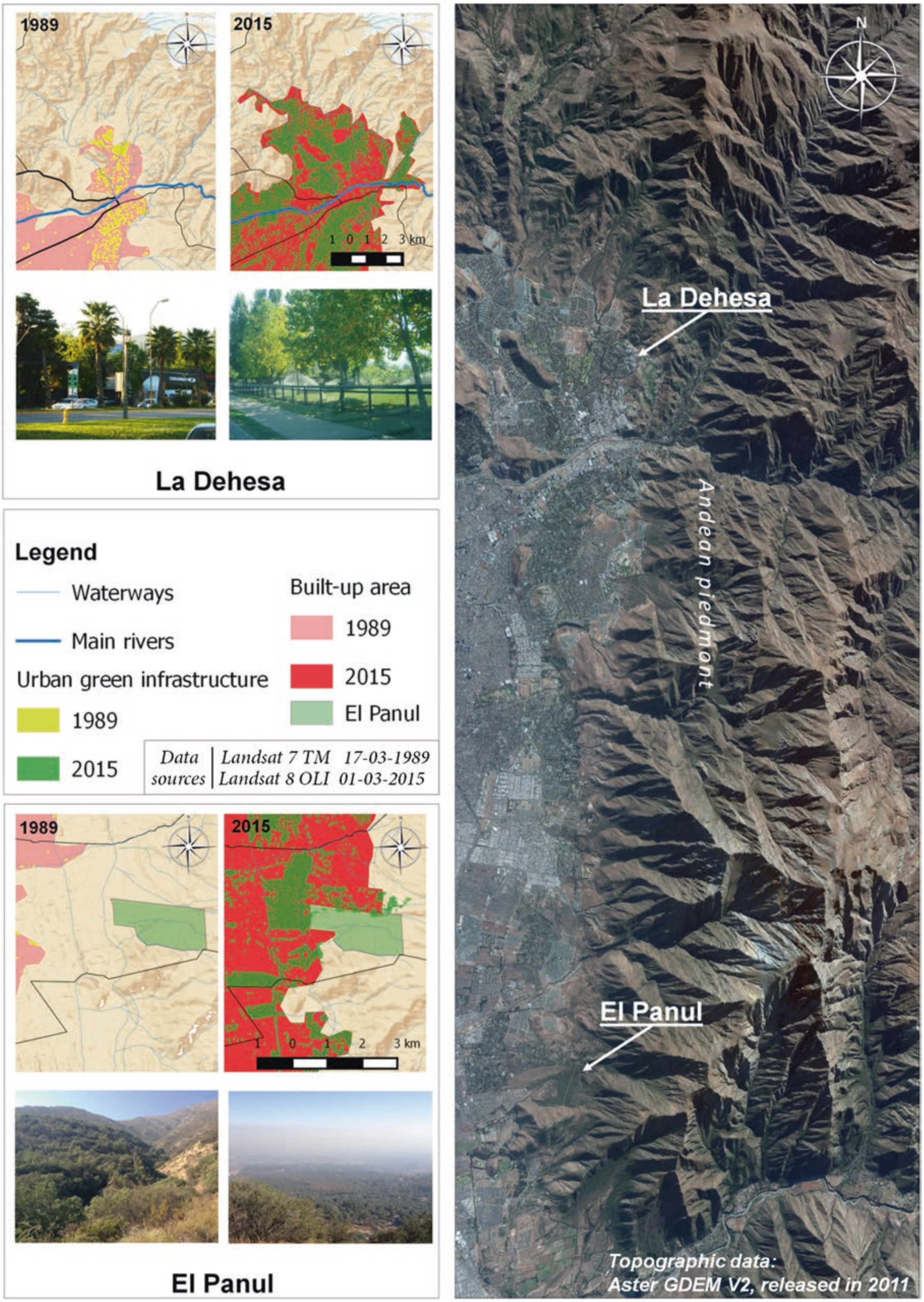

Fig. 25.7 Location and land-use changes of La Dehesa and El Panul. Photos by F. De la Barrera; data for built-up area and urban green infrastructure are calculated based on Landsat satellite images; topographic base is derived from ASTER GDEM satellite images; administrative data from National Statistics Institute of Chile (INE); data for waters from Military Geographical Institute, Chile

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26 Climate Regulation by Diverse Urban Green Spaces: Risks and Opportunities Related to Climate and Land Use Change

Sonja Knapp, Madhumitha Jaganmohan, and Nina Schwarz

26.1 Setting the Scene

The regulation of urban climate (called "climate regulation" here) by urban green spaces such as parks and forests is one of the main ecosystem services for urban areas [1], as it mitigates the urban heat island (UHI) effect [2] (Fig. 26.1). UHIs can negatively affect human health and wellbeing, especially in summer. Climate regulation will become even more important when adapting cities to climate change [3].

Urban green spaces not only have lower air temperatures within their boundaries, but this effect can also extend into the surrounding areas [4] (Fig. 26.2). Large irregularly shaped green spaces and especially forests in urban areas have a stronger cooling effect than small green spaces and parks [4]. The cooling effect is mainly due to evapotranspiration and shading by trees [5]. Evidence suggests that tree species differ in the strength of their cooling effect [6]. This raises the question whether optimal cooling effects can be best achieved by planting a high diversity of tree species or single "super-performing" tree species. In addition, cooling might not primarily depend on the diversity or identity of tree species but rather on the traits of trees, i.e., their morphological, physiological, biochemical, and phenological characteristics, as well as their diversity [7].

Our measurements of air temperature gradients for 60 green spaces (forests and parks; see Fig. 26.2 for one example) and their residential surroundings in the city of Leipzig, Germany [4, 8] suggest that the effects of tree diversity are less important for climate regulation than the size of green spaces or whether a green space is a forest or a park. Nevertheless, we were able to explain much more of the variation in temperature among green spaces and adjacent residential areas by including selected mean traits of trees (height and stem diameter) and these traits' diversity into our calculations. A higher diversity of these traits (which equals a more diverse vegetation structure, e.g., varying tree height) improved the cooling effects of parks, while species diversity did not.

Which ecosystem services are addressed? Climate regulation (a regulating service).

What is the research question addressed? How should urban green spaces be designed to provide climate regulation in the face of climate change? Does climate regulation by urban green spaces improve with higher tree diversity or is a single tree species sufficient?

Which method has been applied? We measured air temperature gradients for 60 green spaces and their residential surroundings in the city of Leipzig, Germany, and we determined tree species with their height and stem diameter in these green spaces.

What is the main result? Forests and large green spaces have higher cooling effects than parks and small green spaces. A more diverse vegetation structure improves the cooling effect of green spaces.

What is concluded, recommended? When planting trees, species' suitability for current and future climatic conditions should be considered. Within the pool of suitable species, trees should be selected in a way that ensures a high diversity of trees with relevance for climate regulation.

26.2 Climate Regulation at Risk

Climate regulation by urban green spaces and trees is at risk due to two direct drivers: First, urban green spaces are increasingly threatened by land-use change, specifically by high demands for residential land, which can increase with the growth of urban population (Fig. 26.3). Global estimates show that between 430,00 km^2 and 12,568,000 km^2 of non-urban land currently have high probabilities of urban expansion until 2030 (depending on the scenario applied; [9]). Thus, the amount of urbanized land will roughly triple until 2030 as compared to the year 2000 [10].

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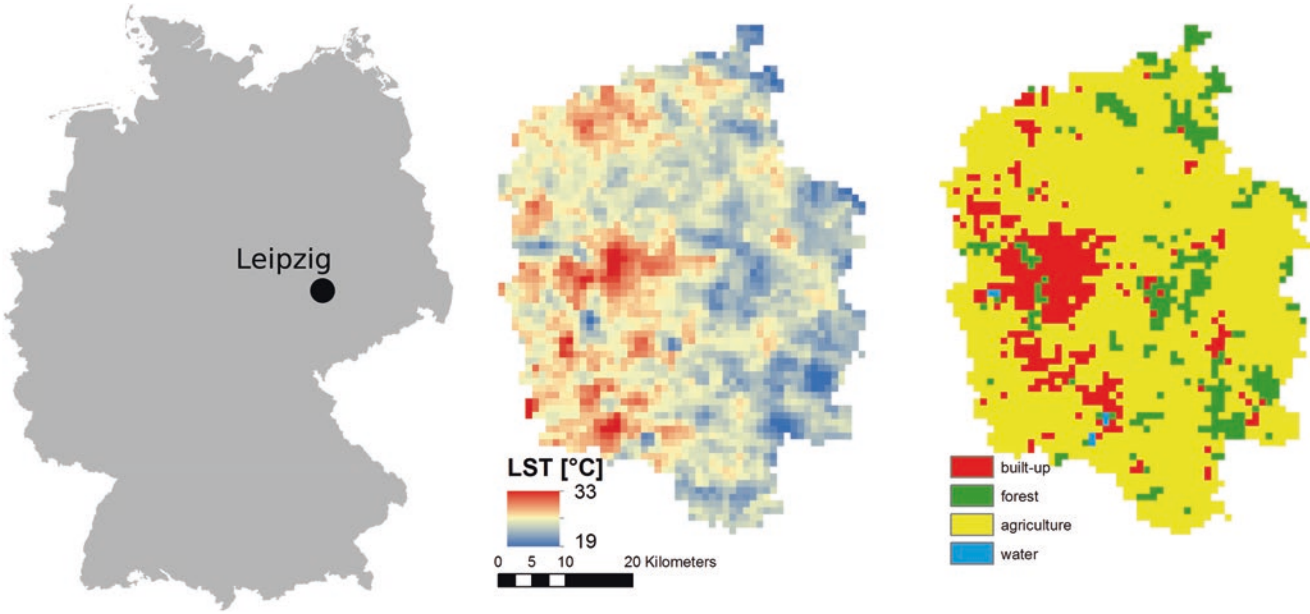

Fig. 26.1. Visualization of an urban heat island for the larger urban zone of Leipzig, Germany. The colors in the left map indicate mean daytime land surface temperatures (LST) in June, July, and August in 2001 derived from MODIS data (resolution 1 × 1 km²), while the right shows the underlying land cover using the classification of the International Geosphere-Biosphere Programme [15]. Comparing the two maps reveals that built-up areas are warmer than the surrounding agricultural landscape and the cooling effect of forests

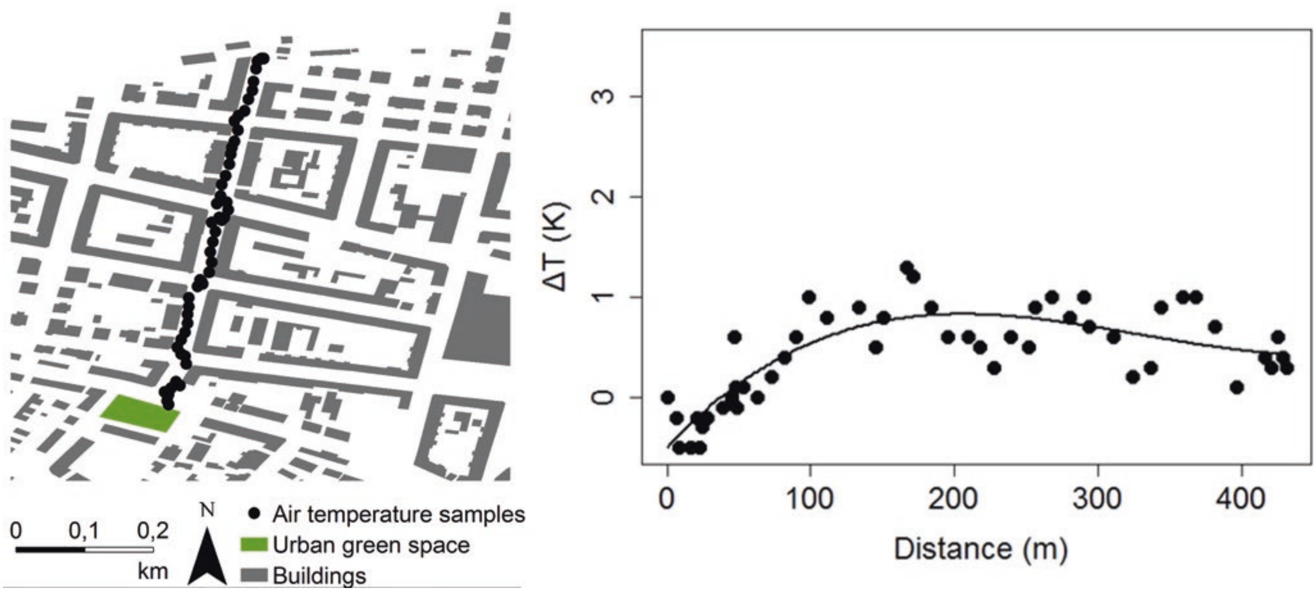

Fig. 26.2 Cooling effects of an exemplary urban green space onto the surrounding residential area in the city of Leipzig, Germany. Air temperature was measured about 1 m from the ground while walking from the green space into the nearby residential area. The map on the left gives the locations of air temperature measurements. The graph on the right displays measured air temperatures normalized as a temperature difference to the boundary of the green space. Air temperatures rise with increasing distance from the green space and then level off

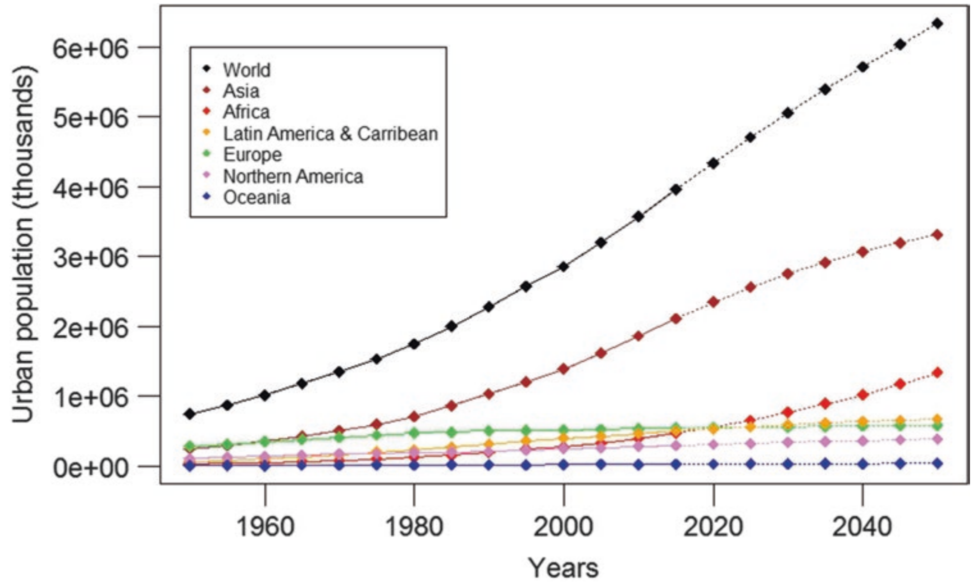

Fig. 26.3 Global increase in urban population between 1950 and 2050, differentiated by continents based on [16]. Solid lines symbolize observed values (1950–2015); dotted lines symbolize estimated values (2015–2050)

Second, rising temperatures (Fig. 26.4) influence the vegetation of urban green spaces: In the short-term, hot and dry summers can lead to a decrease in the potential of trees to regulate temperatures, as trees can only transpire if water is available [11]. In the long run, a change in existing tree species is needed [12], particularly in paved spaces and streets, catering to local needs and conditions. Moreover, the composition of tree species not cultivated or managed also changes due to changing climatic conditions, as urban vegetation generally does [13].

26.3 The Challenge for Urban Planning

The demand for housing and other types of urban development can lead to the degradation or loss of green spaces. For the planning and management of urban areas it is crucial to develop a network of green spaces (often called "green infrastructure") that is capable of protecting and even enhancing urban biodiversity and the provision of ecosystem services. As space is limited, green infrastructure should be as multifunctional as possible. Besides regulating urban climate, tree species and their diversity can, for example, provide a multitude of other ecosystem services, such as driving carbon and nutrient cycles [14].

When planting trees, species' suitability for current and future climatic conditions should be considered. Within the pool of suitable species, trees should be selected in a way that ensures a high diversity of trees with relevance for climate regulation (Fig. 26.5). Traits can help guide this selection. However, our empirical results also hint at potential trade-offs between diversity of tree species and selecting trees for their climate regulation potential, e.g., by maximizing certain traits. Increasing our knowledge base on these matters is crucial for managing existing urban green spaces, but also for climate adaptation plans or environmental impact assessment of plans and projects, among others.

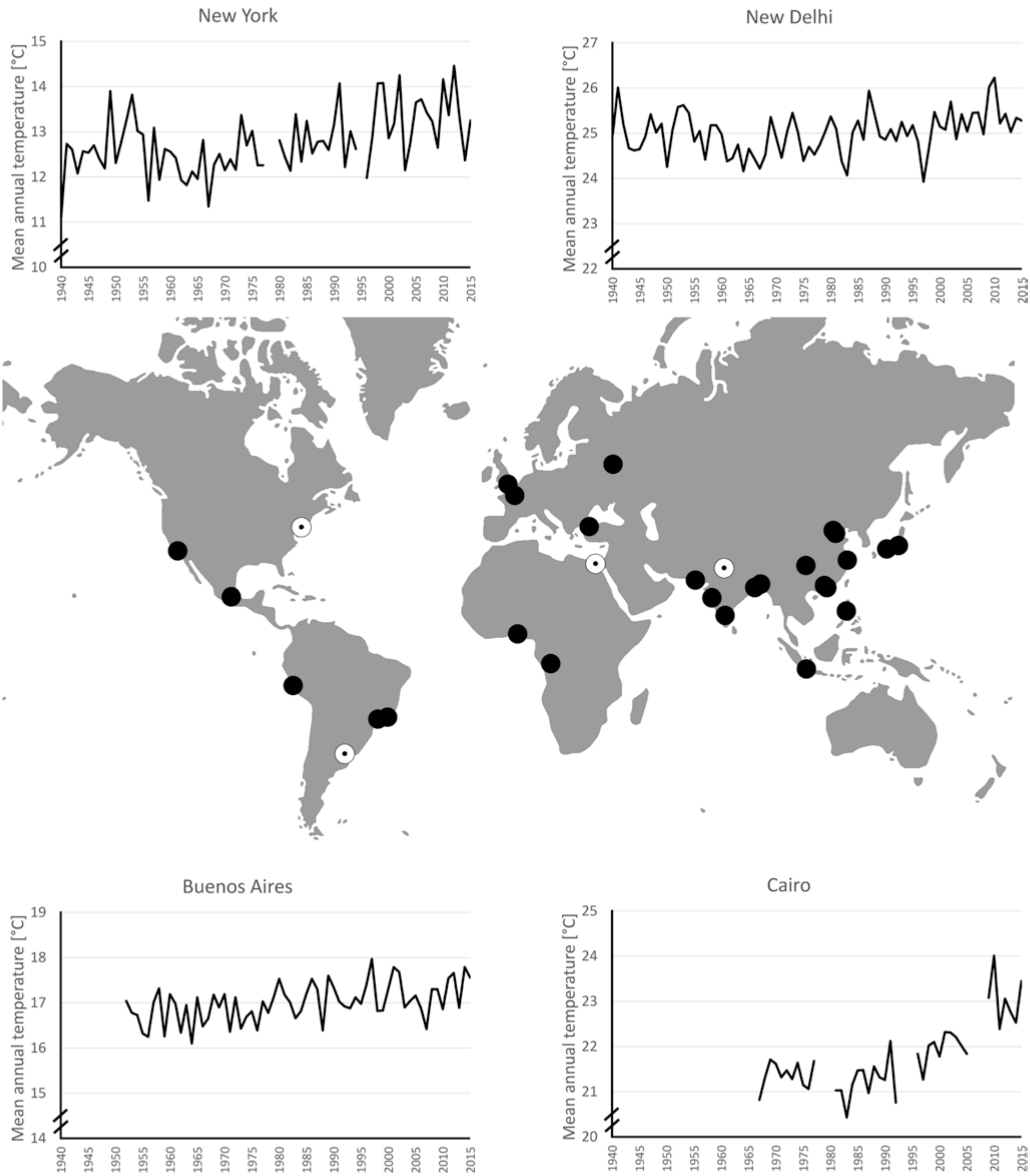

Fig. 26.4 Distribution of the world's 30 largest cities in 2015 and exemplary mean annual temperature dynamics at urban weather stations (data sources: [17], open street map data)

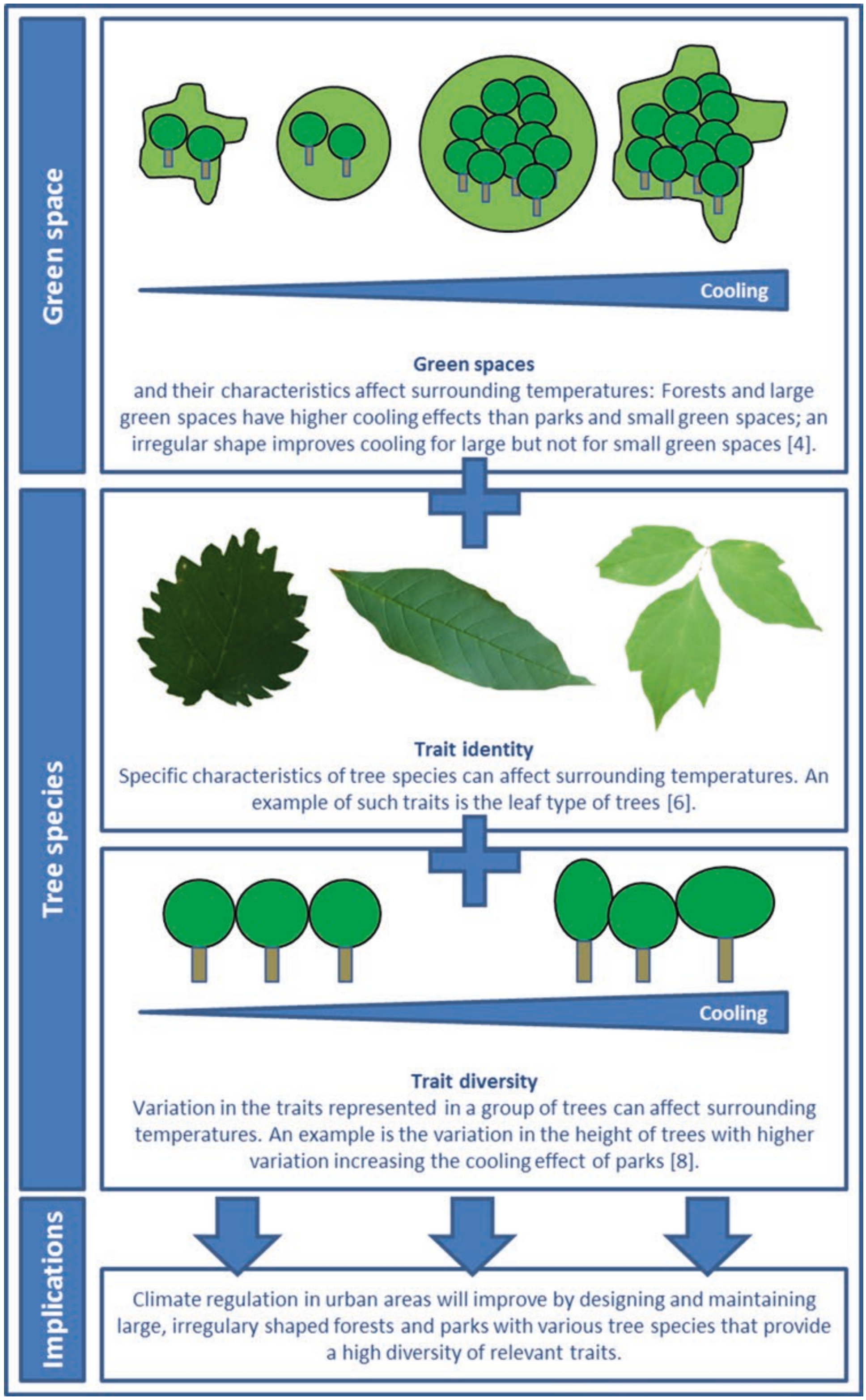

Fig. 26.5 Summary of findings on the effects of green space configuration and tree diversity on climate regulation that should be considered when creating new urban green spaces, based on our own studies in the city of Leipzig [4, 8] and a study in the city of Basel, Switzerland [6]

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27 Climate Change as Driver for Ecosystem Services Risk and Opportunities

Andreas Marx, Markus Erhard, Stephan Thober, Rohini Kumar, David Schäfer, Luis Samaniego, and Matthias Zink

27.1 Introduction

Climate change is already affecting terrestrial ecosystems and biodiversity in Europe [1]. It is one of the five major pressures on ecosystems, as defined by the Millennium Ecosystem Assessment [2], and a "driver" in the concept of ecosystem service risk (Schröter et al., Chap. 1). The relative importance of climate change is projected to increase in the future [3]. Climate change does not only affect mean values, but also extreme events such as heavy precipitation, droughts, and heat waves [4]. Observed impacts (first order risks) include changes in phenological stages, changes in species composition in communities, predator-prey relationships, the migration of species, and changes in soil conditions. Agricultural droughts have already negatively affected ecosystems, e.g., in the ability to store CO_2 [5]. This may reduce ecosystem services (second order risks) that provide means to adapt to negative impacts of climate change, e.g., green infrastructure in cities, which helps locally to reduce heat stress.

Which ecosystem services are addressed? Provisioning services include food, livestock, fibre, bioenergy, fish, timber, and non-wood forest production; regulating services are climate regulation/carbon sequestration, pest control, and natural hazard and water quality regulation; cultural services include recreation, tourism, and aesthetic/heritage aspects.

What is the research question addressed? How does climate change affect ecosystem services, and which role does soil drought play in this game?

Which method has been applied? Literature review and consistent modelling study.

What is the main result? There is a north-south gradient in Europe with increasing opportunities and decreasing negative effects of climate change on ecosystem services in the northward direction. In Germany, ecosystem services that strongly depend on soil moisture dynamics may benefit due a reduction in drought months in the northeastern part of Germany until the end of the century. By contrast, ecosystem services risk is expected to increase in the southwestern part.

What is concluded, recommended? The climate impacts presented and the resulting changes in ecosystem services show the need for adequate management and climate adaptation strategies.

27.2 Impacts of Climate Change on Ecosystem Services in Europe

The scientific field of climate impacts and ecosystem services is rapidly developing, with increasing numbers of publications in the field over the past years. A useful and comprehensive overview of published literature for the years 2004–2013 was included in the latest IPCC report [6]. The effects of climate change on ecosystem services were investigated for European macro-regions (see Fig. 27.1). These were aggregated based on an environmental stratification after Metzger et al. [7]. Ecosystem services were classified into provisioning, regulating, and cultural services. Provisioning services include food, livestock, fiber, bioenergy, fish, timber, and non-wood forest production; regulating services are climate regulation/carbon sequestration, pest control, and natural hazard and water quality regulation; cultural services include recreation, tourism, and aesthetic/heritage aspects (more information on ecosystem services classification is available at CICES [Common International Classification of Ecosystem Services], www.cices.eu). The aggregated results are displayed in Fig. 27.2. The bar charts show the number of studies per ecosystem service category published in the years 2004–2013, categorized in positive, neutral, or negative climate change impacts and aggregated per IPCC region.

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Fig. 27.1 European macro-regions after Metzger et al. [7]. (*Data from* M. Metzger, personal communication.)

From a regional point of view, there is a north-south gradient with decreasing negative effects of climate change on ecosystem services in the northward direction. In the northern region, more studies found positive than negative effects on provisioning services. In the southern region, the provision of ecosystem services is projected to decline in all service categories. It is the only region where climate change has a strong negative effect on regulating services. For all other regions, gains and losses in regulating services due to climate change are rather balanced.

For cultural services, there are only a few studies available with mostly negative effects in the northern and southern macro-region. In the other regions, positive and negative impacts of climate change on cultural services reach the same order of magnitude.

Soils provide the foundation for the production of biomass in terrestrial ecosystems. They filtrate, transform, and store nutrients, substances, and water. Climate change influences the rainfall pattern and its magnitude [6]. Consequently, it impacts the soil moisture dynamic. Soil water availability influences, among others, provisioning services in food production, regulating services in the storage of CO_2, and cultural services in recreational areas. Soil drought may affect these services negatively, e.g., due to effects on net primary production or mineralization, respiration of soil organic carbon and forest fire risk. For example, the 2003 drought event in Europe had major implications on the greenhouse gas balance of terrestrial ecosystems. Ciais et al. [5] simulated continental-scale changes in primary productivity and their consequences for the net carbon balance. Due to heat and drought, the year 2003 resulted in a 30% reduction in gross primary productivity as compared to 1998–2002 over Europe, which translates into a net source of carbon dioxide to the atmosphere. The total amount of 4 years of net ecosystem carbon sequestration was emitted in 2003. In particular, the Mediterranean region faces an increasing risk of droughts under climate change. This may turn temperate ecosystems into carbon sources. Consequently, droughts pose an ecosystem services risk.

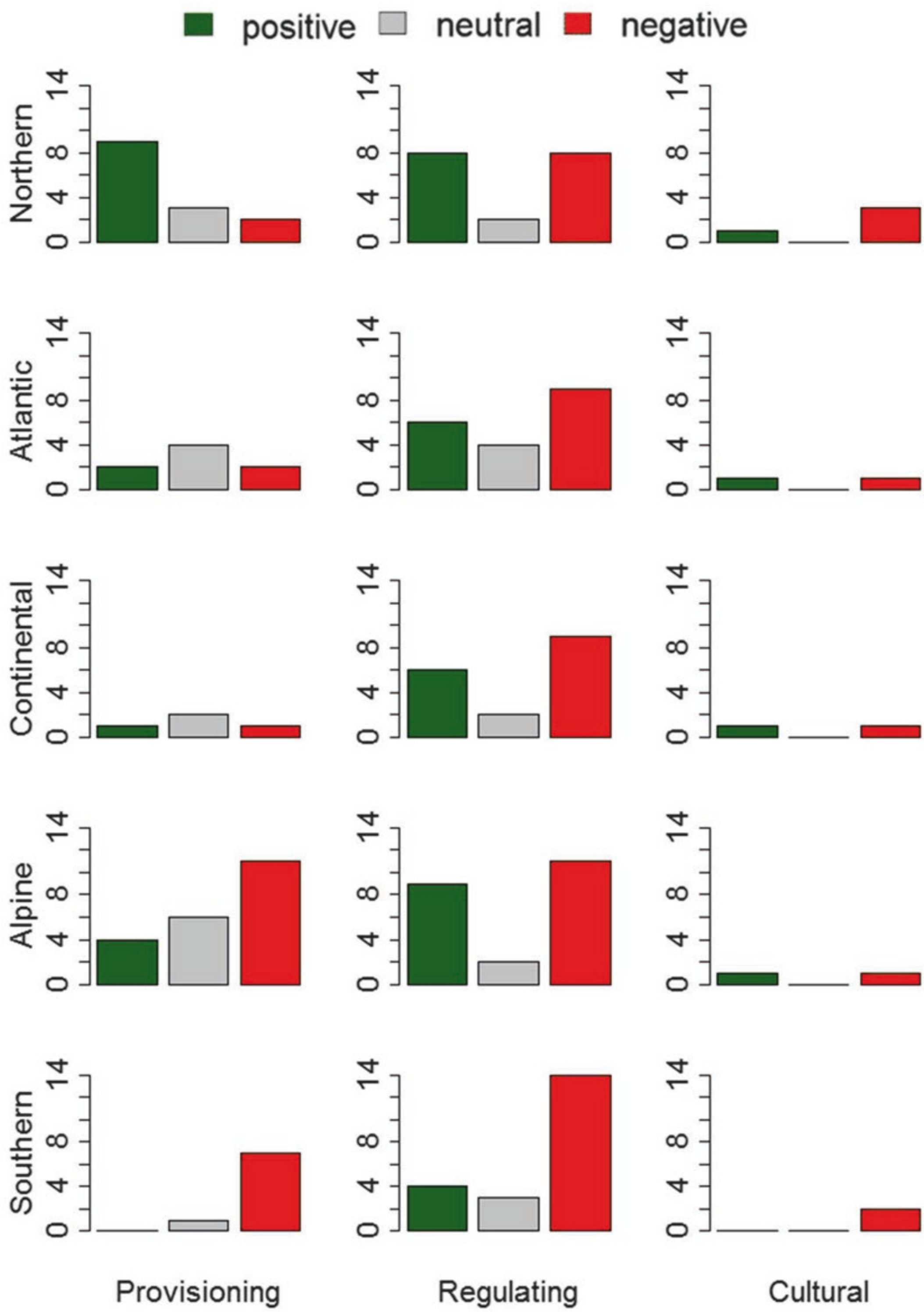

Fig. 27.2 Climate change impacts on ecosystem services for European macro-regions based on the IPCC AR5 literature review [6]

27.3 Climate Change Impact on Soil Moisture Droughts in Germany

Drought is a driver for first and second order risks. Increasing droughts have a variety of negative impacts on ecosystems and their services. Drought events are a risk for regulatory services (as shown above or, e.g., in increasing dust generation and resulting air quality issues), in provisioning services (e.g., reduced crop productivity), and cultural services (e.g., compromising recreational areas and parks). The

2003 drought event had major implications in Germany, e.g., on provisioning services in forestry and agriculture. The combination of soil drought and heatwave led to economic losses in the order of 1.5 billion Euros in agriculture alone [8].

Various modeling studies showed that mean annual soil moisture trends for Germany are negative (e.g., Marx et al. [1]). Samaniego et al. [9] investigated seasonal drought trends based on the soil moisture index (SMI) simulated by the mesoscale Hydrologic Model (mHM; [10, 11]). Monthly soil moisture index trend estimations indicate that there are large areas of Germany showing significant positive trends (i.e., getting wetter) during winter months and negative trends (i.e., getting drier) in summer months for the period 1951–2013. Percentile-based approaches such as the SMI make it possible to compare soil moisture estimations from different models.

Future soil drought trends can be investigated using regional climate simulations. A subset of simulations (see Table 27.1) from the ENSEMBLES project has been used as meteorological forcing for the hydrological model mHM. These regional climate models have been nested within the global climate model ECHAM5 under the A1B emission scenario, representing a best estimate of 2.8 °C global warming until the end of the century compared to 1980–1999 [12].

Daily soil moisture fields at various depths simulated with mHM are aggregated to estimate monthly soil moisture at a spatial resolution of 4×4 km^2. The mean soil depth in Germany is around 1.8 m. Soil moisture of the total soil column is used to estimate the SMI following the approach proposed by Samaniego et al. [9]. The soil moisture index is a percentile-based index, and the period 1971–2000 is used as a reference here to estimate empirical distribution functions for every cell, calendar month, and climate model. The soil moisture index was calculated for the five climate-hydrology simulations for the years 2021–2050 and 2070–2099.

Table 27.1 List of the regional climate models used in this study as meteorological forcing for the hydrological model mHM[a]

Model ID	Model name (RCM-GCM)	Developing institute
1	HIRHAM5-ECHAM5	Danish Meteorological Institute (DMI)
2	RACMO2-ECHAM5	Royal Netherlands Meteorological Institute (KNMI)
3	RCA-ECHAM5	Sweden's Meteorological and Hydrological Institute (SMHI)
4	RegCM3-ECHAM5	The Abdus Salam International Centre for Theoretical Physics (ICTP)
5	REMO-ECHAM5	Max-Planck-Institute for Meteorology (MPI)

RCM = regional climate model, GCM = general circulation model
[a]All models under the A1B emissions scenario

Figure 27.3 shows the number of drought months (SMI ≤ 0.2) for the different 30-year periods. The upper row shows the average number of months under drought conditions over the five ENSEMBLE simulations. The differences between 2021 and 2050 and the reference period are small. For the far future (2070–2099), the numbers of drought months almost double in the southwestern and western parts of Germany, while no significant changes can be observed in the northeast. Over the entire area of Germany, the increase of drought months is almost 30%. The middle and lower rows of Fig. 27.3 show the single-model realizations that lead to the most extreme changes in drought months. In the middle row, the model chain ECHAM5-HIRHAM5-mHM leads to the lowest number of drought months among all models in the near and far future. The spatial average of drought months over Germany reduces from 62 months in the reference period to 29 months in 2070–2099. The lower row in the figure shows the model chain ECHAM5-REMO-mHM, which resulted in the maximum number of drought months among all models. An average increase from 61 to 108 drought months can be observed for the period 2070–2099. Overall, the number of drought months in southwest Germany increases with low confidence because of the large spread of changes in the multi-model ensemble. Ecosystem services that strongly depend on soil moisture dynamics (e.g., provisioning of food and carbon sequestration) may benefit due to a reduction in drought months in the northeastern part of Germany until the end of the century. In contrast, ecosystem services are under risk due to increasing drought months in the southwestern part of the country. Potential losses may appear in forests, for example, under direct effects (e.g., reduction in growth rate) and indirect effects (e.g., due to increased risk of forest fire). Ecosystem services under risk include the provision of wood, recreational areas, and sequestration of atmospheric CO_2. Furthermore, other ecosystem services and sectors such as agriculture would be affected. Peichl et al. [13] could show that silage maize yields in Germany are very sensitive to soil moisture conditions. Soil drought in August and September reduce silage maize yield more than 10% compared to a long-term mean yield. The climate impacts presented here and the resulting ecosystem services risks show the need for adequate management and climate adaptation strategies.

Risks to ecosystem services posed by drought events presented here are only examples; they do not comprise a complete list. Soil drought may propagate to a hydrological drought, which would affect water availability and water quality and influence additional ecosystem services. The results shown here for soil drought in Germany have recently been expanded for Europe in the project EDgE (End-to-end Demonstrator for improved decision making in the water sector in Europe; edge.climate.copernicus.eu). The results on soil droughts in Europe [14] showed an increasing drought

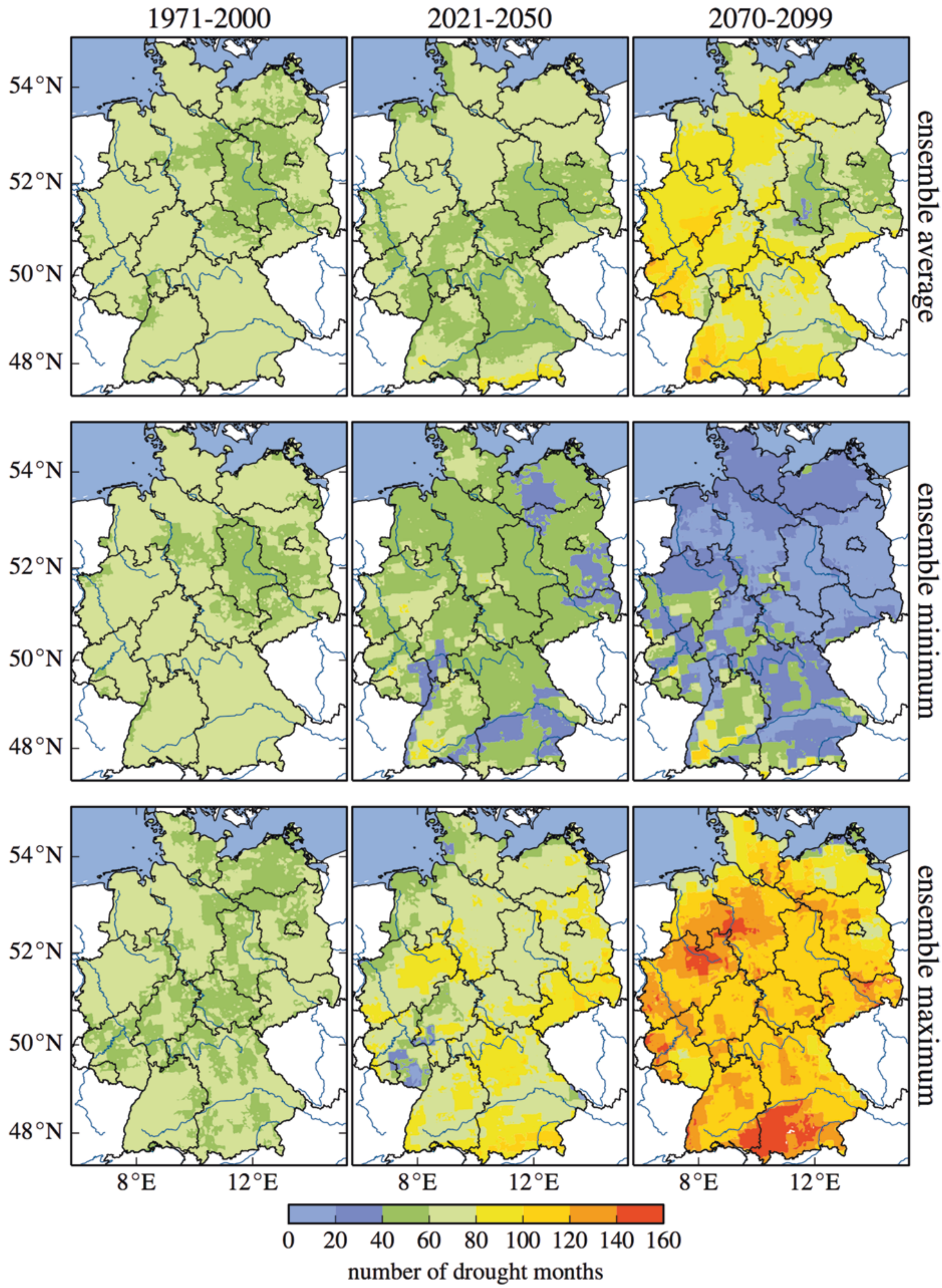

Fig. 27.3 Number of months under drought conditions within the periods 1971–2000 (left column), 2021–2050 (center column), and 2070–2099 (right column). The upper row shows the average number of drought months based on five different climate model inputs. The middle and lower rows depict a single-model realization leading to the minimum and maximum number of drought months, respectively

risk all over Germany under a global warming of 3°C. These data may then provide the basis for estimating European ecosystem services risk in future assessments.

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28 Capacity of Ecosystems to Degrade Anthropogenic Chemicals

Lukas Y. Wick and Antonis Chatzinotas

28.1 Introduction

Pollution of ecosystems by a constantly increasing load of anthropogenic chemicals (i.e., chemicals caused or produced by humans) is a major driver of ecosystem service risk. Knowledge to suggest boundaries for increasing chemical pollution at the planetary scale is vastly insufficient [1]. This knowledge gap encompasses all scales, even down to small-scale ecosystem compartments. While individual chemicals can disturb the functions of ecosystems, the combined actions of multiple anthropogenic chemicals are of particular concern because mixtures of chemicals may cause effects even when individual chemicals are present at concentrations too low to be individually effective. When introduced into the environment, a chemical is influenced by many abiotic and biotic processes that determine its persistence, degradation, transport, and ultimate destination. Abiotic processes may include chemical and photo-degradation, physical binding, unspecific interaction with organisms, volatilization, and waterborne leaching. Biodegradation, by contrast, is driven by the chemical's structural stability towards biochemical reactions, its bioavailability, and the functional effectiveness and stability of the natural microbial communities as principal actors for decontamination towards a "non-toxic environment." According to the specifications of the Swedish Environmental Protection Agency this is an environment where "the occurrence of man-made or extracted substances in the environment must not represent a threat to human health or biological diversity," and where "concentrations of non-naturally occurring substances will be close to zero and their impacts on human health and on ecosystems will be negligible" [2]. The environmental fate of a chemical hence is driven by its individual molecular determinants, and its interactions with given environmental compartments, which themselves are subject to continuous change induced by biogeochemical and ecological processes. Given the structural richness of natural organic compounds that do not accumulate in the environment (and for which an average global turnover time of about 23 years has been calculated [3]) the degradation of most synthetic organic chemicals appears possible. Nevertheless, many of these chemicals build up in the environment, suggesting limited microbial degradation, either because the chemicals and their mixtures were not bioavailable, too toxic, and/or could not develop sufficient "value for life" for the degrader

Which ecosystem services are addressed? Providing an environment where anthropogenic organic chemicals will have no or only a negligible negative impact on ecosystems and on human well-being.

What is the research question addressed? What makes a chemical compatible and available for biodegradation?

What makes an ecosystem capable of biodegradation?

What are the challenges faced when predicting the ability of an ecosystem to biodegrade an anthropogenic chemical?

Which method has been applied? Review of literature of environmental chemistry, environmental microbiology, and microbial ecology.

What is the main result? Opinion statement on key drivers challenging the capacity of ecosystems to degrade anthropogenic chemicals.

What is concluded, recommended? Management of the capacity of ecosystems to degrade anthropogenic chemicals requires (1) adequate availability of the chemical to be degraded; (2) provision of sufficient microbial activity and biodiversity essential for the degradation of chemicals in an ecosystem; (3) development of tools for the prediction of the fate of chemicals and their microbial biodegradation based on sets of easily accessible data; (4) understanding of which ecological networks and core functional groups are most vulnerable to distinct environmental perturbations; and (5) application of the key principles of sustainable chemistry.

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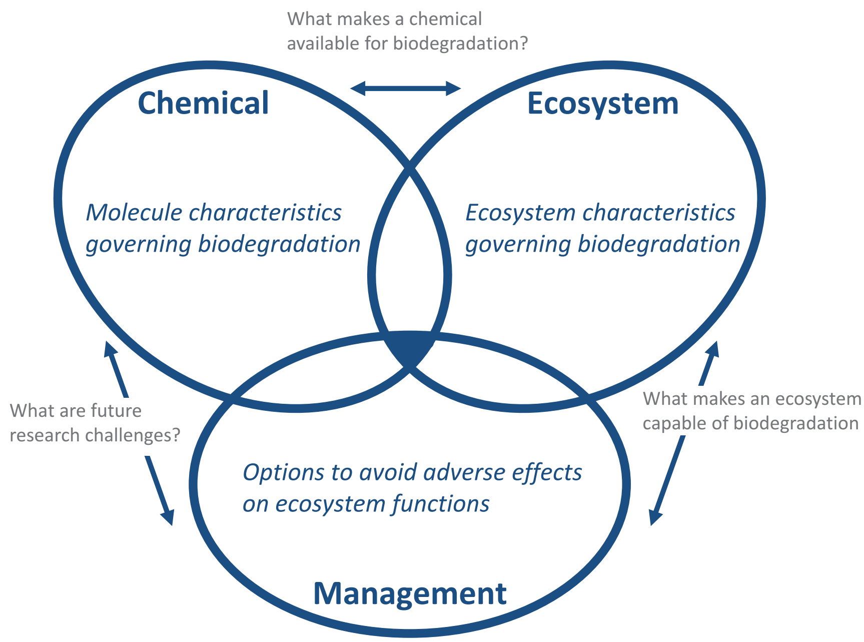

Fig. 28.1 Factors and questions relevant for better understanding and managing the capacity of ecosystems to degrade anthropogenic chemicals

communities. Here we describe factors that enable and limit an (microbial) ecosystem to biodegrade anthropogenic *organic* chemicals (Fig. 28.1). We ask the following questions: "What makes a chemical available for biodegradation?" and "What makes an ecosystem capable of biodegradation?". We further summarize future research challenges for predicting the ability of an ecosystem to biodegrade an anthropogenic chemical.

28.2 What Makes a Chemical Available for Biodegradation?

Generally, microorganisms can benefit from those chemicals that have "value" for them, i.e., that provide elements or building blocks for biomass synthesis or their energy production for survival and growth. Biogenic organic compounds mostly fulfill these purposes, since they reflect 4 billion years of coevolution of producer, consumer, and decomposer organisms. By contrast, novel chemical structures of synthetic chemicals may more easily result in biochemical recalcitrance. Generally, poor benefit can be derived from chemicals that are present at low environmental concentrations (e.g., persistent and mobile organic compounds [4]), from polymers (e.g., plastics), from highly oxidized chemicals and/or hydrophobic monomers (e.g., the chemicals banned by the United Nations Stockholm Convention). Still, complete persistence of anthropogenic chemicals is rarely reported. It appears, rather, that most of the pollutants can be attacked by the wealth of the existing enzymes and/or ongoing evolution of new microbial catabolic pathways [5]. Whether a chemical is degraded may also be contingent on, e.g., the prevailing redox conditions, nutrient status, or the time an environment had to adapt to the chemical. This raises the question of which constraints control the bioavailability and biodegradation even of a *per se* degradable chemical.

A prerequisite for biodegradation is optimal accessibility and availability of chemicals to microorganisms [6]. The term *bioavailability* refers to the degree of interaction of chemicals with living organisms. The bioavailability for degradation is a dynamic feature that can suitably be expressed by the rate of mass transfer of a chemical to a microbial cell relative to its intrinsic catabolic activity [7]. Limited bioavailability (and subsequent biodegradation) may have three major causes: 1) unfavorable environmental conditions (e.g., redox and water potentials, pH, temperature, or the absence of electron acceptors and macro-elements) do not allow for high intrinsic catabolic or co-metabolic activity; 2) the amount of a chemical in the environmental compartment of interest is low (e.g., in the case of micropollutants); or 3) the accessibility or the flux of a chemical to degrading cells is low (e.g., for poorly water-soluble substrates and sorbed chemicals) or even nearly zero (e.g., for polymers). In addition, competing chemodynamic processes (e.g., abiotic degradation and partitioning or reactive transport) may affect the available concentration and biodegradation of chemicals and their degradation products [8]. Bioavailability and biodegradation are hence dynamically interlinked and systems biology approaches are needed to understand pollutant turnover in an ecosystem that typically is subject to changing environmental conditions (Fig. 28.2) and/or is inhabited by higher organisms such as plants [9].

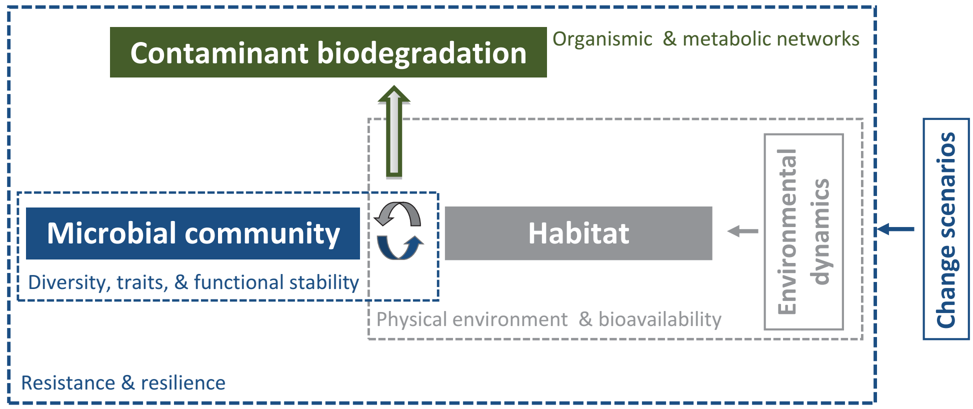

Fig. 28.2 Drivers and processes of the functional stability of an ecosystem towards the biodegradation of an anthropogenic chemical

28.3 What Makes an Ecosystem Capable of Biodegradation?

Next to biophysical aspects as outlined above, microbial activity, interactions [10], and diversity are key drivers for the capacity of ecosystems to degrade chemicals. An increased microbial diversity is thereby believed to provide a larger variety of biodegradation pathways [11]. Various mechanisms produce non-competitive diversity patterns in soil. For instance, spatial isolation within a habitat may provide separate resource niches, thus increasing the number of genotypes on a small scale [12]. Other site-specific characteristics, such as carbon content and resource heterogeneity, are not only major drivers of microbial community structure, but may also counteract a possible pollution-induced decrease of microbial diversity and thus microbial ecosystem functions [13]. Species richness provides a pool of species with potentially relevant traits; these species may turn out to be essential performers or core partners in new interspecific interactions after environmental change [14]. Microbes degrading anthropogenic chemicals often act simultaneously as co-existing and interacting microbial communities within a microbial food web, rather than as single species. Moreover, many of the different metabolic pathways are interconnected. Global environmental change is likely to impair microbial interaction networks that are relevant for distinct functions [15, 16]. In this context, systems with a higher microbial diversity may be more resilient and compensate the loss of species without loss of function, and thus increase the stability and predictability of microbial functions. Microbial functions in already stressed—and hence less diverse—systems can therefore be expected to be less efficient and less predictable. Despite functional redundancy among soil microorganisms, specialized, but unique and essential functions, which are known to be limited to only a few groups, will be disproportionally affected in stressed ecosystems [17].

The bacterial metagenome (i.e., the genomes of the microorganisms representing the genetic potential of the population) can serve as a source to recruit genes or to provide the genetic building blocks for new catabolic pathways in order to either access chemicals as nutrient sources or to detoxify them. These genes can be spread in bacterial communities by horizontal gene transfer (HGT; i.e., the transmission of genetic material between different species by mechanisms other than from parent to offspring). Gene duplication (followed by subsequent mutations) is one of the driving forces in the evolution of novel genes. Duplication is favored for genes coding for enzymes with dual function or promiscuous activity, thus potentially leading to drastically increased capabilities [5]. However, selection is only successful if the new mutation increases the fitness of an individual bacterium. Physicochemical constrains like structural features, accessibility, or availability of pollutants may reduce the evolutionary potential. In fact, many emerging pollutants are often present only at very low concentrations, and thus exert only low selective pressure. Consequently, if bacterial fitness is not affected, new enzymes for pollutant degradation may not evolve because detoxification would not be and the potential substrate would not serve as a valuable nutrient source. Also, stressed systems may provide less energy for the development of resistance to further stress or for the acquisition of additional abilities related the degradation of potentially new pollutants [18].

28.4 What are Future Research Challenges?

The lifecycle (production, use, recycling, elimination, disposal, etc.) of any anthropogenic chemical should take place with the least possible adverse and rebound effects on the well-being of humans and affected ecosystems (“non-toxic environment” [2]). Such is the goal of the design of chemicals and processes that optimize the “lifecycle” of chemicals (“Green Chemistry”). It also includes appropriate tools for quantitative prediction of the fate of chemicals in an ecosystem based on sets of easily accessible data.

Beyond such a chemical’s perspective, sustainable use of chemicals also must address social, economic, and—relevant for this chapter—ecosystem perspectives of chemicals ("Sustainable Chemistry" [19]). It must consider the functional stability of ecosystems to maintain their capacity to adapt to environmental changes. Likewise, knowledge of management options to maintain and foster the biodiversity essential for the degradation of chemicals at all ecosystem scales is required. First attempts of environmental biotechnologists to develop ecology-based conceptual frameworks to improve and predict the performance of microbial-engineered ecosystems (e.g., in waste gas or waste water treatment) may also serve as a blueprint for the management of microbial resources in natural systems [20, 21]. Sustainable use of chemicals will further need to understand at which spatial and temporal scales ecological networks and core functional groups are most vulnerable to anthropogenic chemicals (and mixtures thereof); this concerns in particular ecosystems exposed to varying levels of additional disturbances. Future research should thus also describe the links between the spatial heterogeneity of microbial functional groups, environmental parameters (e.g., water content, redox potential), and pollutants. Filling such knowledge gaps is a prerequisite for quantitative prediction, monitoring and management of the environmental fate of chemicals and, hence, for urgently needed definitions of boundaries for chemical pollution at larger scales. It will simultaneously contribute to better fulfillment of the United Nations Sustainable Development Goals, such as "Responsible Consumption and Production" (SDG 12), "Clean Water and Sanitation" (SDG 6), and "Life on Land" (SDG15).

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29 Impacts of Nitrogen Deposition on Forest Ecosystem Services and Biodiversity

Wim de Vries and Lena Schulte-Uebbing

29.1 Introduction

Nitrogen deposition affects the capacity of forest ecosystems to provide services such as wood production (provisioning service), carbon storage (regulating service), and water quality regulation (regulating service). Nitrogen deposition also affects forest biodiversity. As most forest ecosystems are nitrogen limited [1], increased nitrogen deposition usually decreases biodiversity [2], while it increases net primary production and thus carbon sequestration [3, 4]. Carbon sequestration is an important forest ecosystem service as it slows the growth of atmospheric CO_2 concentrations and thus mitigates climate change. Currently, forests account for more than 90% of the terrestrial carbon sink [5]. In the first part of this chapter, we provide examples of several forest ecosystem services that are affected by nitrogen deposition, and show how the capacity of forests to provide these services relates to the level of nitrogen deposition. In the second part, we present estimates of the contribution of nitrogen deposition to the global forest carbon sink.

Which ecosystem services are addressed? Focus on timber production and climate regulation, links with water quality regulation, pest and disease regulation, wild plants and animals.

What is the research question addressed? What is the global scale impact of nitrogen deposition on forest carbon sequestration, and what are appropriate management approaches to enhance the beneficial and/or reduce the adverse impacts of nitrogen load in forests?

Which method has been applied? Literature review.

What is the major result? Nitrogen deposition generally enhances forest growth and is estimated to be responsible for approximately 10–20% of the global terrestrial carbon sink. At regional scale the effect can be lower or even negative, depending on the level of nitrogen deposition. Furthermore, nitrogen deposition has an adverse effect on the provision of several other forest ecosystem services, such as water quality regulation, pest and disease regulation, and biodiversity.

What is concluded, recommended? Measures to enhance benefits or reduce impacts of nitrogen overloads are achieved by nitrogen removal by grazing or litter removal or addition of phosphorus, calcium, magnesium, and potassium by slow-release fertilizers to alleviate the adverse impacts of nutrient imbalances and acidification.

29.2 Impacts of Nitrogen Deposition on Forest Ecosystem Services and Biodiversity

29.2.1 Nitrogen Deposition as a Risk and Opportunity to Forest Ecosystem Services and Biodiversity

Global environmental changes such as deforestation, climate change, and human perturbation of the nitrogen cycle put the provision of forest ecosystem services at risk [6]. Research on nitrogen deposition effects on forests has been going on for decades, but connections to ecosystem services research are just emerging recently. Atmospheric nitrogen deposition affects the quality of forest soils and thereby forests' capacity to provide services such as wood production, carbon sequestration, and water quality regulation (see Table 29.1 for details).

Figure 29.1 illustrates the conceptual relationships between cumulative nitrogen input on the one hand, and forest growth, nitrogen leaching, and plant species diversity on the other hand. In this example, forest growth and nitrogen leaching can be seen as proxies for a forest's capacity to provide the ecosystem services wood production/carbon

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Table 29.1 Examples for effects of nitrogen deposition on forest ecosystem services (After De Vries et al., [27])

Ecosystem services	Examples of nitrogen effects	Causal link with nitrogen deposition	Critical N deposition level
Provisioning services			
Wild plants and animal products	Decline in biodiversity	N-induced eutrophication and soil acidification affect plant and faunal species diversity and thereby the provision of biodiversity-based products.	5–10 kg N ha^{-1} $year^{-1}$
Timber/wood fuel	Increase in wood production	In N-limited forests, N increases forest growth and wood production; however, in N-saturated forests, N can induce mortality.	20–35 kg N ha^{-1} $year^{-1}$
Regulating services			
Climate regulation	Increased carbon (CO_2) sequestration (cooling effect)	In N-limited systems, N deposition increases forest growth and related carbon sequestration, though it can enhance mortality in some species. N deposition can also increase litterfall and reduce decomposition, leading to soil carbon sequestration.	20–35 kg N ha^{-1} $year^{-1}$
	N_2O and NO_x emissions (warming effect)	N deposition enhances N_2O and NO_x emissions. NO_x emissions in turn induce O_3 formation. N_2O and O_3 are greenhouse gases, and O_3 is toxic for plants and reduces forest growth and thus carbon sequestration.	5–10 kg N ha^{-1} $year^{-1}$
Water quality regulation (water purification)	Decline in groundwater and surface water (drinking water) quality	N eutrophication and N-induced soil acidification cause a decrease in soil C:N ratio and base cations/pH, leading to: • Increasing NO_3, Cd and Al concentrations in groundwater and surface water, which may exceed drinking water quality criteria in view of human health effects.	10–15 kg N ha^{-1} $year^{-1}$
		• Increasing Al concentrations in acid-sensitive surface waters resulting in the reduction or loss of fish (salmonid) populations and reduction of aquatic diversity at several trophic levels (acidification).	
		• Increasing NO_3 concentrations in surface waters causing fish dieback by algal blooms and anoxic zones (eutrophication). Eutrophication is also affected by silica and phosphorus in estuaries.	
Pest/disease regulation	Increase in forest pests	Elevated N input weakens the resilience of forests and increases infestation rates, such as beech bark disease, in response to e.g. increased foliar N concentrations.	15–20 kg N ha^{-1} $year^{-1}$

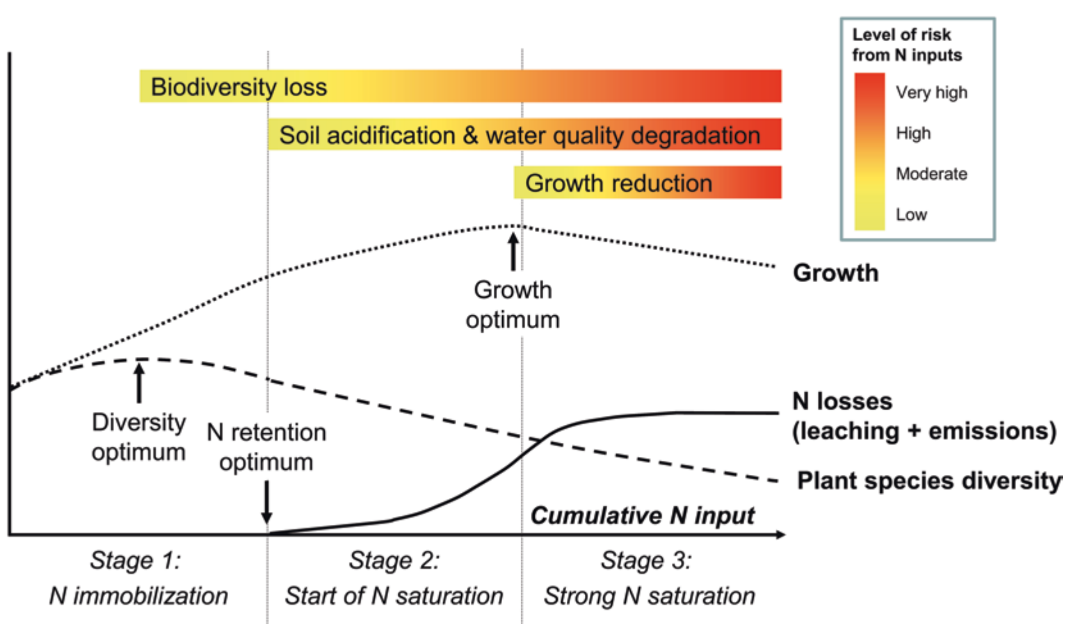

Fig. 29.1 Hypothetical relationship between the stage of nitrogen saturation and the effects on terrestrial ecosystems in terms of soil processes, vegetation changes, and growth. "Cumulative N input" on the *x*-axis refers to cumulative input at a constant annual rate

sequestration and water quality regulation, respectively. Plant species diversity is used as a proxy for biodiversity.

Nitrogen deposition can be both a risk and an opportunity for ecosystem services and biodiversity, depending on the level of nitrogen input. The relationship between nitrogen availability and plant species diversity is best described by a skewed unimodal curve [7]. At very low nitrogen deposition levels, plant species diversity increases up to an optimum, after which plant species diversity starts to decrease (Fig. 29.1, dashed line). At nitrogen deposition levels where negative effects on plant species diversity start to occur, nitrogen leaching is generally still negligible, as most of the nitrogen entering the forest is retained in biomass or immobilized in the soil [8]. At higher nitrogen deposition levels, however, nitrogen leaching starts to increase as the forest approaches “nitrogen saturation” [8] (Fig. 29.1, solid line). In this stage, the soil acidifies and the forest is no longer able to buffer all external nitrogen inputs, which means that nitrogen enters groundwater and surface water, with possible negative effects on aquatic biodiversity and freshwater quality. Forest growth, on the other hand, generally still increases at nitrogen input levels where adverse impacts on plant species diversity and soil and water quality already occur (Fig. 29.1, dotted line). At even higher levels of nitrogen deposition, forests start to approach full nitrogen saturation. In this stage, nitrogen immobilization is negligible and soil acidification becomes more extreme. Forest growth is reduced due to both leaching of base cations (resulting in nutrient imbalances in roots and leaves) and toxicity effects from elevated aluminium concentrations [8].

29.2.2 Nitrogen Deposition Thresholds for Minimizing Risks to Ecosystem Services

Figure 29.1 illustrates the trade-off between the positive impact of nitrogen enrichment on forest growth and related carbon sequestration on the one hand, and the negative impact of nitrogen enrichment on other ecosystem services (e.g., water quality regulation by nitrogen retention) and on biodiversity on the other hand. In this section, we discuss at which levels of (cumulative) nitrogen deposition adverse impacts on plant species diversity, nitrogen retention, and forest growth start to occur.

The optimal nitrogen load for plant species diversity is generally very low. Even at low levels of nitrogen deposition, forest floor plant species composition shifts towards more nitrophilic species [2]. Most relationships between nitrogen deposition and plant species richness indicate a continuous decline above 5–10 kg N ha^{-1} $year^{-1}$ [9, 10]. Based on an extensive literature review, Bobbink et al. [2] propose thresholds for nitrogen impacts on biodiversity between 5 and 10 kg N ha^{-1} $year^{-1}$ for boreal forests and 10–20 kg N ha^{-1} $year^{-1}$ for temperate forests. For tropical forests, an effect threshold has not been given since productivity and related biodiversity impacts are often limited by phosphorus and not by nitrogen [2].

The nitrogen deposition level at which nitrogen is no longer completely retained in biomass and soil and at which leaching starts to increase depends on, among other factors, the C:N ratio in the organic layer [11]. Data from hundreds of forest plots indicate that nitrogen leaching is negligible above a C:N ratio of 40 and below a threshold of 10 kg N ha^{-1} $year^{-1}$ [11]. If those thresholds are exceeded, nitrogen availability increases, enhancing both nitrogen leaching (which negatively affects water quality regulation) and emissions of nitrous oxide (N_2O) and nitrogen oxide (NO) (which negatively affects climate regulation, as N_2O is a greenhouse gas and NO stimulates formation of the greenhouse gas O_3).

With respect to reduced forest growth and carbon sequestration, long-term nitrogen addition studies and data from forest monitoring studies indicate threshold values of 20–35 kg N ha^{-1} $year^{-1}$ [12]. Long-term nitrogen inputs above this threshold have been shown to increase deficiencies of other nutrients, such as phosphorus [13]. Next to increasing biomass production, nitrogen deposition may also increase soil carbon sequestration due to increased carbon inputs from litterfall [14] and reduced decomposition of soil organic matter [15]. While low levels of nitrogen inputs generally stimulate heterotrophic respiration, nitrogen inputs above 25 kg N ha^{-1} $year^{-1}$ generally reduce heterotrophic respiration and thus enhance soil carbon sequestration [12, 15]. The magnitude of this nitrogen-induced enhancement of the soil carbon pool is, however, most likely less than the reduction in tree carbon sequestration [12].

Management approaches to reduce the impacts of nitrogen overloads on biodiversity generally focus on removal of nitrogen, either by grazing or litter removal. The adverse impacts of nutrient imbalances and acidification and thereby on forest growth can be reduced by the use of slow release fertilizers that release phosphorus, calcium, magnesium, and potassium during a long period lasting years to decades (see Fig. 29.2).

29.2.3 Regional Variation in Nitrogen Deposition and Implications for Ecosystem Services

Nitrogen deposition rates on the world’s forests for the year 2000 are shown in Fig. 29.3. The critical threshold value for plant species diversity (5–10 kg N ha^{-1} $year^{-1}$) was not exceeded in many boreal forests and in large parts of the

Fig. 29.2 Forest fertilization to amend impacts of soil acidification. (*Image courtesy of* Wim de Vries.)

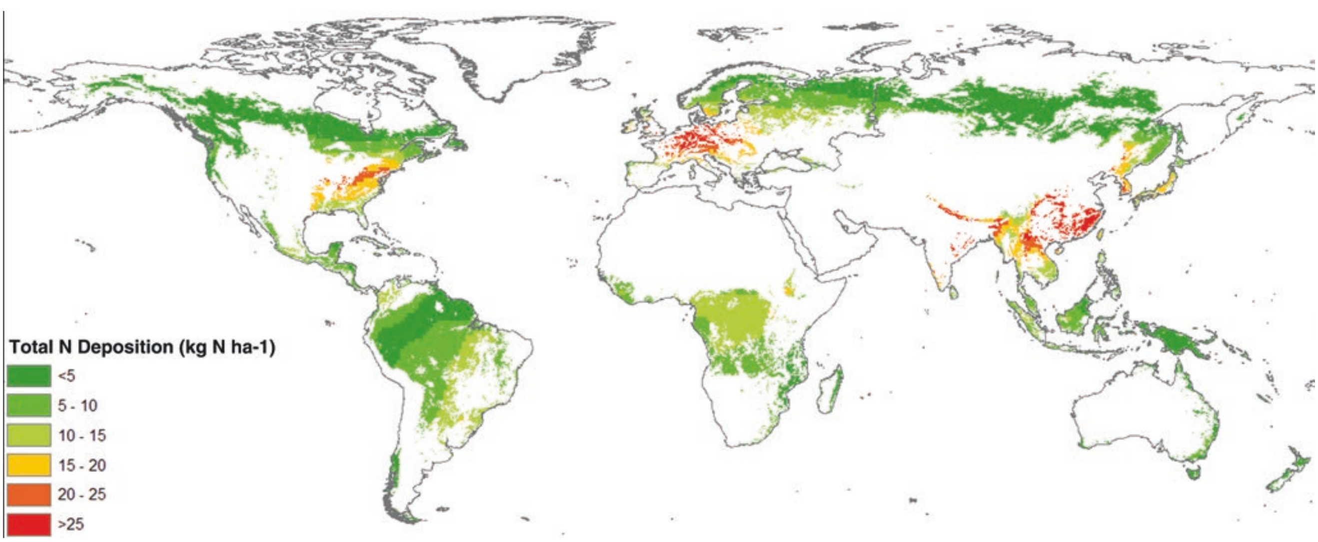

Fig. 29.3 Map showing global nitrogen (N) deposition on forests for the year 2000. Total N deposition ($NH_3 + NO_x$) was based on results of the TM5 model [28]. N deposition is only shown for grid cells where forest cover is higher than 40% according to global data on forest cover for the year 2000 by Hansen et al. [29]

Amazon forests. The critical threshold of 25 kg N ha^{-1} year^{-1}, where the effects of nitrogen deposition on forest ecosystem services are overwhelmingly negative, is exceeded in forested regions with intensive human agricultural and/or industrial activities, such as Central Europe, the Eastern US and China. The threshold for N immobilization near 10 kg N ha^{-1} year^{-1} is also exceeded in parts of Eastern Europe, Latin America and Central Africa (Fig. 29.3).

29.3 Carbon Response to Nitrogen Deposition on Forest Ecosystems

The effect of nitrogen deposition on the capacity of a forest to sequester carbon can be quantitatively expressed by the C–N response ratio, which we define as the additional mass unit of sequestered carbon per additional mass unit of nitrogen deposition. The C–N response varies per forest ecosystem compartment (soil, woody biomass, leaves, roots), due to differing nitrogen retention fractions, C:N ratios, and carbon residence times. Most carbon is stored in trees' woody biomass and in soils, as these compartments have both large carbon storage potentials and slow turnover times. Below we present three approaches used to estimate C–N responses.

29.3.1 Stoichiometric Scaling

The stoichiometric scaling approach is based on the observation that C:N ratios in forest biomass and soils are relatively constant. The effect of nitrogen deposition on carbon sequestration can thus be calculated by multiplying: 1) the fraction of external nitrogen inputs that is retained in the forest ecosystem with 2) the fraction of retained nitrogen allocated to different forest ecosystem compartments (woody biomass, non-woody biomass, and soil), and 3) the C:N ratio of each compartment [12]. Using this approach, de Vries et al. [12] estimated the forest C–N response for biomass and soil combined at 40 kg C per kg N in boreal forest, 31 kg C per kg N in temperate forest and, 11 kg C per kg N in tropical forest. The soil carbon pool accounted for 35–45% in these estimates, the biomass carbon pool for 55–65%. The decline in C–N response from boreal to tropical forests is mainly due to a decline in nitrogen retention and C:N ratios in woody biomass, correlated with an increase in nitrogen availability [12].

29.3.2 Experimental Nitrogen Addition Studies

Experimental nitrogen addition studies can directly estimate C–N responses by estimating plant and soil carbon sequestration in control plots and in experimental plots with different nitrogen treatments, and dividing the treatment effect in terms of carbon sequestration by the amount of added nitrogen. However, results from these experiments are only valid for the specific location where they have been performed, which limits their use in regional and global assessments [16]. Long-term nitrogen addition experiments in boreal and temperate forests indicate biomass C–N responses of 15–25 kg C per kg N [17] and soil C–N responses of 10 kg C per kg N for boreal forests [18] and 36 kg C per kg N for temperate forests [15].

29.3.3 Field-Based Monitoring Studies Along Nitrogen Deposition Gradients

Gradient studies estimate C–N responses by correlating growth observations at forest monitoring plots or eddy correlation measurements of net carbon exchange (see Fig. 29.4) with environmental variables, including nitrogen deposition. However, as nitrogen deposition may co-vary with other environmental factors affecting forest growth, such as increasing temperature due to climate change, this approach requires a careful accounting for these influences. Sutton et al. [16] analysed the effects of nitrogen deposition on net ecosystem productivity in 22 European forests and estimated a C–N response of 50–75 kg C per kg N (biomass + soil). Growth observations at more than 350 long-term monitoring plots in Europe indicated an aboveground biomass C–N response of 19–26 kg C per kg N [19, 20], which is lower than the response of 51–82 kg C per kg N found by Thomas et al. [4] for US forests. Fleischer et al. [21] found a comparable biomass C–N response (20–30 kg C per kg N) in an evaluation of a global data set relating eddy covariance measurements of net carbon exchange from 80 forest sites with modelled nitrogen deposition, environmental variables, and stand characteristics.

29.4 Global-Scale Estimates of the Contribution of Nitrogen Deposition to Forest Carbon Sequestration

The effect of nitrogen deposition on global forest carbon sequestration can be estimated by multiplying the amount of nitrogen deposited on forests with the C–N response ratios obtained by either stoichiometric scaling [12], nitrogen addition studies [22] or gradient studies [4]. Using C–N response ratios from stoichiometric scaling, de Vries et al. [12] estimate a global nitrogen-induced carbon sink of 0.28–0.45 Pg C $year^{-1}$ in forest biomass and soils.

In addition, dynamic global vegetation models (DGVMs) are used to estimate nitrogen-induced forest carbon storage. These models are based on a mathematical description of carbon-cycle dynamics in dependence of temperature, CO_2 concentration, and nitrogen deposition. Some models even include carbon–nitrogen–phosphorus interactions. The effect of nitrogen deposition on global carbon sequestration can be isolated by comparing carbon-cycle dynamics in model runs with and without nitrogen deposition [23, 24]. Estimates of global nitrogen-induced carbon sequestration based on these models vary mostly between 0.2–0.5 Pg C $year^{-1}$ [23], which is similar to the estimate by de Vries et al. [12].

Compared to a global terrestrial carbon sink of 2.6 ± 0.8 Pg C $year^{-1}$ [25], this indicates that nitrogen deposition is responsible for 10–20% of the global terrestrial carbon sink. However, the notion that nitrogen deposition is

Fig. 29.4 Eddy covariance tower for carbon and water flux measurements in an oak tree savanna, Sardon catchment, Spain. (*Image courtesy of* Lena Schulte-Uebbing.)

desirable because it enhances forests' capacity to sequester carbon is problematic for three reasons. First, as we argue in the first part of this chapter, nitrogen deposition levels that are beneficial to forest growth and thus carbon sequestration are detrimental to biodiversity and water quality. Second, the effect of nitrogen deposition on forest carbon sequestration is likely to diminish over time, as other nutrients (such as phosphorus, magnesium, or potassium) are depleted and become increasingly limiting [13]. Other global change drivers might amplify these deficiencies. A recent meta-analysis [26], for example, found that warming and drought increase plant N:P ratios, which implies that forests in a warmer and drier climate need more phosphorus relative to nitrogen. Third, anthropogenic disturbances of the nitrogen cycle that are responsible for increased levels of nitrogen deposition on forests also affect atmospheric greenhouse gas concentrations in several other ways, most importantly by enhancing emissions of the greenhouse gas nitrous oxide (N_2O), and by enhancing ozone (O_3) formation, which is both a greenhouse gas and reduces forest productivity and thus carbon sequestration. The overall effect of human nitrogen fixation on net greenhouse gas emissions is uncertain, but it is likely that the increase in N_2O emissions and O_3 production due to human nitrogen fixation for food and energy production more than offsets the climate benefit of increased nitrogen-induced carbon sequestration [22].

29.5 Conclusions

Nitrogen deposition has a beneficial or adverse effect on the provision of several forest ecosystem services, depending on both the level of nitrogen deposition and the service considered. Biodiversity and services such as water quality regulation are much more sensitive to increasing nitrogen deposition levels than the services wood production and carbon sequestration. Beyond a threshold of 15–25 kg N ha^{-1} $year^{-1}$, however, effects of nitrogen deposition on forest ecosystem services are nearly always negative. This threshold is currently already exceeded in much of Central Europe, Eastern US, and China. Estimates of the contribution of nitrogen to global forest carbon sequestration indicate that elevated nitrogen deposition is responsible for approximately 10–20% of the global terrestrial carbon sink of 2.6 ± 0.8 Pg C $year^{-1}$; this beneficial climate effect, however, is likely at least partially offset by adverse effects of enhanced N_2O emission and O_3 production due to human nitrogen fixation for food and energy production. In areas with persisting high levels of nitrogen deposition, forests can become saturated with nitrogen, which represents a risk to the permanence of this service in those areas. Management approaches to reduce the impacts of nitrogen overloads include nitrogen removal by grazing or litter removal or addition of phosphorus, calcium, magnesium, and potassium by slow-release fertilizers to alleviate the adverse impacts of nutrient imbalances and acidification.

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Ecosystem Services from Inland Waters and Their Aquatic Ecosystems

30

Karsten Rinke, Philipp Steffen Keller, Xiangzhen Kong, Dietrich Borchardt, and Markus Weitere

30.1 Introduction

Inland surface waters (i.e., lakes, reservoirs, running waters, and wetlands) are highly valued ecosystems [1] and are greatly relevant to global biodiversity [2]. Their services to humankind and their role as a primary resource makes inland waters an attractor for human settlement and activity: around 80% of the human population lives downstream of renewable freshwater resources [3] and is served by this grand benefit. Water resources are degraded by human activities in terms of both quantity and quality. Degradation of inland waters comes along with a degradation of the services provided by their aquatic ecosystems, which suffer from water abstractions, habitat change (including damming), pollution, eutrophication, climate change, and invasive species [4].

This paper provides an overview over key services provided by aquatic ecosystems as well as the global distribution of inland surface waters. Additionally, it reviews the most important risks for these ecosystem services and discusses corresponding challenges for water resources management.

Which ecosystem services are addressed? Provisioning, regulating, cultural , and supporting ecosystem services (ESS) from aquatic ecosystems (overview), nutrient retention in aquatic ecosystems.

What is the research question addressed? We provide an overview of the global distribution of inland waters. We review ecosystem services from aquatic ecosystems and their risks. We explore options to manage nutrient retention in aquatic ecosystems.

Which method has been applied? GIS-based methods and literature review.

What is the main result? Existing management policies (e.g., EU-WFD) for aquatic ecosystems do not include the ESS approach.

What is concluded, recommended? New management concepts are required that integrate ESS into the existing framework.

30.2 Global Distribution and Occurrence of Inland Surface Waters

A thorough assessment of inland surface waters was established by the Global Lakes and Wetlands Database (GLWD) [5], which compiles geographic data from a great variety of sources. Global and continent-specific surface areas of lakes, reservoirs, rivers, and wetlands are provided in Table 30.1, and Fig. 30.1 illustrates their global distribution. According to GLWD, for example, lakes and reservoirs in total cover an area of 2.7×10^6 km^2, which corresponds to about 2.0% of the global land surface area when leaving out Antarctica and glaciated Greenland. An extrapolation [5] by the authors suggested that the total number of lakes may exceed 15×10^6 (for lakes $\geq$1 ha) resulting in a total coverage of even 3.2×10^6 km^2. These surface areas covered by inland waters are not, of course, static values, but continuously change. Surface waters are undergoing dynamic changes in different regions due to anthropogenic drivers each with serious implications. Between 1984 and 2015, for example, almost 90×10^3 km^2 of permanent surface water disappeared on the continents while at the same time about 184×10^3 km^2 were formed [6]. The creation of new surface waters was attributed to reservoir construction, i.e., it occurred in regions where water was already present. Over 70% of the loss of surface waters, however, was taking place in the Middle East and central Asia, where conditions are already rather dry. The expansion of surface waters by the filling of reservoirs is projected to increase further in future, as about 3700 large dams [7] are currently under planning or construction and rising markets for hydropower promote this trend.

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Table 30.1 Surface areas of lakes, reservoirs, rivers and wetlands, separated by continents excluding Antarctica and Greenland

Category	Africa	Asia	Australia	Europe	North America	Oceania	South America	Total
Lakes	227.25	447.91	8.10	550.54	1095.98	5.07	97.32	2432.18
Reservoirs	38.27	47.77	4.19	44.47	68.74	1.00	47.44	251.87
Rivers	45.98	128.56	0.52	16.82	49.31	1.14	119.36	361.71
Wetlands	1075.28	2435.08	147.70	237.76	2794.54	13.19	1485.19	8188.74
Total	1386.77	3059.33	160.51	849.60	4008.58	20.40	1749.31	11,234.50

Note that all wetland categories from Fig. 30.1 were lumped together into one group except intermittent wetlands, which were omitted from the analysis. Surface areas are given in 1000 km^2. (Data from the Global Lakes and Wetlands Database (GLWD) [5]; http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database)

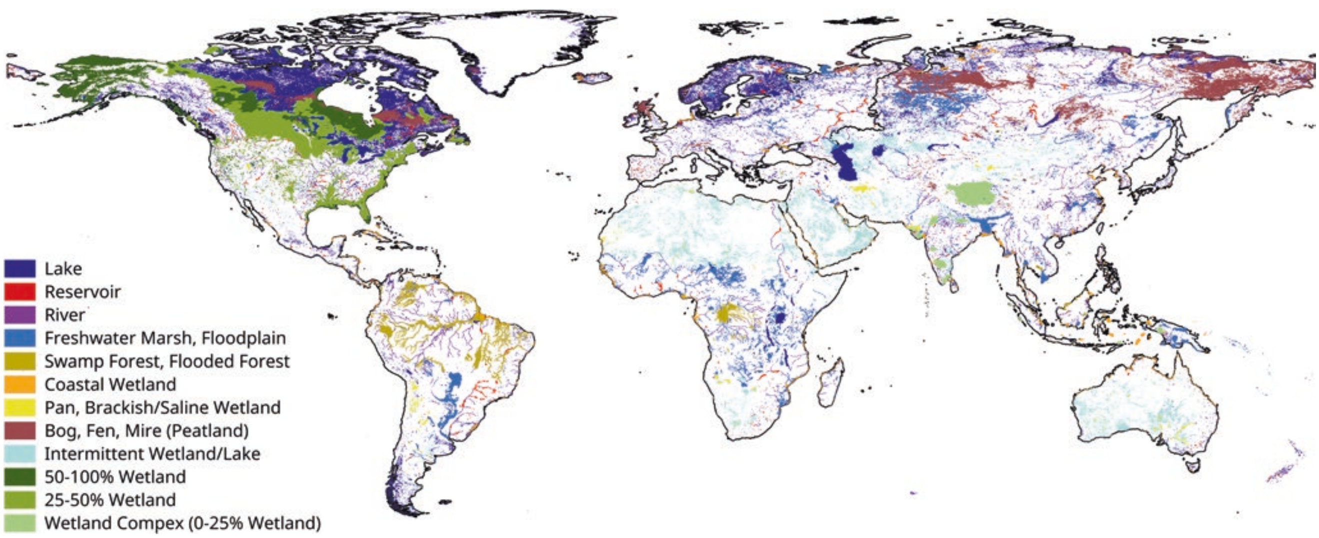

Fig. 30.1 Global distribution of Inland waters of different categories as provided by the Global Lake and Wetland Database [5], online available at http://www.worldwildlife.org/pages/global-lakes-and-wetlands-database. Shorelines from NOAA NCEI [20]

Although the surface areas compiled in Table 30.1 suggest that inland waters cover only a small fraction of the earth's surface—inland waters cover less than 1% of the earth—their importance for the continental biosphere and related ecosystem services is disproportionately high. Global inland waters harbor about 6% of global biodiversity (ca. 100,000 out of 1.8×10^6 species) [2] and even up to 35% of vertebrate diversity [8]. Given the spatial separation between river basins and large, ancient lakes (e.g., East African lakes), freshwater ecosystems further sustain very high numbers of endemic species [2].

30.3 Ecosystem Services of Inland Waters

Ecosystem Services (ESS) are usually defined [4] in a rather broad sense as "the benefits people obtain from ecosystems". By following the classification of ecosystem services proposed in the Millennium Ecosystem Assessment (MEA) reports, we distinguish four major categories of Ecosystem Services [4]. First, *Provisioning Services* include products generated from inland waters (e.g., food, fibers) as well as direct use of water for consumptive and non-consumptive purposes. Second, biological self-purification, the regulation of local hydrology, and climate are typical examples of *Regulating Services*. Third, Non-material values from inland waters are summarized as *Cultural Services* and include important services like recreation and tourism as well as educational, ethical, and aesthetic values. Finally, indirect benefits and aspects of long-term stability are referred to as *Supporting Services* and include soil formation, nutrient cycling, and carbon sequestration. A list of ecosystem services derived from inland waters and freshwater ecosystems is provided in Table 30.2.

30.4 Risks to Ecosystem Services of Inland Waters

Habitat loss and destruction of freshwater biomes: Water abstracted for anthropogenic use and the drainage wetlands induced severe losses of inland water biomes in the past century, to a magnitude of more than 50% loss [4]. Also, flow regulations by reservoirs contribute to this loss, particularly for riparian wetlands. This destruction of inland water biomes goes along with a complete loss of regulatory services as well as biodiversity losses.

Table 30.2 Classification of ecosystem services with examples from aquatic ecosystems including human control and influences

Service	Description and examples	Human controls and influences
Provisioning services		
Food	Fisheries products (e.g. fish, crustaceans), wild game and vegetables	Intensification by aquaculture, manipulation of natural communities by stocking and selective catch
Domestic water use	Drinking water production and other domestic uses	Total withdrawal 464 $km^3\ a^{-1}$, desalinization can substitute freshwater withdrawal, returning as treated/untreated waste water
Industrial water use	Process water in industries and cooling water	Total global withdrawal 768 $km^3\ a^{-1}$, cooling water returns with elevated temperature
Agricultural water use	Irrigation water for production of agricultural goods in arid regions	Total global withdrawal 2769 $km^3\ a^{-1}$
Non-consumptive water use	Hydropower generation and transportation/ navigation	Goes usually along with massive morphological degradation; Hydropower is the major global driver of dam construction
Fiber, fodder, peat	Reed production, animal fodder, peat as energy source	Mostly local/regional importance, e.g. by indigenous communities
Regulating services		
Self-purification	Maintenance of water quality, detoxification, natural filtration, nutrient retention. Great importance of benthic communities (biofilms, particle feeders)	Reduction of contact zones between water and benthic zone by channelization and engineered flood plains. Intact communities are required and high pollution reduces self-purification capacity
Flood buffering	Retention capacities of riparian zones, wetlands, and lakes, buffering of flash floods	Large inundation areas and associated storage volumes are required. Buffering capacities harmed by diking of rivers, melioration of wetlands and water level regulation of lakes
Land-water-interactions	Groundwater recharge from inland waters, transition zones between terrestrial and aquatic ecosystems	Intact flood plains and riparian zones are required. Loss of riparian zones and melioration of wetlands reduce interactions
Climate regulation	Buffering of air temperature and humidity variations by evapotranspiration	Large areas of inland waters are required, loss of wetlands reduce regulating capacity
Cultural services		
Recreation and tourism	Recreational areas are required near urban areas; tourism is of high importance at the regional scale	Recreation/tourism can generate high incomes, High monetary returns from sportfishing, risk of overuse by tourism
Aesthetic/spiritual values	Sacred lakes, ethical heritages, aesthetic landscape elements	Only valued if ethical attitude is developed, often overridden by economic values
Educational values	Education in Ecology, schools, universities and stakeholders (farmers, water managers...)	Education outside academia often limited by economic resources
Supporting services		
Soil formation	Soil formation and fertilization by sedimentation in riparian zones	Soil formation requires connectivity between stream and riparian zones and a quasi-natural flood dynamics
Nutrient retention and cycling	Nutrient storage in rivers and riparian zones, nutrient spiraling in rivers	Intact communities are required, hydromorphological and chemical degradation interferes with nutrient cycling
Biodiversity and food web dynamics	High habitat diversity and species richness mediate resilience (“insurance”)	Habitat loss by morphological degradation and pollution harm diverse communities and biomes

Adapted from Millennium Ecosystem Assessment [4]; withdrawals taken from Aquastat at http://www.fao.org/nr/water/aquastat/main/index.stm

River damming and large-scale water transfers: More than 45,000 large dams [4] lead to fragmentation of river systems, loss of migratory species (largely fish) and a disruption of downstream sediment transport. While the construction of dams enlarges flood buffering capacities and intensifies non-consumptive services, there are immense environmental costs in terms of degrading regulatory services (e.g., for nutrient dynamics in riparian areas or sediment transport) and biodiversity loss.

Pollution and eutrophication: Pollution by point- and diffusive sources is a global problem for aquatic ecosystem services with specific local priorities. Untreated wastewater, particularly (but not exclusively) in developing countries, is a major source of organic pollution that threatens multiple provisioning services such as water use and fishing due to stimulated bacterial activity and the corresponding loss of oxygen. Eutrophication, i.e., the excessive algal production in response to elevated nutrient loads, is also still a worldwide problem causing multiple effects, e.g., on consumptive water use and recreation (high densities auf autotrophs including toxic algal and cyanobacteria) and fish mortality. Eutrophication points to the importance of nutrient retention in anthropogenic and natural systems, underpinning the significance of nutrient retention as a key ESS (see also Box 30.1). Contamination of chemical and emerging pollutants affects provisioning and regulatory services both directly (e.g., poisoning) and indirectly (e.g. fish kills, biodiversity loss, legacy loads). Biological contamination receives increasing awareness as it threatens services such as consumptive water use and recreation. Besides the occurrence of pathogenic viruses, bacteria, and protozoans itself, the evolution and spread of antibiotic-resistant bacteria is a developing water quality problem [9].

Box 30.1 Management of Nitrogen Retention

At the global scale, nitrogen cycling is currently out of natural balance. It is estimated that anthropogenically created reactive nitrogen (Nr) amounted to 187 Tg in 2005 [16] (an increase of 120% from 1970), which amounts to twice the amount of total fixed Nr from natural processes [16]. Inland waters constitute active pipes that transport and process Nr from terrestrial environments into the marine realm. The relative importance of processing versus transport is affected by ecosystem properties and can therefore be actively managed. Real-time monitoring and nitrogen budgets revealed, for example, that river reaches with natural morphology realise far higher nitrogen removal rates than channelized reaches with artificial morphology [17]. Any measure that intensifies the exchange between the stream channel on the one hand and the hyporheic zone or the floodplain on the other hand is supporting nitrogen removal by denitrification [18]. In essence, river restoration projects will not only improve the habitat conditions for certain species, but also improve the nitrogen retention of the landscapes. Standing waters also serve as important sinks for nitrogen. Besides hydro-morphological features like depth and residence time, ecosystem structure also determines retention efficiency. By applying the established biogeochemical lake model PClake to Lake Chaohu [19], it turned out that alternative stable states of the ecosystem influence nitrogen retention. In shallow lakes, alternative stable states occur under a given nutrient load, i.e., the lake can be either dominated by suspended phytoplankton (turbid state) or by bottom-dwelling macrophytes (clear state). We found that the clear state achieves much higher nitrogen removal rates than the turbid state due to positive effects of macrophytes in nitrogen immobilisation. In the case of Lake Chaohu, nitrogen retention efficiency increases from 10% per year in the turbid state to almost 60% per in the clear state. In conclusion, nitrogen retention can be affected by ecosystem properties and management measures can be implemented that intensify nitrogen retention. This would constitute an important contribution to the protection of coastal waters, which suffer from massive nitrogen pollution. We therefore advise to include nutrient retention into existing management plans of inland waters.

Invasive species: The introduction of non-native species into existing ecosystems is of global concern and heavily affects inland waters, which naturally form rather isolated units. Human activities (water transfers, shipping, etc.) are progressively connecting these previously separated units. Effects from invasive species in inland waters are well documented [4] and include not only the elimination of local communities, but also affect whole ecosystems and their interactions. Effects of invasive species on ecosystem services are diverse and can be negative, neutral, or positive. Examples are the Ponto-Caspian zebra and quagga mussels (*Dreissena polymorpha* and *D. bugensis*), which, on the one hand, can efficiently control eutrophication [10], but on the other hand cause large expenditures related to fouling because, e.g., they clog the water intake pipes of thermal power plants [11].

Climate Change: Besides the direct effects of climate change—e.g., warming of waters, reduction of oxygen, and acceleration of biological rates—loss of inland waters and a substantial restructuring of aquatic communities are also anticipated. Provisioning services will suffer from climate change, particularly in semi-arid and arid regions, due to increasing evapotranspiration and incidence of extreme events. Inland waters affect greenhouse gas concentrations by carbon sequestration on the one hand and emissions of CO_2, CH_4, and N_2O on the other hand.

Overfishing and aquaculture: Overharvesting of stocks and fish stocking both severely affect community composition and biodiversity. The rise of aquaculture (increasing about 9% per year since 1970) [4] also appears as a major driver. Aquaculture farms contribute to pollution, habitat degradation, and the spread of non-native species and pathogens. While aquaculture increases food provisioning, the environmental costs for regulating and supporting ESS are high.

30.5 Trade-Offs and Management Approaches

The multiple uses of aquatic ecosystems lead to multiple trade-offs, and we can provide only a superficial insight into this important field of research that certainly deserves more attention in science, engineering, and management. A classical trade-off is the dual use of inland waters for waste water discharge and for municipal water supply. Water contamination by human and animal feces is considered the greatest microbial risk to drinking water and human health [12], and global mortality from water-associated diseases still exceeds five million people per year [13]. Another systematic conflict exists between provisioning and regulating/supporting ESS. While water abstraction from inland waters for anthropogenic use is recognized as an ecosystem service, it is also accepted that aquatic ecosystems require a certain discharge of water to maintain ecosystem functions and the corresponding and regulating/supporting ESS. Although the concept of Environmental Flows provides an outline of good management practice, overexploitation of provisioning ESS leads to severe ecosystem damages and loss of other ESS in many regions of the world.

The practice of management of ESS from inland waters is still in its infancy. Natural protection is usually focused on protecting habitats and endangered species, but not on protecting ESS. For aquatic environments, the European Water Framework Directive (WFD) implements a more comprehensive approach aimed at achieving a "good ecological status." Although successfully bridging protection, pollution control, and management, the WFD fails to explicitly include ESS. Current monitoring procedures of the WFD focused on biological structure, not on function or ecosystem services [14], and the operationalisation of the ecosystem service approach is perceived as a "wicked" problem and should be considered for inclusion in the WFD [15]. Approaches for managing ESS are exemplified in Box 30.1 for the key regulating service of nitrogen retention.

Selected aspects of aquatic ecosystem goods and services can be optimized by technical solutions. Examples include the optimization of shipping by channel constructions or the storage of drinking water in reservoirs. At the same time, technical solutions can partly mediate resilience of humankind against water-related threats. Diking, for example, protects against floods, reservoir construction can buffer water scarcity or flood intensities, and desalinization can alleviate water shortage in urban areas. Although such technical approaches can provide powerful engineering solutions in water resources management and can optimize access to aquatic ecosystem goods and services, they come with environmental costs and trade-offs. Diking separates riparian zones and wetlands from river dynamics and thus affects regulatory and supporting services such as nutrient retention and fertilization of riparian soil. Furthermore, it accelerates downstream confluence of water masses exacerbating flood risks at lower river reaches. Dam construction interferes with the sediment transport in rivers, causing massive downstream erosion, and interrupts the river continuum, which leads to the collapse of migratory populations, particularly fish. Managers and decision-makers must thus carefully consider and balance such trade-offs among ecosystem services when technical solutions are favoured.

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31 Groundwater Ecosystems and Their Services: Current Status and Potential Risks

Christian Griebler, Maria Avramov, and Grant Hose

31.1 Introduction

Earth's ecosystems provide a multitude of goods and functions, recently conceptualized under the term *ecosystem services* (ES; [1]). Many of these services and their associated ecosystems have received considerable attention, but groundwater ecosystems, soils, and sediments that are hidden below our feet are often overlooked. In fact, subsurface ecosystems deliver services of immense societal and economic value, most prominently the purification of water through (1) nutrient cycling; (2) biodegradation of contaminants; (3) inactivation and elimination of pathogens; and (4) storage and transmission of water that can mitigate floods and provide a stable water supply during droughts. Several of these services are directly connected to the presence and activity of the microorganisms and metazoans living in groundwater. We argue that, due to global and climate change, many of the groundwater ecosystem services are at serious risk. The pressures on groundwater ecosystems include aspects of global change such as local (point) and diffuse (non-point) sources of contamination, and overexploitation of groundwater resources. Moreover, even though groundwater ecosystems are located below ground, their organisms and the services they provide are affected by climate change—*inter alia* through changes in temperature regime as well as changes in recharge patterns and hydrological conditions due to floods and droughts.

Which ecosystem services are addressed? Provisioning: Provision and storage of clean water for domestic, agricultural, and industrial uses; geothermal energy for heating and cooling; provision of habitat and refuge for species that cannot survive elsewhere.

Regulating: Water purification (biodegradation of contaminants and elimination of pathogens); nutrient cycling; buffering of floods and droughts; sustaining the water balance of groundwater dependent ecosystems; bioturbation.

Cultural: Biodiversity (rare and endemic species; pool of functions and genetic resources; bioindicator species and species for ecotoxicological testing); caves and springs with spiritual, religious, and/or aesthetic value.

What is the research question addressed? Which ecosystem services do groundwater ecosystems provide and what are the main anthropogenic threats that act upon the provision of these services?

Which method has been applied? Literature review.

What is the main result? Groundwater ecosystems provide several ecosystem services that are of vital importance to humanity (listed in the text). Anthropogenic pressures act upon the provision of these services through direct and indirect mechanisms. An overload with pressures often leads to a decrease or loss in the provision of ecosystem services.

What is concluded, recommended? Wise and sustainable management of ecosystem services must take into account the processes and biotic interactions within the ecosystems and make sure that the ecosystems' capacity to deal with stressors is not exceeded.

31.2 Important Services Provided by Groundwater Ecosystems

Groundwater ecosystems provide essential services to humanity: they store and supply the majority of the water used for drinking and irrigation (*see* Chap. 32), they provide geothermal energy (heat and cold), and balance hydrological extremes by receiving surface water during floods and returning it to streams during droughts (Fig. 31.1). The base flow of rivers, lakes, and wetlands, particularly in dry climates, is often sustained by the discharge of groundwater. Of utmost importance are the biogeochemical processes that are mediated by groundwater microorganisms. These processes

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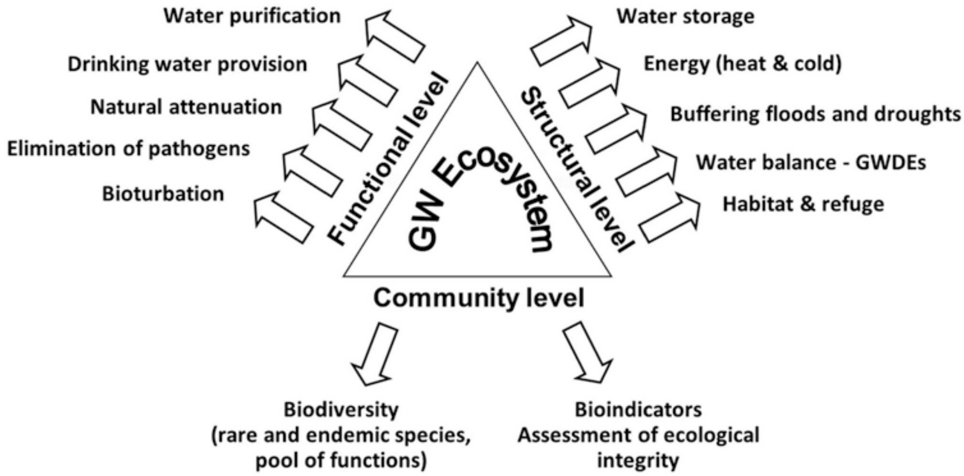

Fig. 31.1 Groundwater ecosystem services sorted according to the ecosystem levels that they mainly depend on. Many of the services can directly be translated into ecosystem goods (e.g., clean water). (*Modified from* Griebler et al. [21]; *with permission*)

include the cycling of organic carbon and nutrients, the biodegradation of contaminants, and the elimination of pathogenic microorganisms and viruses (disease control) [2].

31.3 Threats to Groundwater Ecosystem Services

On a global scale, there are three critical threats to groundwater resources and consequently to groundwater ecosystems. These threats are (1) contamination with anthropogenic chemicals, nutrients, and heat; (2) overuse and overabstraction of groundwater; and (3) climate change. Each of these threats alone has the potential to alter the structure and functioning of groundwater ecosystems, and hence the delivery of ecosystem services. In reality, however, the threats occur simultaneously (Table 31.1). The following sections describe each major threat, as well as its potential impacts on the provision of ecosystem services.

31.3.1 Contamination of Groundwater Ecosystems

Growing industrialization, waste deposition, and the exponentially increasing production and use of synthetic chemicals (currently over 80 million registered), places ecosystem goods and services at serious risk (Table 31.1) [3]. Indeed, groundwater quality is poor in many areas of the world due to contamination with nutrients (e.g., nitrate), pesticides (e.g., triazines), and pathogens (e.g., fecal bacteria and viruses) (Fig. 31.1). Additionally, there are several million sites worldwide—such as industrial sites, landfills, and cemeteries—that are heavily contaminated and await cleanup. Emerging contaminants of concern are also increasingly detected in aquifers [4]. These include micropollutants such as pharmaceuticals, personal care products, artificial sweeteners, and nanoparticles; see Table 31.2. The direct effect of these chemicals on groundwater biota and ecosystem service provision is unknown.

Although aquifers have the capacity to purify incoming water to a high quality, this ecosystem service is based on a sensitive balance between the low microbial biomass and activity in aquifers and the flux of organic carbon and nutrients to the aquifer. Groundwater ecosystems can buffer inputs of dissolved organic carbon and nutrients by increasing microbial biomass and activity. This is particularly the case at the boundary of the saturated and unsaturated zones, as well as in the hyporheic zones. However, an overload with carbon or nutrients can exceed the ecosystem's capacity for "natural attenuation" [5] and thus compromise the provision of high quality water.

For many contaminants (e.g., heavy metals, nanoparticles, hydrophobic compounds), aquifers are places of sorption and thus storage. Sorption and/or complexation of contaminants in many cases are redox sensitive. Turning aquifers from oxic to anoxic conditions may lead to the remobilization of contaminants (e.g., arsenic, uranium, phosphorus), putting groundwater users and connected ecosystems at risk. In summary, aquifers are not able to endlessly increase their attenuation and biodegradation services and simultaneously provide clean water.

An overload of the ecosystem's capacity to provide ecosystem services can also occur with respect to the elimination of pathogens. Due to their vast dimensions, their relatively long water residence times, and large sediment surface area available for sorption, aquifers serve as unique bioreactors with great potential to naturally retard, inactivate, and eliminate pathogens [6]. However, climate change-induced short-term pulses of highly contaminated water in combination with alterations in recharge-pattern (see below) can reduce an aquifer's capacity to eliminate pathogens.

Micropollutants are a particularly important and emerging issue within the broader threat of aquifer contamination.

Table 31.1 Selected groundwater ecosystem services and examples for the various anthropogenic risks they are exposed to

Groundwater ecosystem service(s) at risk	Risk	Cause(s)	Involved mechanism(s)
→ Storage and provision of clean water (for domestic, agricultural and industrial uses) → Mineral water, hot springs, recreation and tourism, spa, caves and springs of cultural importance	Contamination with chemicals	Contamination of aquifers	Entry into the groundwater from various sources and via different entry paths (see Table 31.2)
	Increase in chemical concentrations	• Groundwater table drawdown as a result of overexploitation • Climate change	• A lower volume of water available in the aquifer to dilute contaminants • Reduced groundwater recharge due to droughts; contaminated stormwater runoff during floods
	Persistence of chemicals in groundwater	Low concentrations of micropollutants	Inefficient biodegradation by microorganisms due to thermodynamic, enzymatic, and transport-related limitations
	Change in redox state (oxygen depletion)	Nutrient loading (DOC)	Oxygen depletion due to microbial biodegradation
	Mobilization of heavy metals and other contaminants	• Oxygen depletion/ change in redox state • Changes in pH	• Microbial biodegradation processes (e.g., as a result of organic contamination) • A shift in the lime/carbonic acid equilibrium due to temperature changes (e.g., as a result of geothermal energy usage)
	Nutrient loading (nitrate)	Contamination of aquifers from areas used for agriculture	Entry into the groundwater via different entry paths
	Saltwater intrusion in coastal areas	Increasing groundwater demand during droughts as a result of climate change	Groundwater table drawdown as a result of overexploitation
	Contamination with pathogens	Contamination of aquifers	Entry into the groundwater from various sources and via different entry paths (see Table 31.2)
	Reduced flow velocity and aquifer permeability	Clogging of sediments due to decreased bioturbation by groundwater fauna	Disturbance to or loss of groundwater fauna (e.g., due to toxic pollution, changes in natural temperature regime, oxygen depletion, etc.)
→ Biodegradation of contaminants and elimination of pathogens	Reduced efficiency	Overload, short-term and extreme pulses	Degrader communities do not have enough potential and time to react
→ Buffering of floods and droughts → Sustaining the water balance of groundwater-dependent ecosystems (GDEs)	Reduced water storage capacity // Insufficient amount of available water	Droughts; Land sealing // Groundwater overdraft; Land sealing	Reduced water uptake capacity of dry soils during droughts // Overexploitation; reduced groundwater recharge due to land sealing
→ Biodiversity (rare and endemic species, pool of functions, potential for the discovery of new processes/future knowledge, etc.) → Provision with bioindicator species (e.g. for biomonitoring) and species for ecotoxicological testing → Bioturbation (to maintain sediment permeability)	Loss of species	• Toxic pollution • Oxygen depletion • Changes in natural temperature regime • Loss of habitat • Competition with incoming surface water species	• Acute and chronic toxic effects on groundwater organisms • Microbial biodegradation processes (e.g., as a result of organic contamination) • Heating and cooling due to geothermal energy usage; climate change • Water table fluctuations/drawdown due to groundwater abstraction • Intrusion of surface water species during floods

Since they occur at very low concentrations, they cannot be degraded efficiently by microbes, and hence persist in the environment for long periods. Consequently, chronic toxic effects become a threat for the higher organisms (i.e., invertebrates) in the ecosystem. Being an integral part of the ecosystem, invertebrates certainly contribute to the provision of ecosystem services and do so by grazing upon and stimulating the microbial communities [7].

Another type of contamination is the discharge of heat into aquifers either via recharge of warm water because of cooling or due to geothermal energy usage [8]. Such temperature alterations impact the activity, structure, and composition of microbial and invertebrate communities, and also affect the provision of specific groundwater ecosystem services, although the interrelations between specific processes and ecosystem services are not fully understood. Apart from geothermal sources, long-term temperature changes in the subsurface may also result from climate change (see Sect. 31.3.3).

Table 31.2 Prominent pressures on groundwater ecosystems

Impact(s)	Source(s)
Legacy contaminants	
Petroleum hydrocarbons	Disposal of waste materials, leaks and spills, oil, gas and mining activities, hydraulic fracturing ("fracking")
Chlorinated compounds	Disposal of waste materials, leaks and spills
Radioactive substances	Disposal of waste materials
Heavy metals	Storm water runoff, leakage from landfills
Nutrients (nitrate, ammonia, organic carbon, etc.)	Agricultural practice, extreme hydrological events, waste water infiltration
Pathogens	Insufficiently treated wastewater effluents, sewage and wastewater disposal, storm water runoff
Contaminants of emerging concern	
Pesticides	Agricultural practice
Pharmaceuticals	Insufficiently treated wastewater effluents
Personal care products	Insufficiently treated wastewater effluents
Nanomaterials	Waste water, storm water runoff
Physical impacts	
Overexploitation	Irrigation, drinking water supply
Heating and cooling of groundwater	Geothermal energy use, heat discharge from cooling processes

31.3.2 Groundwater Drawdown

Abstraction of groundwater from many aquifers worldwide exceeds the natural renewal rate [9]. In 2000, 39% of the global yearly groundwater abstractions were overdraft (Fig. 31.2) [10]. The consequences of this overexploitation for humanity are obvious and are of great concern to governments all over the world, as reflected by the UN Millennium Declaration [11]. With respect to ecosystem services, lowering the groundwater level threatens the ability of the ecosystem to provide water for drinking and irrigation. Moreover, it can lead to salt water intrusion in coastal areas and land subsidence.

In comparison, the ecological consequences of groundwater abstraction for aquifers and groundwater-dependent ecosystems have received little attention. The drawdown of groundwater tables has devastating effects on natural streamflow and the water balance of groundwater-fed wetlands and related ecosystems. Furthermore, groundwater depletion leads to the loss of habitat [12], and in turn to losses of populations, species, and ecosystem processes and services [13, 14].

An example of an ecosystem service that is threatened by groundwater drawdown is the "attenuation of nutrients." While aquifers themselves are characterized by rather low biological activities, they share highly active transition zones with surface waters. The hyporheic and riparian zones, in particular, are places where nutrients (e.g., nitrate) from intense land use and agricultural activities in the catchment area are transformed on their way towards the surface waters [5]. The tremendous loss of river-associated wetlands and riparian areas due to regulation activities and lowering of groundwater tables significantly reduces this reactive volume.

Last, the open corridor between surface waters and aquifers (via the hyporheic zone) enables surface fauna to find temporary refuge in times of harsh conditions, e.g., during floods, droughts, or temperature elevations caused by anthropogenic impacts and global warming [15]. The transient or permanent disconnection between surface waters and aquifers will hamper this important ecosystem service [16].

31.3.3 Climate Change Effects

Climate variability and change influences groundwater ecosystems in many ways. Indirectly, a warmer climate and higher frequency and duration of droughts lead to higher irrigation demands, which govern current groundwater use and depletion [17, 18]. A direct consequence of climate change is also the quantitative and qualitative alteration of groundwater recharge patterns. Specifically, climate change leads to more frequent and intense hydrological extremes—droughts and floods—in many areas. Since the base flow of surface waters and wetlands is often sustained by the exfiltration of groundwater, extended periods of droughts lead to drier riverbeds and wetlands. Moreover, dry soils have a reduced capacity to absorb precipitation. As a result, more rain water flows into surface waters via surface runoff, instead of recharging the aquifers. Even a total net increase in precipitation, as predicted for humid regions, will not necessarily provide more groundwater recharge, if the rain falls at times when it is directly lost by evapotranspiration.

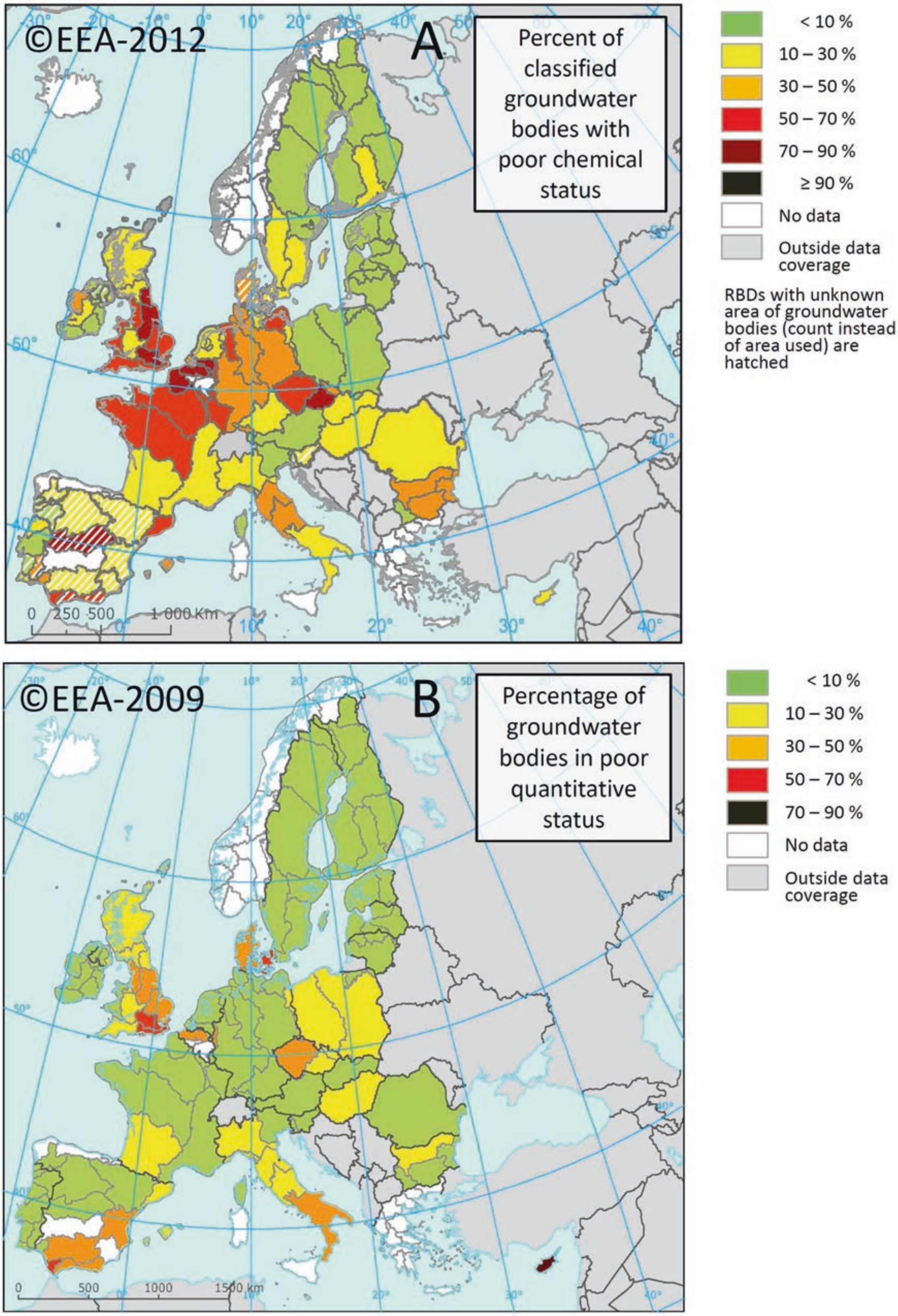

Fig. 31.2 Groundwater ecosystems and services provided by their organismic communities are under serious risks due to an increasing pollution and overexploitation. (**a**) Percentage of European groundwater bodies at poor chemical status in 2012. (**b**) Percentage of European groundwater bodies at poor quantitative status in 2009. (*From* European Environment Agency http://www.eea.europa.eu/data-and-maps/figures#c5=&c15=all&c0=15&b_start=0&c8=groundwater; *with permission*)

At high latitudes and elevations, global warming changes the spatial and temporal distribution of snow and ice. Warming results in a decreased snow accumulation and an earlier onset of snowmelt, as well as in increased winter precipitation in the form of rain, causing a reduced seasonal duration and magnitude of groundwater recharge [18].

Another issue is that irregular, extreme rain events in urban areas also cause high amounts of surface runoff. This water is collected in overflow channels, mixed with untreated wastewater, and discharged into rivers and streams, thus releasing various contaminants and pathogens untreated into the environment. Similarly, groundwater recharge from heavy rainfalls and floods can cause contamination of shallow aquifers, and thus can seriously impact groundwater quality [19]. Moreover, the synergistic effects of increases in temperature and pollution with nutrients (e.g., ammonia) and mixtures of pesticides may alter the sensitivity and thus the survival rate of invertebrate species, with detrimental effects to biodiversity and ecosystem services [20].

31.4 Conclusions

As briefly compiled in this chapter, various qualitative and quantitative impacts to groundwater ecosystems pose direct and indirect risks to the provision of its ecosystem services. Many of these services are related to human well-being and the health of groundwater-dependent terrestrial ecosystems and groundwater-associated aquatic ecosystems. Box 31.1 summarizes our main conclusions.

Box 31.1

- Groundwater ecosystems provide essential services to humanity, and many of these services depend on the organisms living in groundwater and on their biotic interactions.
- Various anthropogenic pressures threaten the provision of ecosystem services through direct and indirect mechanisms.
- An overload with pressures often leads to a decrease or loss in the provision of ecosystem services.
- Wise and sustainable management of ecosystem services must take into account the processes and biotic interactions within the ecosystem and make sure that the ecosystem's capacity to deal with stressors is carefully managed and not exceeded in order to maintain the benefits—for today and for future generations.

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32 Drinking Water Quality at Risk: A European Perspective

Jeanette Völker and Dietrich Borchardt

Half of the drinking water consumed in Europe originates from groundwater resources and about one-third from surface waters. EU directives and national legislation should ensure that drinking water is safe and clean, and that drinking water resources will be protected sustainably against anthropogenic forces resulting from extraction, pollution, or inadequate drinking water infrastructures. Yet significant numbers of groundwater bodies and surface waters in Europe are at risk with regard to providing safe drinking water resources: high nutrient surplus in agriculture, emerging chemical contaminants, pathogens and antibiotic resistance, and the changes in the hydrological regimes pose major challenges for the management of drinking water resources (Fig. 32.1).

Which ecosystem services are addressed? Provisioning of drinking water, soil and water quality regulation, nitrogen retention (denitrification), hydrological regulation.

What is the research question addressed? What are the potential risks posed upon drinking water (groundwater) resources and the challenges of water management?

Which method has been applied? Literature review; assessment of European drinking water data (reporting period 2011–2013).

What is the main result? The drinking water quality in Europe is still good, but rising nitrate concentrations, pesticide contamination from agriculture, and changes in the hydrological regime pose increasing risks on drinking water resources.

What is concluded, recommended? In order to protect drinking water quality as a resource, the resilience of the groundwater and surface water ecosystems should be strengthened by a consistent reduction of the pollution pressures, and responsible polluters should be actively involved in the water resources management strategies.

32.1 Drinking Water: Ecosystem Service and Management

According to the Millennium Ecosystem Assessment [1], ecosystem services are the benefits that people receive from ecosystems. These include the provision of services such as food or water and the regulation of services such as regulation of flooding, drought, and soil degradation. These ecosystem services are highly relevant to the resource of drinking water. Thus, the high attenuation capacity of soils and aquifers is based on the assimilation capacity of organic and inorganic compounds. These include numerous biological, chemical, and physical processes such as the elimination of nitrate via denitrification and the aerobic or anaerobic conversion or removal of organic components.

The provision of drinking water in sufficient quantity and quality is therefore a key challenge for sustainable water resources and aquatic ecosystem management. This management comprises a proper regulation of the hydrological system (e.g., runoff, infiltration, storage, aquifer recharge) and controls on water quality (pollution prevention, control, and treatment). Key elements for the improvement of environmental conditions comprise: controlling sources of hazardous or otherwise dangerous constituents including pathogens; avoiding excessive nutrient load in surface water and groundwater by limiting fertilizer application in agriculture; treatment of wastewater; improving hydromorphology; and protecting wetlands.

32.2 Legal Framework

The European Drinking Water Directive (98/83/EC, DWD) came into force in 1998 with the objective to provide water for human consumption that is, from a quanti-

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Fig. 32.1 Water tap

tative and qualitative perspective, wholesome, clean, and safe. Water must therefore be provided in sufficient quantities and, at the same time, be of sufficient quality with regard to microorganisms or substances that could potentially endanger human health. The directive applies to all waters intended for human consumption. The Drinking Water Directive (1) sets quality standards for 48 drinking water quality parameters at the tap; (2) obliges Member States to monitor drinking water quality regularly and to take remedial actions where required; and (3) obliges members to provide consumers with adequate and up-to-date information on drinking water quality.

Drinking water is also an issue of the European Water Framework Directive (2000/60/EC, WFD). Here, the water-protected areas need to be identified and reported within the river basin management plans. In this regard, the Water Framework Directive should promote sustainable water use based on the long-term protection of water resources, and it contributes to the provision of sufficient supply of high-quality sources of drinking water for human consumption.

While the Drinking Water Directive considers the raw water entering the drinking water systems, all storage and treatment infrastructures, and systems that distribute water to the consumer, the Water Framework Directive covers the water abstraction source, its catchment, and the wider environment. According to both of these European legislations, a straightforward link between drinking water quality and surface water or groundwater status assessments under the Water Framework Directive is currently missing. In February 2018, the European Commission adopted a proposal for a revised Drinking Water Directive also to strengthen the connection between different directives.

32.3 Potential Risks to Raw Water and Drinking Water Quality: European Results

Significant pressures for groundwater as well as surface waters, with high potential risks to drinking water quality, include nutrient input, chemical pollution, or microbiological pollution. These pressures are caused by diffuse sources like agriculture, and point sources like effluents from wastewater treatment plants. Furthermore, water quantity and the sustainable balance between recharge and extraction play a major role with respect to drinking water quality.

32.3.1 Drinking Water Sources

Figure 32.2 shows the distribution of sources of drinking water in Europe.

The main source for drinking water in the European Union's 27 member states (reference years 2011–2013) is groundwater, which provides 50% of the total resource use, and surface waters (37%), such as lakes, reservoirs, and rivers. Bank filtration and artificial groundwater recharge as modified water systems play a minor role in drinking water sources in the EU [2].

Because of the natural filtering and attenuation function of the soil, groundwater may require only disinfection to ensure adequate raw water quality (depending on the grade and type of contamination). Surface water normally requires more extensive treatment, like coagulation, sedimentation, and filtration, in addition to disinfection. What is not considered in Fig. 32.2 is the fact that some 10% of the EU population uses untreated groundwater, particularly from private wells [3].

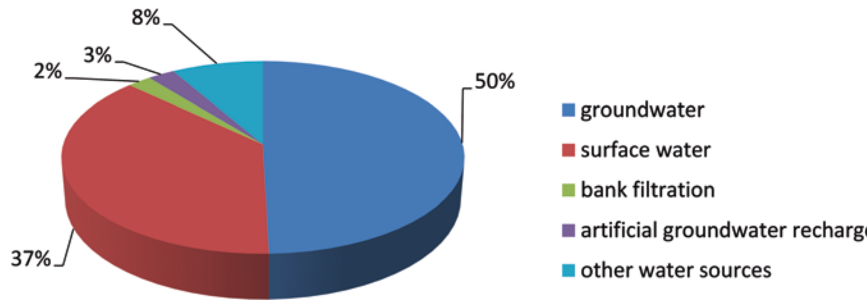

Fig. 32.2 Sources of drinking water in the EU, 2011–2013. (*Adapted from* European Environment Agency [2]; *with permission*)

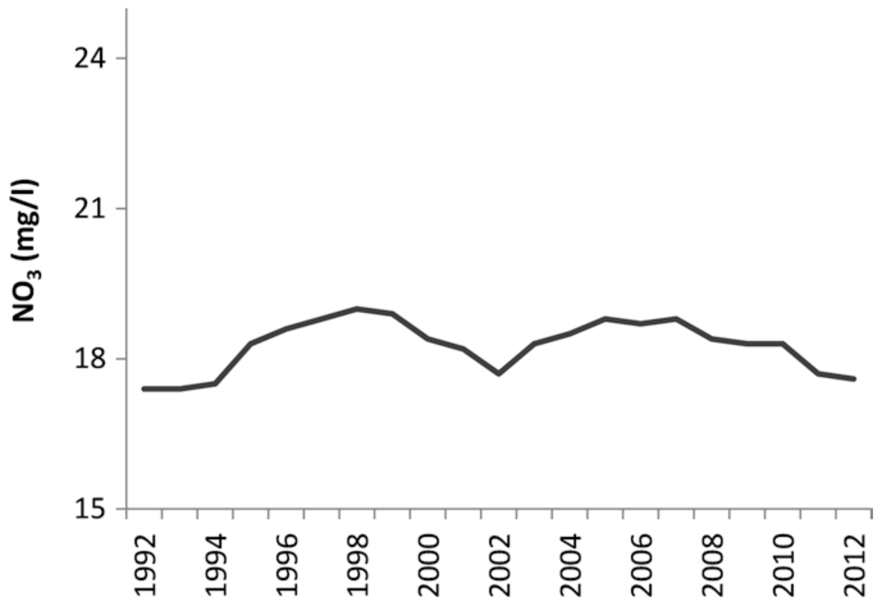

Fig. 32.3 Nitrate concentration in Europe's groundwater. (*Modified from* European Environment Agency (EEA) http://www.eea.europa.eu/data-and-maps/indicators/nutrients-in-freshwater/nutrients-in-freshwater-assessment-published-6; *with permission*)

This means that in the EU, clean drinking water must be ensured for more than 250 million people whose source of drinking water is groundwater, and for some 150 million people whose source of drinking water is surface waters. The demand for water for public supply in Europe comprises some 55,000 millions of m^3 per year. This corresponds to a share of 21% of the total abstracted water of 288 km^3 per year in the EU, which is used from natural water sources [4].

32.3.2 Nutrient Pollution

The intense use of manure and mineral fertilizers leads to high nutrient surpluses in agricultured soils which are washed out into the groundwaters underneath. For example, the results of the first river basin management plan according to Water Framework Directive show that a poor chemical status of groundwater is attributable to excessive nitrate concentrations in 54% of all observed groundwater bodies [5]. Furthermore, results of nitrate concentration in Europe's groundwater from 1992 to 2012 shows high concentration level between 17.4 and 19.0 mg/L NO3 and giving hardly any trend overall (Fig. 32.3). For comparison, the natural nitrate concentration in groundwater ranges between 2 and 8 mg/L NO3, and depends strongly on soil type and on the geological situation.

According to surface waters, elevated nutrient inputs from agriculture and municipal wastewater contribute to water quality degradation in many parts of Europe. The diffuse pollution from agriculture is still significant in more than 40% of all European rivers, while point source pollution caused by storm water overflow as well as by discharges from wastewater treatment plants affected 20–25% of the surface waters in Europe [5].

32.3.3 Chemical Pollution

The main drivers of chemical pollution in surface waters and groundwater are urban development, industry, agriculture, and mining. In surface waters, point sources from urban development and industry predominate, i.e., discharges from municipal and industrial wastewater treatment plants resulting in imissions of heavy metals and industrial chemicals. In

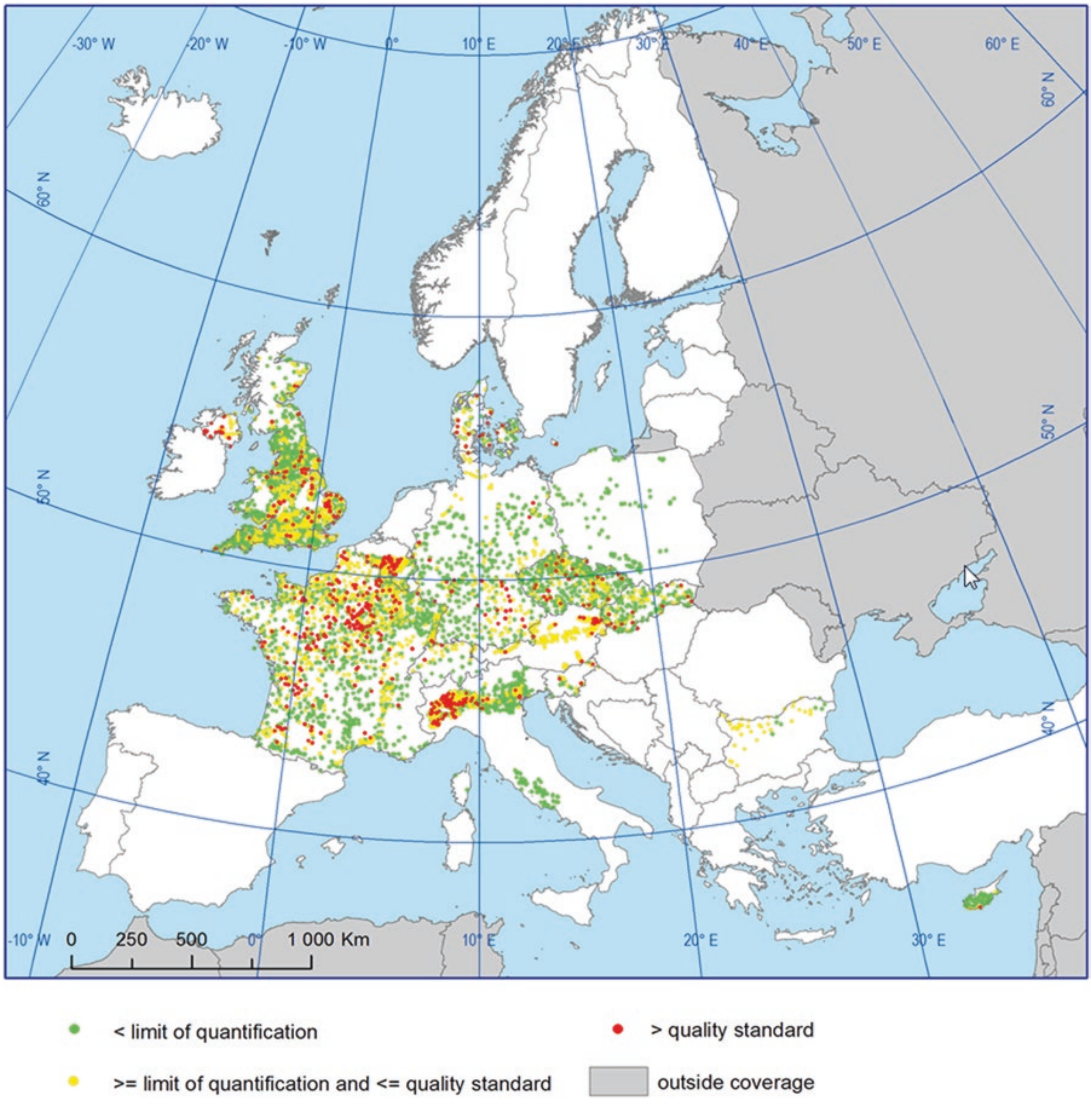

Fig. 32.4 Occurrence and exceedance of selected pesticides in groundwater monitoring stations, 2010–2011. European Union, 1995–2013. (*From* http://ec.europa.eu/eurostat/statistics-explained/index.php/File:Occurrence_and_exceedance_of_selected_pesticides_(listed_in_Figure_1)_in_groundwater_monitoring_stations,_2010-2011.png; *with permission*)

groundwater, diffuse chemical pollution from agriculture via surface run-off and leaching plays a major role, mainly caused by pesticides and veterinary medicines from farmlands.

According to the results of the first river basin management plans of the Water Framework Directive, less than 20% of all surface water bodies in the EU were contaminated by chemical substances. Sixty percent of groundwater bodies in poor chemical status are characterised by an exceedance of a quality standard (threshold value) for one or more pollutants like pesticides [5].

Figure 32.4 shows the occurrence and exceedance of 31 selected pesticides in groundwater monitoring stations from 2010 to 2011. Several countries in Europe report that groundwater has concentrations of pesticides that exceed the quality standards according to chemical status assessment of the Water Framework Directive.

The chemical parameters used to assess drinking water quality were selected for their potential impact on human health, and they do not closely match the list of priority substances under the Water Framework Directive. The 26 Drinking Water Directive parameters include trace elements, such as arsenic, nickel, and lead, and other substances such as cyanide, polycyclic aromatic hydrocarbons, and nitrogen compounds (nitrate and nitrite). Results based on the last reporting (2011–2013) of drinking water quality in the EU

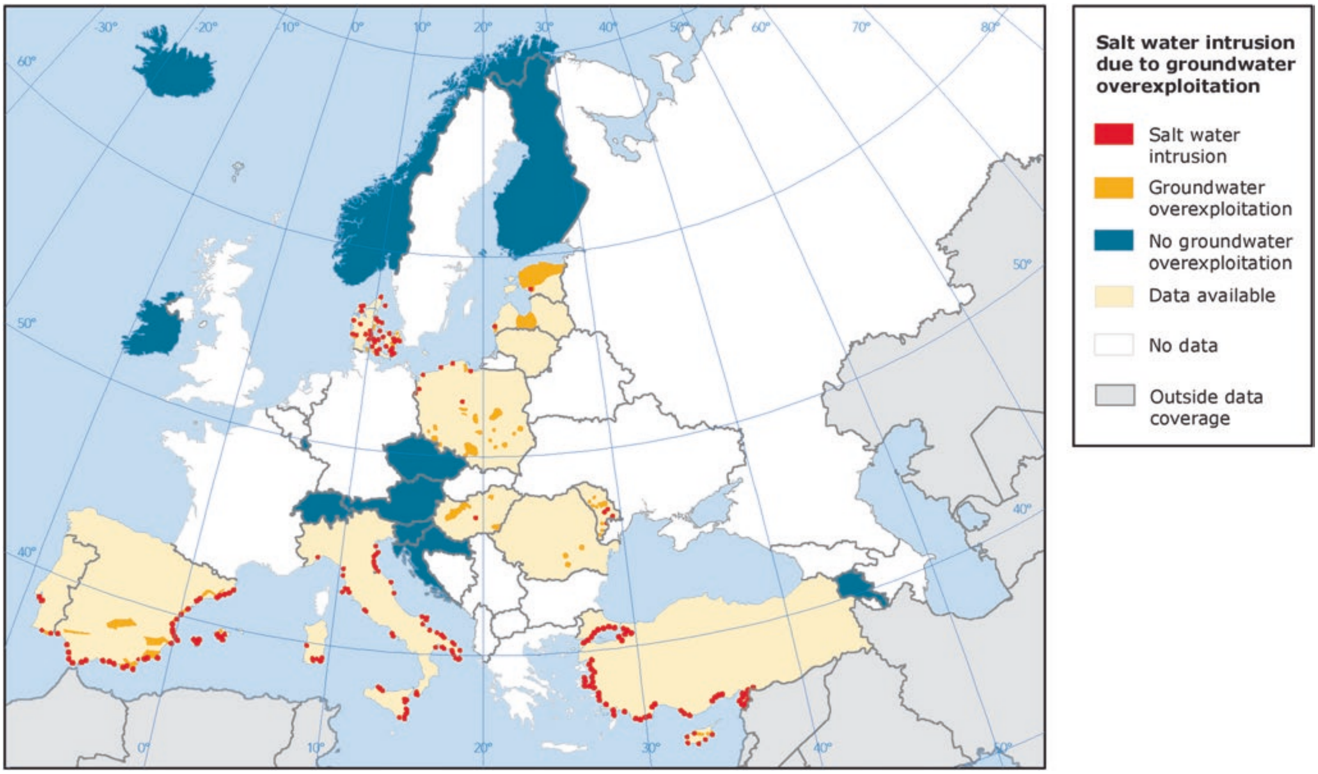

Fig. 32.5 Groundwater overexploitation and saline intrusion in the EU. © European Environment Agency (EEA). (https://www.eea.europa.eu/data-and-maps/figures/groundwater-overexploitation-and-saltwater-intrusion-in-europe-2/groundwater_graphic31.eps/image_large; *with permission*)

countries showed that the majority of chemical parameters do not exceed threshold values. This is understandable because the water must be cleaned from these substances. But it also has been shown that groundwater pollution with pesticides increases and more treatment of the polluted raw water is needed.

32.3.4 Microbiological Pollution

Wastewater discharges in surface waters are the major source of microbiological pollution. In general terms, the greatest microbial risks are associated with ingestion of water that is contaminated with human or animal feces. Owing to legally binding standards, the quality of drinking water is ensured at least at the tap. The future challenge here is to ensure the quality of raw water so that clean water resources do not become scarce [6].

32.3.5 Water Quantity

Changes in land use, urban development, high amounts of water abstraction for irrigation, and climate change alter the natural regime of the hydrological cycle.

Results show, for example, that agriculture accounts for some 30% of whole water abstracted in Europe, and for up to 80% of whole water abstracted in parts of Southern Europe. Furthermore, since 1880, the average length of summer heat waves has doubled in Europe. It is predicted that these changes will continue over the coming decades in the EU [7].

Human-induced water scarcity leads to desertification and saltwater intrusion in groundwater in coastal zones, especially of Southern Europe (Fig. 32.5) .

At present, water scarcity affects some 10% of the European population. It has been estimated that 20–40% of the water in pipes is being wasted by, e.g., leakages in water supply systems and dripping taps. It is also predicted that water consumption by the public, industry, and agriculture will increase by about 16%. For these reasons it is necessary to implement measures to reduce water scarcity problems [7].

32.4 Conclusion

The large number of anthropogenic influences on natural water resources has a significant impact on drinking water quality in Europe. Drinking water as one of the essential ecosystem services is a vulnerable resource whose protection requires an integrated water management. "End-of-pipe solutions" such as an extensive water treatment are necessary but are not the sole solution. Rather, the resilience of the system should be strengthened by a consistent reduction of the pollution pressures. The responsible polluters should be actively involved in reduction measures and in charge of restoration initiatives, also in the context of the relevant legal frameworks.

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33 Pesticide Effects on Stream Ecosystems

Saskia Knillmann and Matthias Liess

33.1 Pesticides in Freshwater Ecosystems

Pesticides are currently applied on a large scale in agricultural crops, but also in urban areas, private gardens, and households. Pesticides enter freshwater ecosystems for example via surface runoff, spray drift, or wastewater treatment plants. The study from Ippolito et al. shows that more than 40% of the global land area is at risk to insecticide, as displayed in Fig. 33.1. The authors modelled insecticide exposure using the runoff potential model [7]. Up to 18% of the global land area is predicted to cause a high to very high insecticide runoff into draining freshwaters. Parameters that contribute to a high runoff potential are predominantly pesticide use, proportion of cropland, precipitation, slope and soil characteristics. For validation of exposure, the authors compared the predicted runoff potential with measured pesticide concentrations in streams from field studies in Europe and Australia.

While the runoff potential model mainly represents a risk potential of certain regions towards pesticide exposure, high pesticide concentrations in freshwater systems have also been reported in several studies. Examples are included in a recent study by Stehle and Schulz [8] that detected insecticide concentrations exceeding regulatory thresholds in 50% of the investigated concentrations at a global scale. Also, Malaj et al. [9] reviewed the available exposure monitoring studies of organic pollutants in European freshwater systems and identified pesticides as one of the major contributors to toxicant exposure of freshwater ecosystems.

Which ecosystem services are addressed? Water quality.

What is the research question addressed? What is the risk of pesticides for stream communities and how is it expected to develop under global climate change?

Which method has been applied? Literature review.

What is the main result? Streams in more than 40 % of the global land area can be affected by insecticide runoff. Pesticides have negative effects on biodiversity and the ecosystem function leaf litter degradation. Finally, the ecological risk due to insecticides is expected to increase under climate change.

What is concluded, recommended? Despite existing regulations, pesticides present a major stressor for stream ecosystems. More realistic effect predictions are necessary and effective mitigation measures can be implemented at the landscape level or reduce the use of pesticides.

33.2 Impacts on Invertebrate Communities, Biodiversity, and Ecosystem Functions

Effects on aquatic invertebrate communities could be linked in several field studies to measured pesticide concentrations. These field observations in streams show a decline of the trait-based indicator $\text{SPEAR}_{\text{pesticides}}$ in Germany [12], but also across different biogeographical zones in Europe, Russia, and Australia [10].

The trait-based indicator $\text{SPEAR}_{\text{pesticides}}$ represents the ratio of pesticide vulnerable against invulnerable taxa and gives a measure for the freshwater ecosystem effects of pesticides [1]. In addition to structural changes in the invertebrate communities, the study by Beketov et al. [3] observed a strong decline in biodiversity due to pesticide exposure for different biogeographical regions in Europe and Australia (Fig. 33.2). The authors highlight that species diversity significantly decreased to 58% at sites with high pesticide exposure compared to sites with low pesticide exposure.

Relevant ecosystem functions in headwater streams comprise especially leaf litter degradation and primary production as basic energy sources. A reduction of these ecosystem functions can, in turn, affect ecosystem services such as water purification. In contrast to primary production, leaf lit-

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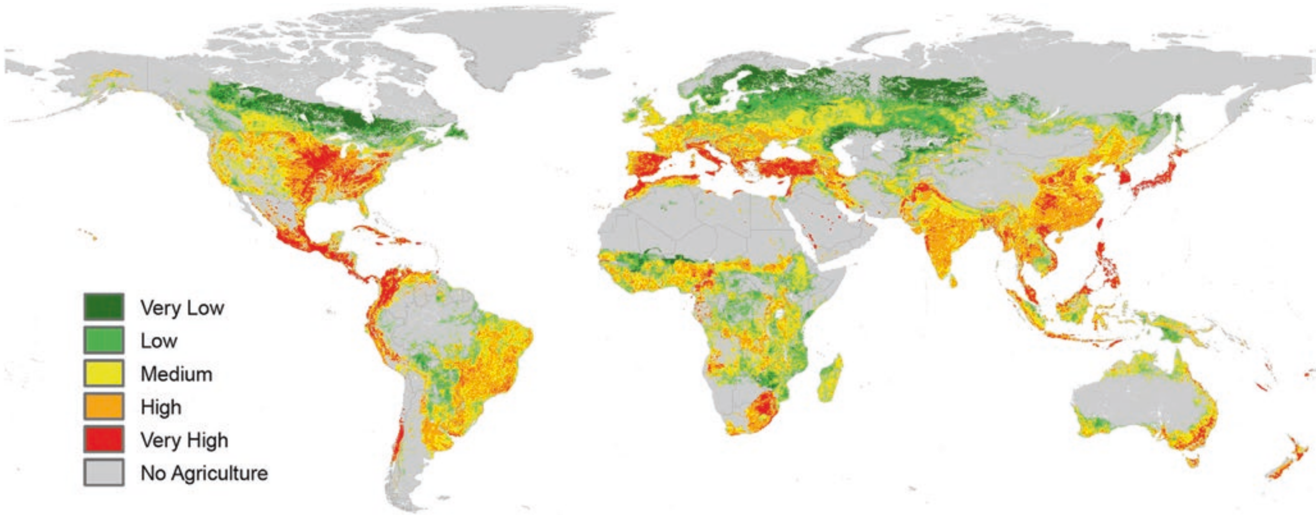

Fig. 33.1 Global insecticide runoff potential map. The map shows the spatial distribution of potential insecticide runoff to stream ecosystems considering agricultural activities, geomorphological and climatic conditions. The class boundaries of the runoff potential (−3; −2; −1; 0) follow the same definition as in [6]. (*Reprinted from* Ippolito et al. [2]; *with permission*)

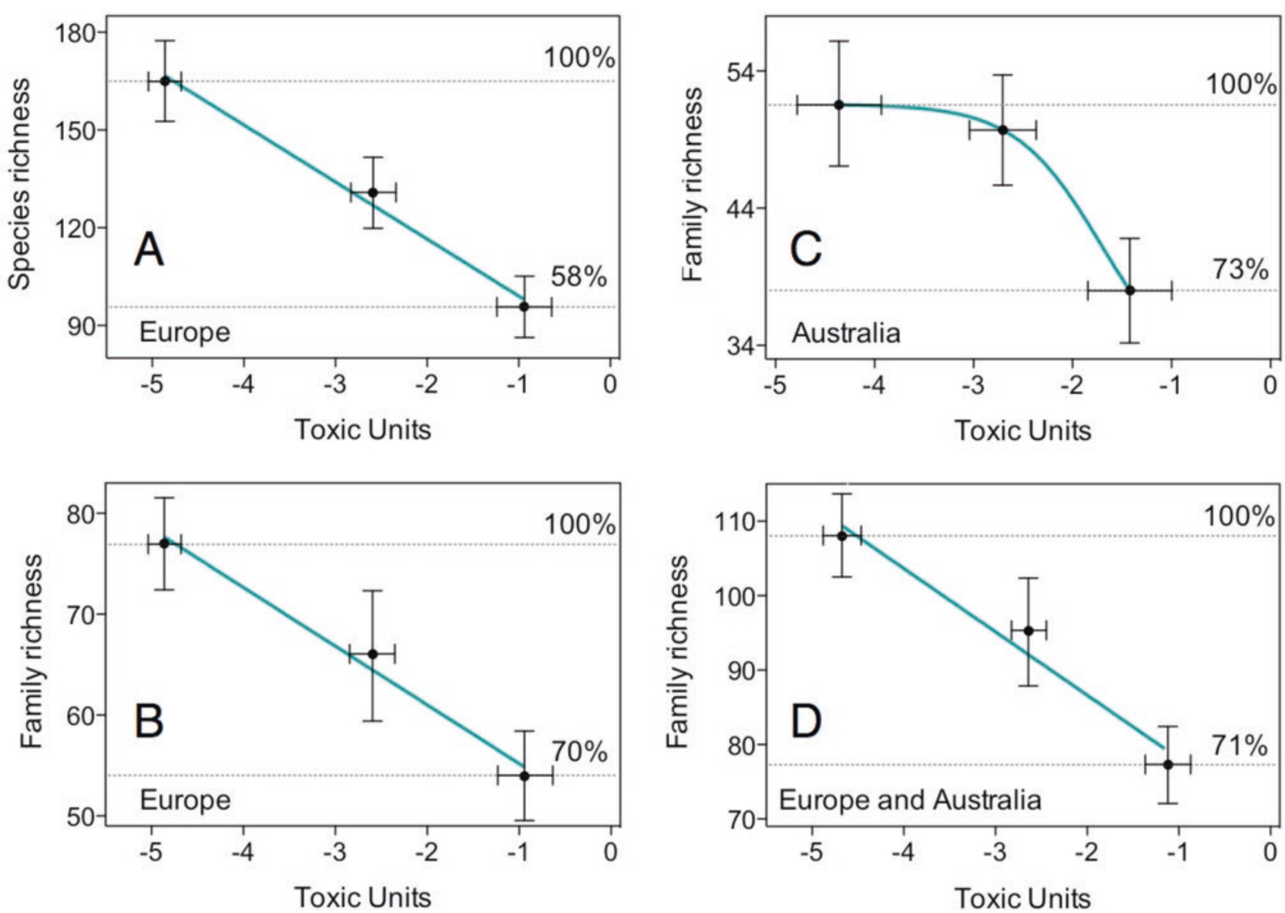

Fig. 33.2 Concentration–response relationships between the pesticide concentration (Toxic Unit) and mean overall taxa richness of stream invertebrates. The relationships are given for species and family richness in the investigated regions Europe (**a**, **b**), Australia (**c**) and the combined data set (**d**). Pesticide data have been classified in three groups according to the level of pesticide concentrations. Maximum and minimum taxa richness is displayed with dashed horizontal lines. (*Reprinted from* Beketov et al. [3]; *with permission*)

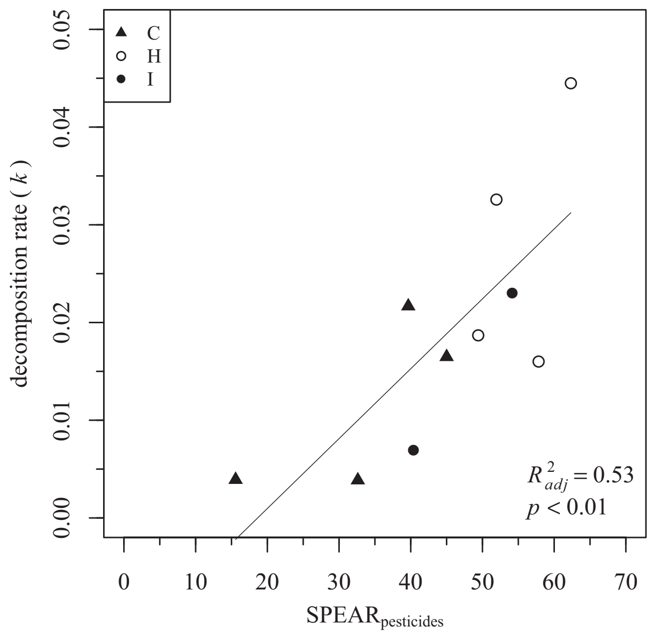

Fig. 33.3 Relationship between the leaf litter decomposition rate k and trait-based indicator SPEAR$_{pesticides}$. The relationship is based on ten stream sites in Germany. Regression line, R_2 and p-values describe the significant linear regression. *C* carbofuran, *H* herbicide, *I* insecticide other than Carbofuran. (*Reprinted from* Münze et al. [5]; *with permission*)

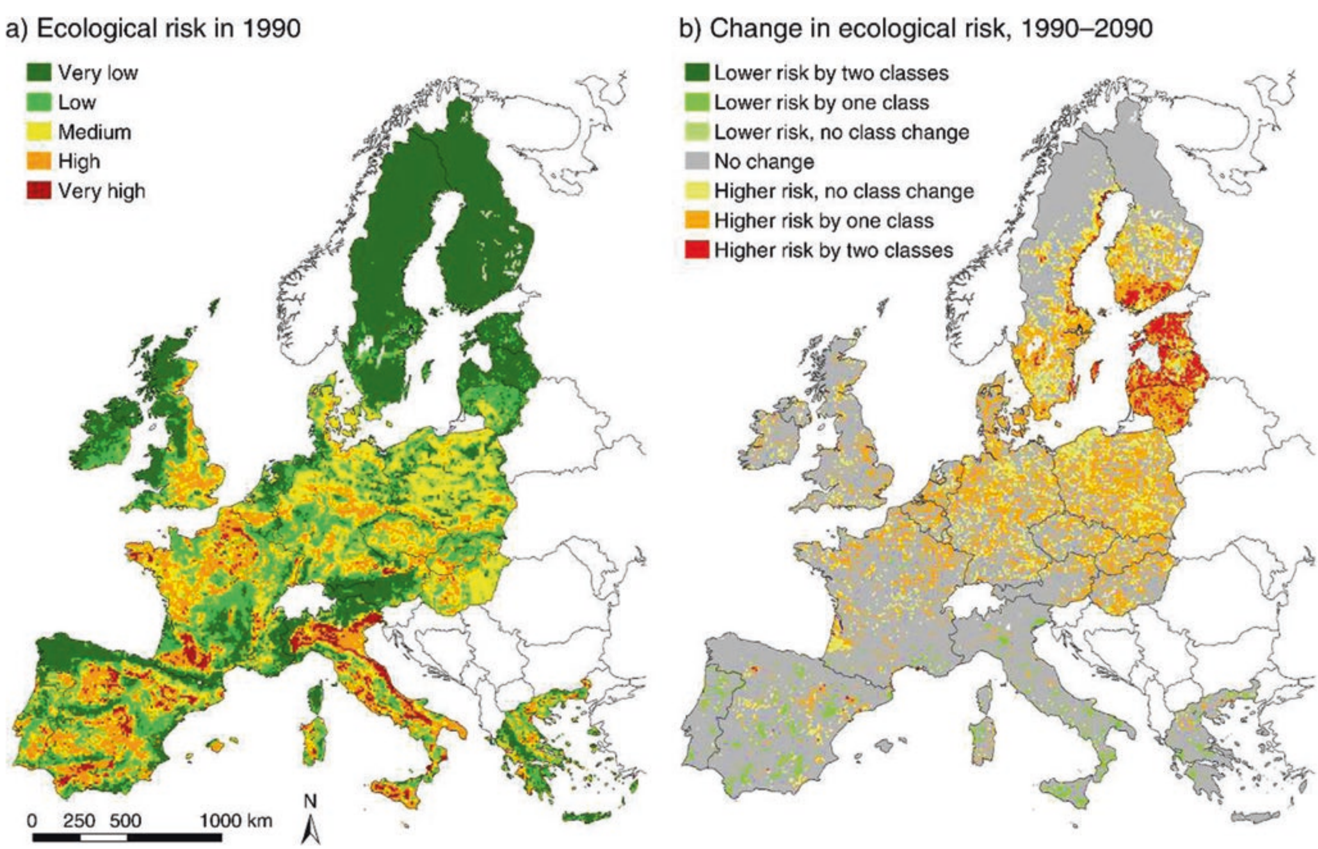

Fig. 33.4 (**a**) Ecological risk in 1990 and (**b**) change in ecological risk from 1990 to 2090 based on the A1B scenario. The ecological risk is based on the empirical relationship between runoff potential, recovery parameters, and changes in the invertebrate communities. The ecological risk has been classified to "very high," 75–100% of all stream sites within a cell were predicted to have an ecological status worse than "good"; for the class "high," 50–75%; for the class "medium," 25–50%; for the class "low," 10–25%; and for the class "very low," 0–10%. The change in the ecological risk represents the deviation from 1990 to 2090. (*Reprinted from* Kattwinkel et al. [6]; *with permission*)

ter degradation has been observed to decrease due to pesticide exposure in Australian streams [4]. Münze et al. [5] also observed a decrease in leaf litter degradation in German agricultural streams that was significantly correlated with the detected pesticide concentrations, but even stronger with a decreasing $SPEAR_{pesticides}$ as shown in Fig. 33.3.

33.3 Pesticide Effects Under Climate Change

Agriculture includes, in large part, the use of pesticides that depend to a significant extent on climate conditions and, hence, are predicted to change under global climate change. The analysis by Kattwinkel et al. [6] identified a positive relation between the mean annual temperature and the rate of applied insecticides across different European states. Applying space-for-time analyses, the authors used this link to predict a runoff potential under future climate and land-use scenarios for freshwater communities. Kattwinkel et al. [6] determined the ecological risk of stream ecosystems based on an empirical exposure-response relationship between the runoff potential, the presence of recovery areas, and the indicator $SPEAR_{pesticides}$. The authors concluded from the analyses that the ecological risk is especially increasing in Central and Northern Europe until 2090 due to increasing insecticide applications and land-use change (Fig. 33.4). Similar shifts in agricultural activities and increased pesticide exposure due to climate change is not only expected for Europe, but in general for higher latitudes and altitudes [11].

33.4 Conclusions and Recommendations

Despite strict regulations and registration procedures as, for example, implemented in the European Union and North America, pesticides in freshwater streams present a major stressor for stream invertebrates, including relevant ecosystem functions and services. Hence, we need to better understand pesticide effects under different environmental conditions. The underlying mechanistic knowledge of realistic effects is necessary to extrapolate from laboratory studies to the field using protective safety factors and effect models that predict current and future pesticide impacts. Regarding mitigation measures at the landscape level, riparian buffer strips and uncontaminated stream sections (refuge areas) have been proven to, respectively, reduce the pesticide exposure and impact on freshwater invertebrates. Other measures, like pesticide taxes, non-chemical alternatives, and the substitution of critical pesticides in terms of environmental effects and human health, focus on the use of pesticides. These measures present important tools to reduce pesticide use in general and the risk of pesticides for humans and non-target organisms.

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34 How Good Are Bad Species?

Sonja Knapp, Marten Winter, Andreas Zehnsdorf,
and Ingolf Kühn

34.1 What Are Biological Invasions?

Trade and traffic connect countries worldwide, crossing natural barriers such as the sea and mountains. By transporting living organisms across these barriers, humans have created pathways for the introduction of species to regions to which they are not native (Fig. 34.1). These introduced species are faced with "novel" environments; at the same time, the native ecosystems are faced with "novel" species. Some of the novel species will not manage to establish themselves, but will vanish again or only persist under human care. An estimated 25% of all introduced non-native plant and invertebrate species and 50% of all introduced vertebrates [1], however, establish themselves in their introduced range. The process from introduction via establishment to spread is called *biological invasion* (Fig. 34.2). Species that pass all steps of this process, with individuals dispersing, surviving, and reproducing at multiple sites across a greater or lesser spectrum of habitats and extent of occurrence [2], and that cause significant environmental, economic, or human health impacts, are called *invasive species* [3].

Which ecosystem services are addressed? A range of ecosystem services related to, e.g., horticulture, agriculture, and forestry, such as food production (provisioning service), water purification, climate regulation (regulating services), cognitive benefits, and ornamental value (cultural services).

What is the research question addressed? Which trade-offs exist between the services and disservices provided by non-native species?

Which method has been applied? Literature review.

What is the main result? Many non-native species have been introduced because they had a value for economy or society. Consequently, many non-native species provide both ecosystem services and disservices.

What is concluded, recommended? To better understand which species' characteristics control ecosystem functions and related ecosystem services, it is necessary to distinguish between those characteristics providing services and those promoting invasions or providing disservices.

34.2 Costs and Disservices of Invasive Species

Biological invasions can severely impact native biodiversity, ecosystem functions and their resulting services, as well as economic values. Two different mechanisms can result in negative impacts related to ecosystem services: (1) Disservices, which are impacts that can only have negative effects on a target system such as human health. An example is the high allergenic potential of *Ambrosia artemisiifolia* L., a plant species that was accidentally introduced from North America to Asia, Australia, Europe, and South America. (2) Loss of services, which in fact is the reduction of a positive service due to the presence of an invasive species, such as decreased crop yield caused by non-native weeds or loss of river bank soil stability through digging activities of invasive *Myocastor coypus* Molina.

More than 5% of all non-native terrestrial plants, roughly 15% of all non-native terrestrial invertebrates, and more than 30% of all non-native terrestrial vertebrates that have been recorded in Europe have published ecological impacts [4]. Known economic impacts have been assessed, e.g., for 24% of terrestrial invertebrates and 38% of terrestrial vertebrates. In Europe, the costs of alien species are estimated to be at least 12.5 billion Euros per year [5]; in Germany between 109 and 263 million Euros per year damage is caused by 20 selected invasive species [6]. Impacts of alien birds alone have been estimated at 24 billion US dollars per year for Australia, Brazil, India, South Africa, the UK and the US together [7].

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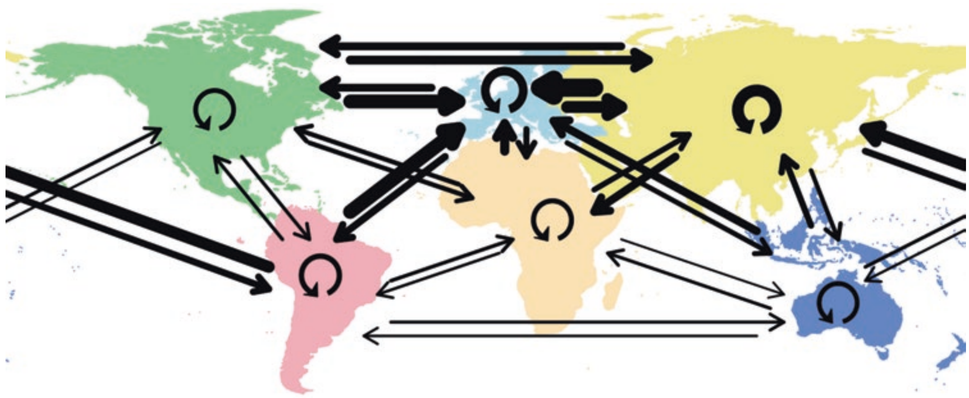

Fig. 34.1 Plant species' invasion pathways across the globe. The relative strength of the spread of naturalized plant species (i.e., species that established in the "novel" region) among and within continents (given in different colors) is indicated by the width of arrows. See Seebens et al. [10] for details. Copyright © Hanno Seebens, Senckenberg Biodiversity and Climate Research Centre (BiK-F)

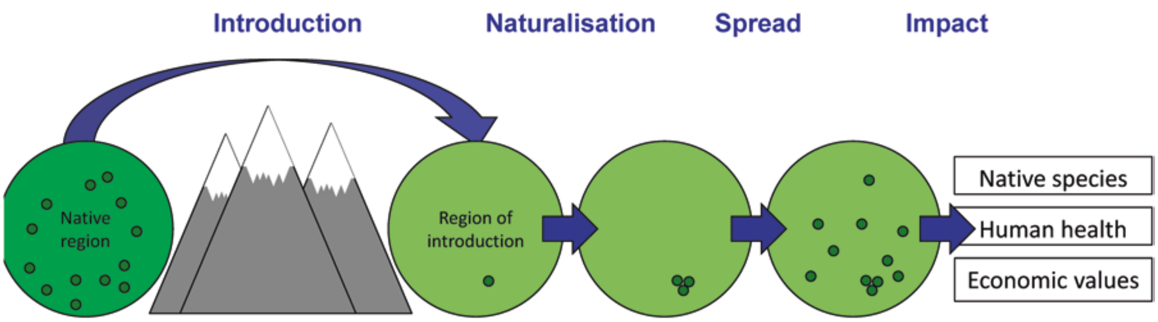

Fig. 34.2 The process of biological invasions and its potential ecological and socio-economic impacts

Controlling an invasive species can save ecosystem services provided by other species, and it can reduce costs – e.g., protecting ornamental trees from infestation by invasive Asian long-horned beetles in Northern Italy saved values worth six times the cost of protection [8].

34.3 Benefits, Ecosystem Services and Service-Disservice Trade-Offs of (Invasive) Non-native Species

The majority of non-native plant species were deliberately introduced for, e.g., horticulture (Tables 34.1 and 34.2), agriculture, or forestry [9], which means they were introduced because they had a value for economy or society. Wholesale trade with plants in Germany was, for example >4500 million Euros in 2015, and is expected to increase in the following years (http://de.statista.com), as global trade as such is expected [10].

Several non-native animals were deliberately introduced as well, e.g., fish and mussel species for aquaria or commercial breeding (Table 34.3), or mammals for fur production (Table 34.4). Even if not intentionally introduced for a specific purpose, alien species can offer services; for example, 3–4.5 million Euros worth of mitten crab was sold as food in Germany from 1994 to 2004 [4]. Irrespective of this double role of non-native species, only few frameworks exist that aim at assessing negative impacts and positive benefits simultaneously. Those that do provide a mixed picture of services and disservices. Katsanevakis et al. [11], for example, summarized that invasive marine species impact food provision in both negative and positive ways. Some of these species provide services (e.g., habitat for other species, cognitive benefits, water purification, and climate regulation), while others cause disservices (e.g., impacting tourism or native species). Species of the genus *Elodea* (Table 34.2), for example, can be harvested, and their biomass has the potential for different interesting uses such as the production of cosmetics [12]. *Elodea* species can also, however, cause secondary eutrophication and impede aquatic sports. They are difficult to manage in the case of a strong increase in a natural stretch of water [13]. Several options for management of invasive

Table 34.1 A terrestrial invasive plant species: *Ailanthus altissima* (Mill.) Swingle, Tree of Heaven, its native range, non-native range and examples of its ecosystem disservices and services. Cf. [15, 17] (Figs. 34.3 and 34.4)

***Ailanthus altissima* (Mill.) Swingle, Tree of heaven** Photograph: © Stefan Klotz, UFZ	
Native to: China (blue dots in Fig. 34.4) **Introduced to**: Africa, Asia, Australia, Europe, Japan, New Zealand, North America and South America (red dots in Fig. 34.4) Map data: © AG Chorologie, Institut für Biologie—Geobotanik & Botanischer Garten, MLU Halle-Wittenberg	
Ecosystem disservices	• Restricts the provision of habitat to native species by occupying these species' native habitat • Can cause dermatitis and other allergenic reactions • Roots can destroy pavement and other built structures
Ecosystem services	• Introduced as species with ornamental value • Wood production • Pharmaceutical use • Indicator of ozone concentration
Management options	• Propagule pressure can be reduced by not planting new individuals of *Ailanthus* • Repeated mechanic removal—but re-sprouting is possible • Herbicide application

Table 34.2 An aquatic invasive plant species: *Elodea nuttallii* (Planch.) H. St. John, Nuttal's waterweed, its native range, non-native range and examples of its disservices and services. Cf. [12, 13, 16] (Figs. 34.5 and 34.6)

***Elodea nuttallii* (Planch.) H. St. John, Nuttal'S waterweed** Photograph: © André Künzelmann, UFZ	
Native to: North America (dark blue dots in Fig. 34.6) **Introduced to**: Europe, Japan (red dots; Fig. 34.6) and parts of non-temperate North America (light blue dots; Fig. 34.6) Map data: © AG Chorologie, Institut für Biologie—Geobotanik & Botanischer Garten, MLU Halle-Wittenberg	
Ecosystem disservices	• Restricts the provision of habitat to native species by occupying these species' native habitat • Binds nutrients when growing and releases them when decaying, promoting secondary eutrophication • Impairs swimming, fishing, and other aquatic sports
Ecosystem services	• Was shown to provide refugee habitat to native dragonflies in Japan • Introduced as species with ornamental value • Compounds usable in natural cosmetics • Bioenergy
Management options	• Limitation of dissolved inorganic carbon in lakes can reduce growth of *Elodea* species • Harvesting *Elodea* in spring before plant regeneration and fragmentation take place can reduce occurrence, but spread of fragments is possible • Introduction of native herbivorous fish and waterfowl • Wait until *Elodea* populations collapse themselves

species exist, but all of them can have side-effects (e.g., fragments of *Elodea* can be spread when plants are cut and resprout later) [13]. For some widespread species (e.g., the North American raccoon in Germany) management options focus on preventing damages to human properties, preventing predation in and on endangered habitats and species, and prevention of health-related problems.

In general, it is especially hard to restrict the further spread of those invasive species that are of significant economic interest. An example is the Pacific oyster (Table 34.3), which is cultivated in countries around the world, and thus has high propagule pressure, one consequence of which is a high likelihood of further spread of feral populations.

Table 34.3 A marine invasive mollusk: *Crassostrea gigas* Thunberg, Pacific oyster, its non-native range in Europe and examples of its disservices and services. Cf. [11, 17] (Figs. 34.7 and 34.8)

***Crassostrea gigas* Thunberg, Pacific oyster** Photograph: by Sonty567 at nl.wikipedia	
Native to: North-West Pacific **Introduced to**: Atlantic, Black Sea, Indian Ocean, Mediterranean, North Sea, non-native parts within the Pacific (red lines in Fig. 34.8) Map data: [17]	
Ecosystem disservices	• Restricts the provision of habitat to native species, including species of high conservation value, by occupying these species' native habitat • Distributes oyster pest and other diseases • Oysters contaminated by microbiota threaten human health
Ecosystem services	• Builds oyster reefs, thus providing habitat • Food; main commercially used mollusc species in Europe • Coastal protection • Climate regulation
Management options	• No effective management options known (as the species is of strong economic interest, a reduction of propagule pressure is unlikely)

Table 34.4 A terrestrial invasive mammal: *Nyctereutes procyonoides* Gray, its native range, non-native range and examples of its disservices and services. Cf. [17] (Figs. 34.9 and 34.10)

***Nyctereutes procyonoides* Gray, Racoon dog** Photograph: by Jukka A. Lång, CC BY 3.0	
Native to: Asia (blue area in Fig. 34.10) **Introduced to**: Europe (red dots in Fig. 34.10) Map data: [17] and IUCN (www.iucnredlist.org)	
Ecosystem disservices	• Predator of birds and amphibians • Vector of rabies, fox tapeworm, and trichinellosis
Ecosystem services	• Hunted for their fur • Popular species in Japanese culture, pictured in arts
Management options	• Reduce the availability of food (e.g., no pet feeding places or compost piles) • Hunting is not effective because the species responds with an increase in litter size

Fig. 34.3 A terrestrial invasive plant species: *Ailanthus altissima* (Mill.) Swingle, Tree of Heaven. © Stefan Klotz, UFZ

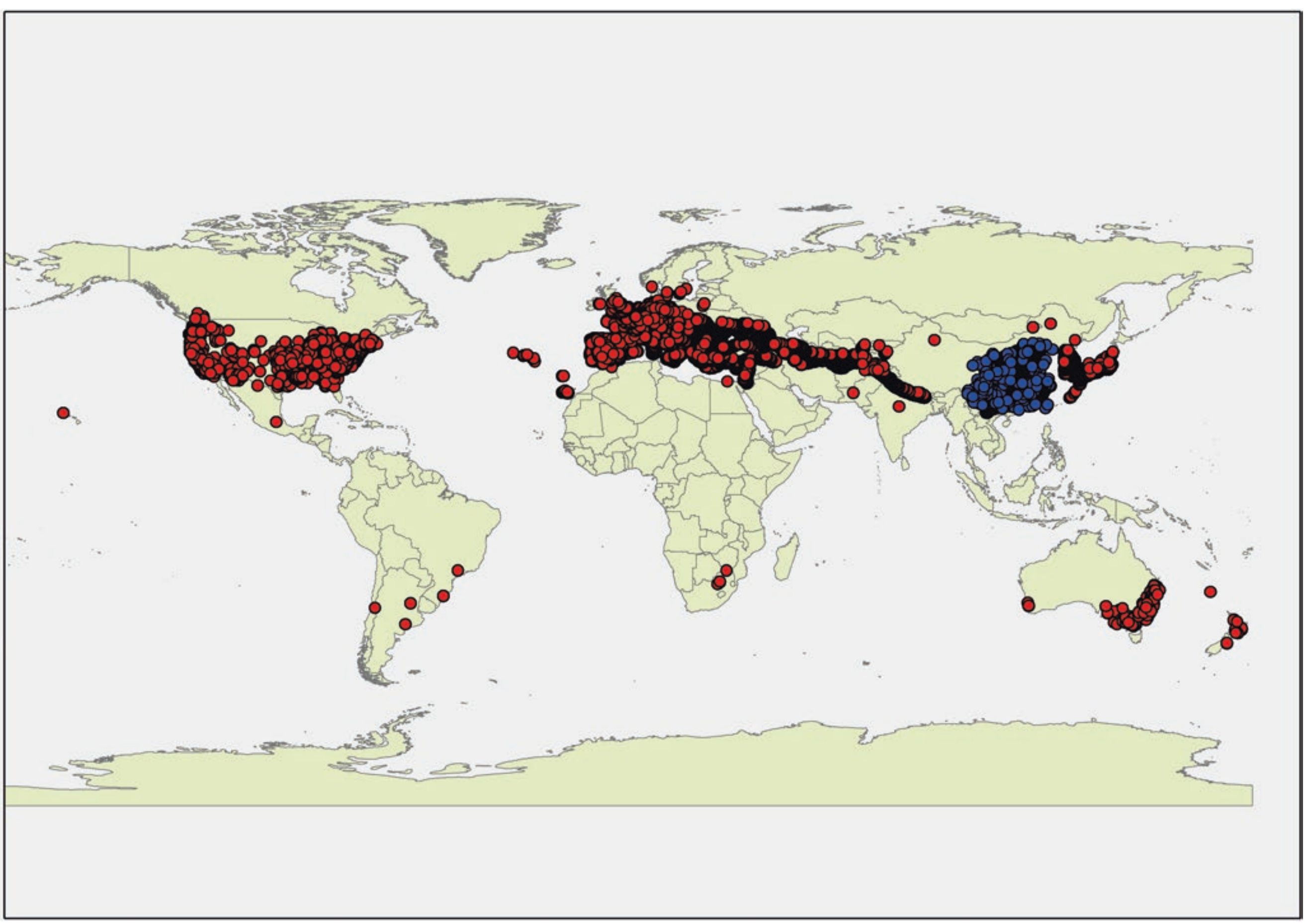

Fig. 34.4 Native (blue dots) and non-native range (red dots) of Tree of Heaven. Map data: © AG Chorologie, Institut für Biologie - Geobotanik & Botanischer Garten, MLU Halle-Wittenberg

Fig. 34.5 An aquatic invasive plant species: *Elodea nuttallii* (Planch.) H. St. John, Nuttal's waterweed. © André Künzelmann, UFZ

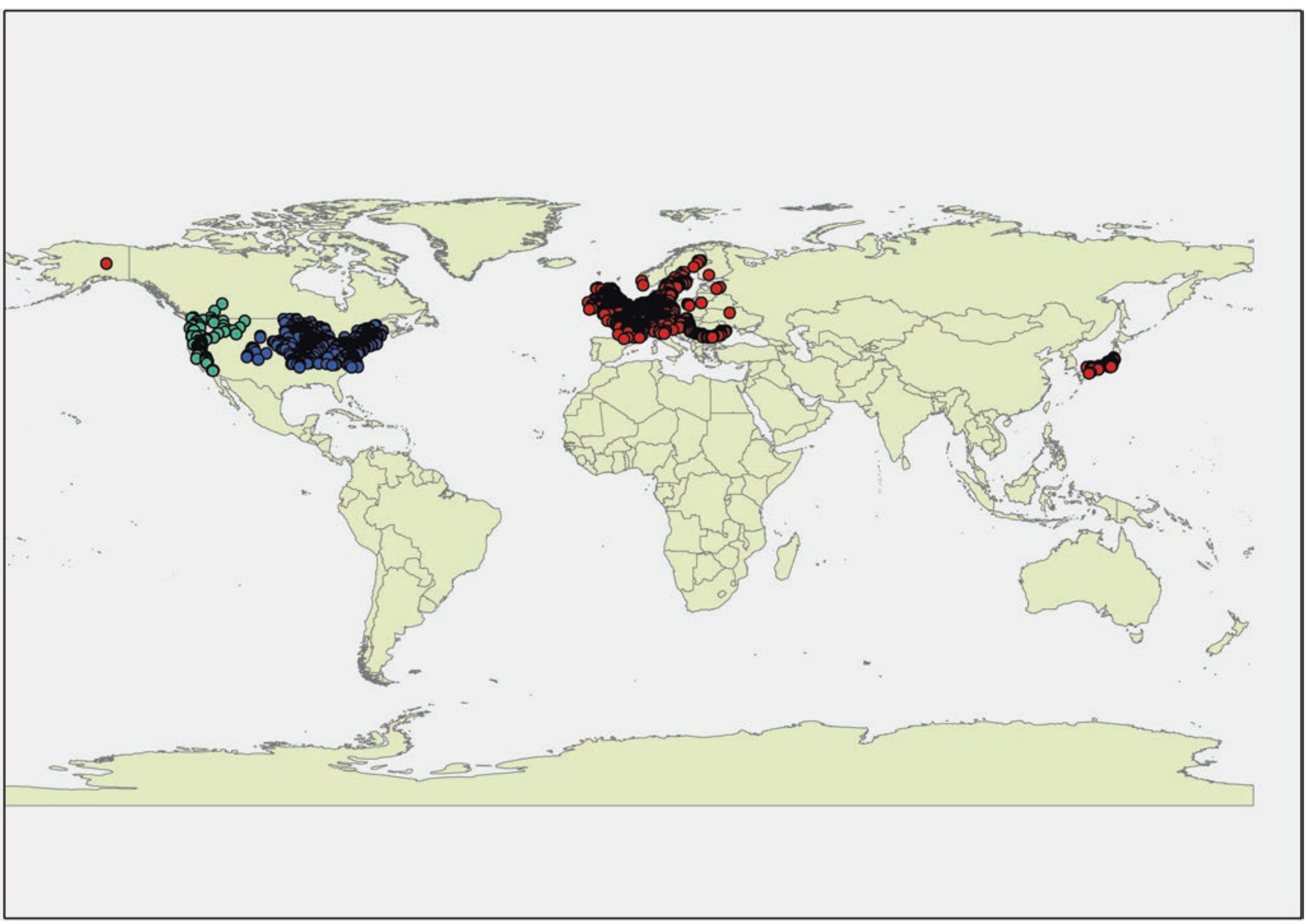

Fig. 34.6 Native (dark blue dots) and non-native range (red dots and light blue dots; the latter being located in North-America but outside of the known native range) of Nuttal's waterweed. Map data: © AG Chorologie, Institut für Biologie - Geobotanik & Botanischer Garten, MLU Halle-Wittenberg

Fig. 34.7 A marine invasive mollusk: Crassostrea gigas Thunberg, Pacific oyster. © Sonty567 at nl.wikipedia

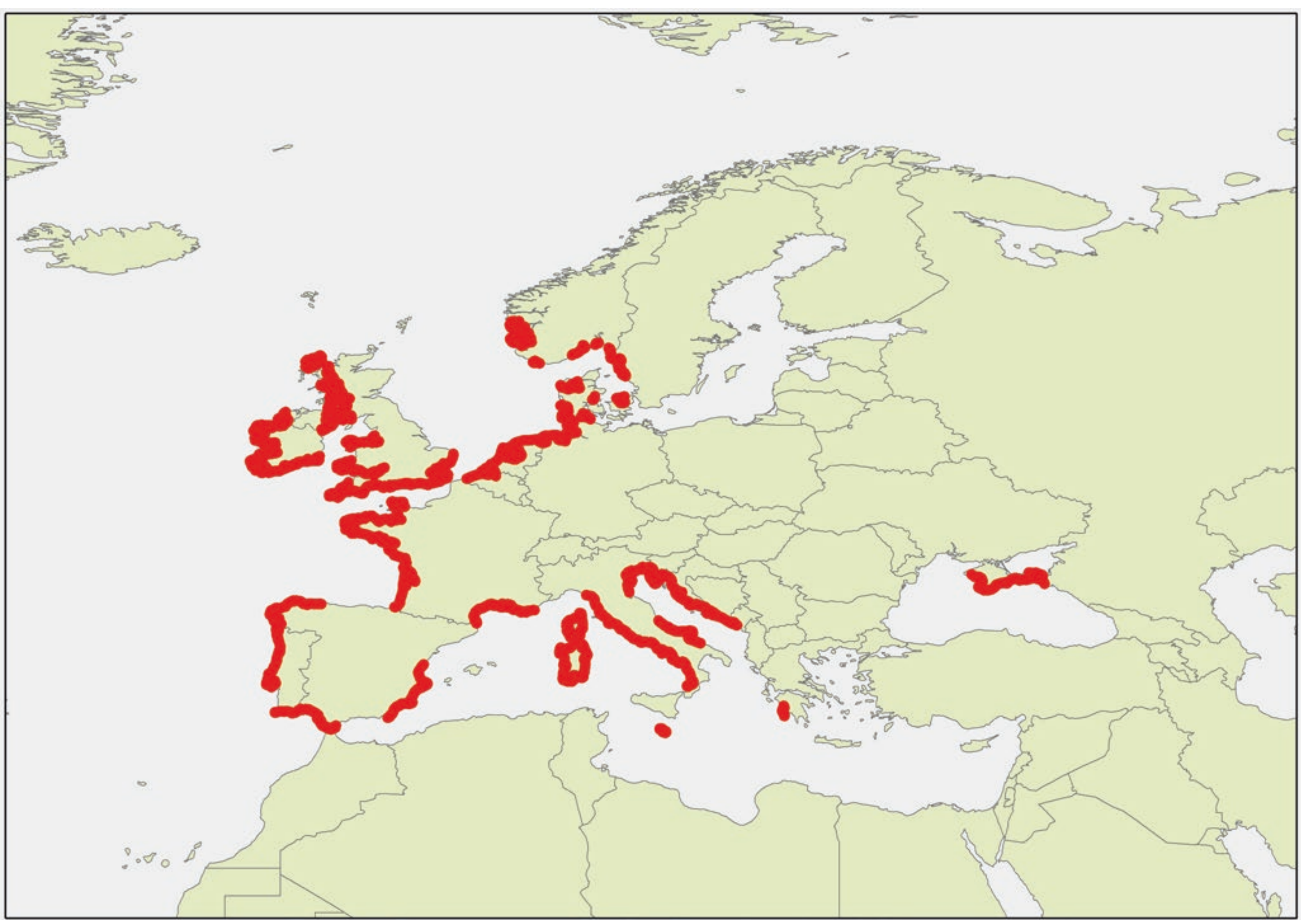

Fig. 34.8 European non-native range (red lines) of Pacific oyster. Map data: DAISIE-project (http://www.europe-aliens.org/)

Fig. 34.9 A terrestrial invasive mammal: Nyctereutes procyonoides Gray, Racoon dog. © Jukka A. Lång

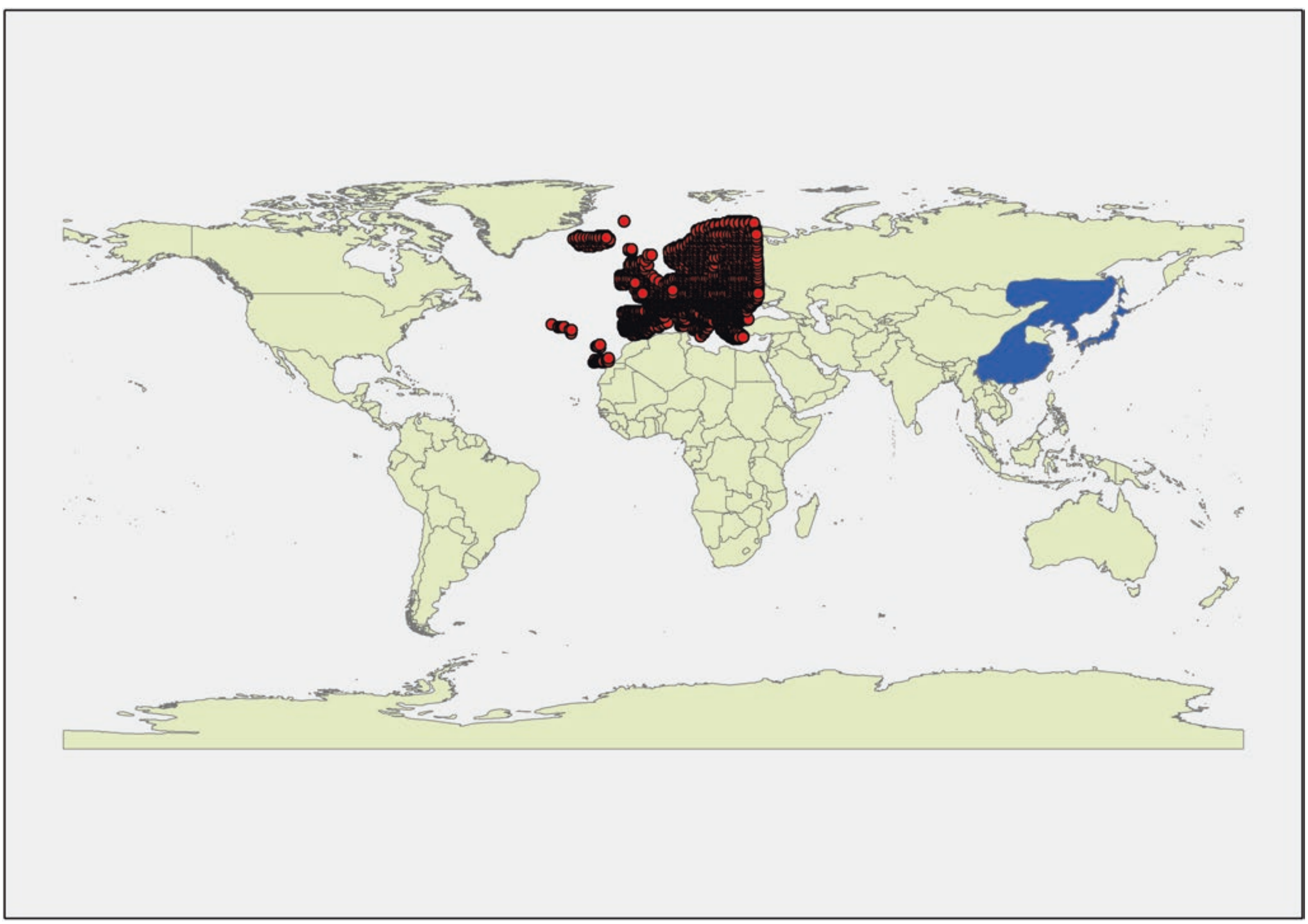

Fig. 34.10 Native (blue area) and non-native range (red dots) of Racoon dog. Map data: DAISIE-project (http://www.europe-aliens.org/) and IUCN (www.iucnredlist.org)

34.4 Conclusions

All in all, research on biological invasions by now has hardly investigated the trade-offs between the services and disservices provided by non-native species. Rather, much focus has been put on their disservices. To get a more balanced picture and to generate improved recommendations for management, case-by-case approaches are required to assess risks vs. opportunities related to the spread of alien species [14]. Historically, those necessary risk assessments often failed or were not applied rigorously enough to evaluate all levels of consequences and the balance between services and disservices. Additionally, to better understand which species' characteristics control ecosystem functions and related ecosystem services (cf. Knapp, in this volume), one would need to distinguish between those characteristics providing services and those promoting invasions or providing disservices.

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35 Alien Planktonic Species in the Marine Realm: What Do They Mean for Ecosystem Services Provision?

Alexandra Kraberg and Gesche Krause

35.1 Introduction

Biodiversity, which in its simplest form is defined as the number of species in a system and their abundance, is considered vital for ecosystem stability [1]. However, rather than using this very simple definition and considering compositional changes to assess ecosystem impacts, it is increasingly considered important to also assess the functions a given species or species trait holds in the system, i.e., functional diversity [2]. Central hereby is whether these vital functions can still be performed if new species are introduced to, or invade the respective system, possibly at the expense of resident species. The different functions species hold in pelagic and coastal ecosystems are highly diverse. These include, among others, primary production functions (which is also considered a supporting ecosystem service), parasitism (important, but understudied), nutrient re-mineralization [3], sediment aeration, and so on.

Species turnover is inevitable even in the most pristine system. However, as evidenced in some time series, such as the Helgoland Roads data set [4], climate change can lead to pronounced increases in temperature and can also affect salinity. Thus, the invasion or introduction of alien species into a new marine system might be considered a proliferative problem as well. This might affect general ecosystem stability and its resilience to further change. This might be expected to be a particularly pronounced problem in coastal seas, which are not only affected by climate change directly but also by other anthropogenic pressures that are increasing because of the growing human populations in coastal areas (Fig. 35.1). The possible mechanisms are manifold, but are often related to functional changes in biodiversity. Ultimately, introductions may also affect changes in the abundance and distribution of commercially important species [6].

Which ecosystem services are addressed? Phytoplankton and zooplankton ecosystem stability under climate change, provisioning and regulating fisheries and aquaculture, multi-trophic marine biodiversity for regulating coastal and marine systems, recreational use of coastal and marine waters.

What is the research question addressed? In what ways does plankton affect marine ecosystem services? What are the central cause-and-effect relationships of invasive species for respective marine ecosystem services?

Which method has been applied? Literature review.

What is the main result? Not all introduced or invading species have a long-term negative (or indeed any) effect, and their potential negative or positive trajectories of invasion into new areas are difficult to assess.

What is concluded, recommended? Reliable, consistent, and internationally comparable long-term data are needed to assess ecosystem changes before effective management decisions can be made. Potential harmful effects should be monitored and investigated in experimental laboratory studies before management action is taken. Similar time series should also be established for socio-economic consequences of climate change, including the long-term effects of non-native species.

35.1.1 Non-native Planktonic Species: The Basics

According to Katsanevakis et al. [7], 1369 species in European waters are to be considered alien (not necessarily invasive). Of these, 382 have been reported from

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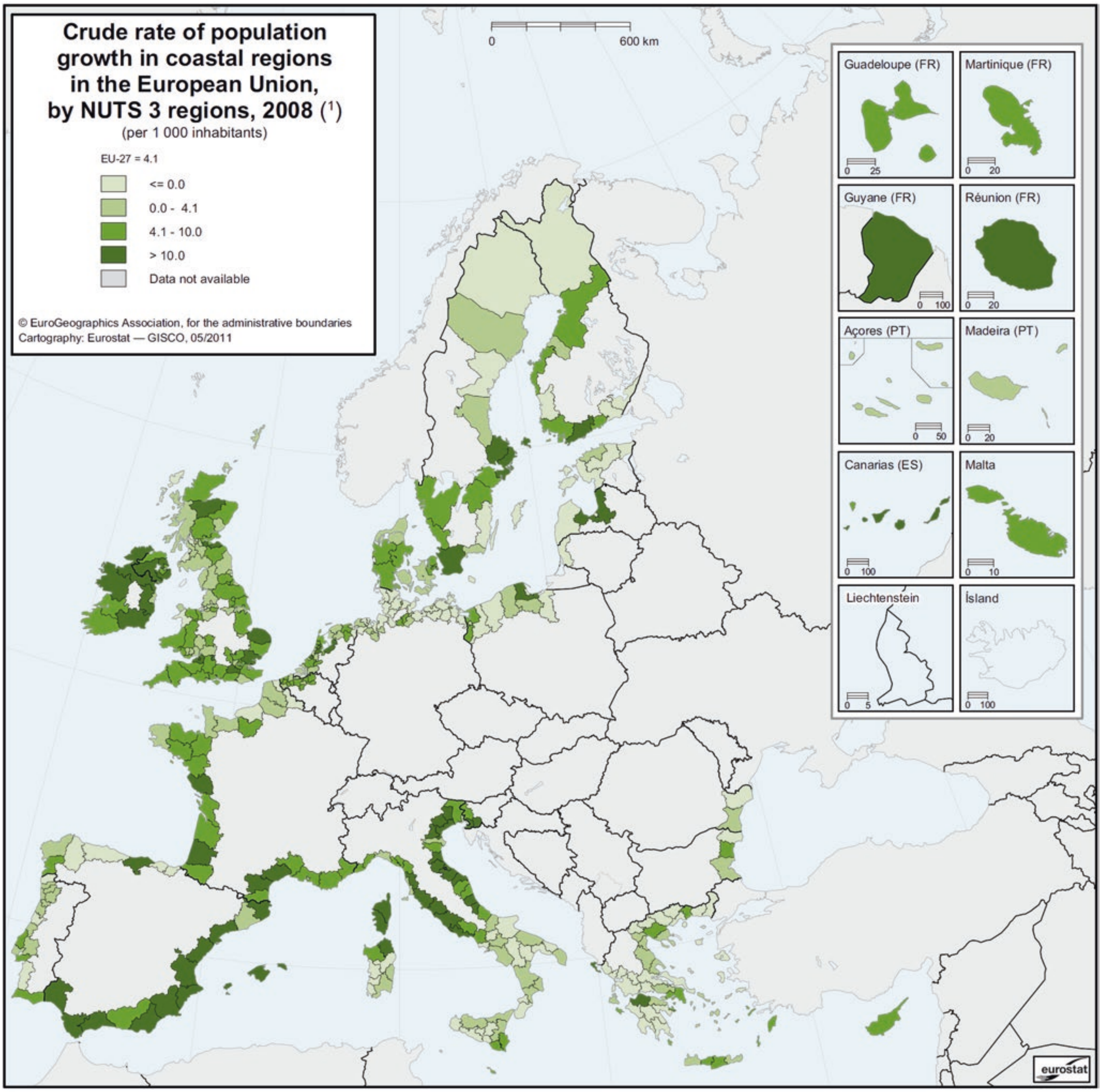


(1) Belgium, Germany, Illes Balears and Canarias (Spain), France and United Kingdom, 2007.

Fig. 35.1 Crude rate of population growth in different NUTS 3 regions in coastal areas (within 50 km from the coast) in the European Union (per 1000 inhabitants), 2007/2008 [5]

outside their native range but do not seem to have established in the recipient system. These numbers exceed by far the numbers in the DAISIE portal, which collected data up to 2006 [8]. In the North Sea, an earlier study found 129 non-native species [9]. However, only a small proportion of these belongs to pelagic groups in the phytoplankton (e.g., *Coscinodiscus wailesii* and *Odontella sinensis*) and zooplankton (e.g., *Mnemiopsis leidyi*), while the majority of non-native species appear to be benthic [9]. Some examples of plankton species, both eukaryotic and prokaryotic, that are considered to be invasive are seen in Table 35.1. It should be noted that while they are considered invasive, native ranges from which they have radiated are not always well defined.

Table 35.1 Traits and impacts associated with different planktonic species considered non-native/invasive in European waters[a]

Species group *(examples)*	Native range	Function/*attributes*	Affected service	Ecosystem service category
Diatoms		**Primary producer**		
Pseudo-nitzschia spp	Several	Bloom formation/ toxicity	Toxic events affect water quality, and can cause economic losses (e.g., contamination of cultured mussels and clams) [5]	Provisioning
Coscinodiscus wailesii	North Pacific	Bloom formation	Fishing operations, mucus production causes clogging of fishing nets [10]	Provisioning
Odontella sinensis	China	Bloom formation	None (not a recent introduction, probably introduced to North Atlantic in the 1900s and now integrated well into local phytoplankton communities)	NA
Dinoflagellates		**Primary producer/ consumer**		
Alexandrium minutum	Unclear	Bloom formation, toxicity	Water purification (e.g., de-oxygenation)	Regulating
Noctiluca scintillans	Unclear	Bloom formation	Water purification (excess ammonium production), food production (fish kills), nutrient dynamics	Regulating, provisioning, supporting
Prorocentrum minimum	Unclear	Bloom formation toxicity	Toxic events affect water quality, and can cause economic losses (e.g., contamination of cultured mussels and clams, pH increases have been reported [11]	Provisioning, regulating
Zooplankton		**Primary and secondary consumers**		
Copepods *Centropages, Acartia tonsa*			Foodweb modification (e.g., outcometing other copepod species	Provisioning
Ctenophores		**Primary/secondary consumers**		
Mnemiopsis leidyi	US East Coast	*Bloom formation*	Food production (competition with fish larvae leading to reduced commercial fish catches) [12]	Provisioning
Bacteria				
Vibrio cholerae, Vibrio vulnificus		Bacterial pathogen	Disease control	Regulating

[a]Service definitions follow the Millennium ecosystem assessment [9, 13]

35.1.2 Ecosystem and Ecosystem Service Risks Associated with Non-native Planktonic Species

From an anthropocentric perspective, the ecosystem service concept provides a framework to link natural capital to human uses of nature [14]. By acknowledging the role of ecosystems as providers of essential goods and services, it links ecosystem functions with livelihoods and well-being [13, 15]. However, as stated in the Millennium Ecosystem Assessment, ecosystem services have been defined as "the benefits provided by ecosystems." This definition has been subject to some debate, because the sheer existence of a certain good does not necessarily result in any benefits. For example, the service of providing habitat and nursery areas for fish exists independently of whether someone is catching the fish or not [16]. Fisher et al. [17] offered an alternative definition in which they understand ecosystem services as "aspects of ecosystems utilized to produce human well-being." It is important to mention that this utilization can either be active or passive. Following this definition, ecosystem services include ecological processes and functions as well as the structure of ecosystems. Thus, ecosystem services are ecological in nature, but their existence as "service" depends on human beneficiaries [16]. Indeed, while scientists have an important place in designing a framework for the management of specific ecosystem services, it is not their responsibility to make final decisions about regulatory policies. A recurring bottleneck in the establishment of an operational framework is the need to define "unacceptable" impacts. While natural science has an important role in advising managers and policy makers on the ecological consequences related to available management options, the setting of impact limits needs to incorporate societal values, needs, and economic realities.

This complexity thus may act as an explanation as to why policy makers have just started to include the ecosystem service concept in their guidelines and programs, for example as part of the Convention on Biological Diversity targets for 2020 [18], and the EU Biodiversity Strategy to 2020 [19]. Especially the importance of ecosystem services from coastal and marine systems are increasingly highlighted [20]. However, the inclusion of the issues of invasive species into decision-making processes has hardly occurred to date. As shown in the selection of groups shown in Table 35.1, different non-native species can have very diverse impacts on the recipient ecosystem and its associated ecosystem services. This lies at the root of why it is so difficult to address these variations within an ecosystem service approach.

Indeed, assessing the actual risk to the service is not a trivial task, as the dynamics of alien species in a system are not easy to predict and might also vary over time. This requires a careful selection of indicators, which need to measure the impacts on the respective ecosystem service across multiple trophic levels as well as the wider risks to society at large. Indeed, it is precisely these perceived risks that can be expected to invoke a governmental response intended to alter the way in which an ecosystem is regulated and managed.

These risks are often described by threshold values. A *threshold* is a general term of value that can be determined by administrative and advisory scientific processes. Thus, in identifying a threshold, it is important to be clear about whether the threshold is determined by policy decisions or by de facto changes in ecosystems [21].

To date, ecosystem managers increasingly use a monitoring endpoint, known as thresholds of potential concern (TPC), to decide when management intervention is needed. TPCs are sets of operational goals along a continuum of change in selected environmental indicators [22, 23]. TPCs are being continually adjusted in response to the emergence of new ecological information or changing management goals. They distinguish normal "background" variability from important changes to, or the risk of degradation of, a given ecosystem service; in this way they provide a conceptual tool that enables ecosystem managers to apply variability concepts in their management plans [23].

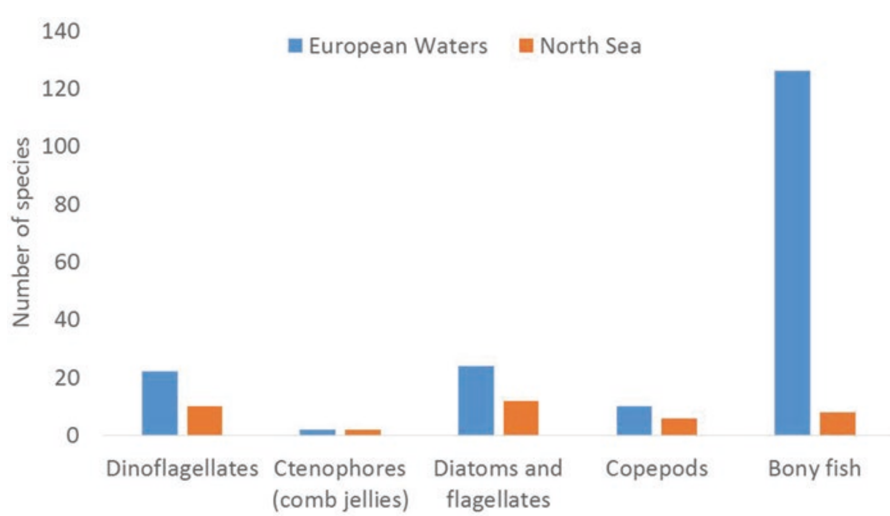
In the case of phytoplankton and zooplankton species, two risks to ecosystem services in particular (and associated additional problems) can be identified.

1. **Blooms and toxic events**: Phytoplankton and microzooplankton can impact a system in two ways: (1) Some species can form extensive blooms in excess of 10^6 cells per litre. These can cause discoloration of the water. When these species die off, the water can undergo deoxygenation and produce an unpleasant odor. An example is the dinoflagellate *Noctiluca scintillans*. Its blooms can cause spectacular bioluminescence, but at the end of the bloom they cause de-oxygenation. This is exacerbated by the fact that cells of this species are known to accumulate high concentrations of ammonium, which is released during cell decay [24]. Some species also produce compounds, that are oderous per se (e.g. in some cyanobacteria) and not necessarily related to mass mortality. Frequent mass blooms can outcompete resident species. Depending on whether they are more or less edible than the native taxa in the system, they can disrupt matter fluxes in the wider marine ecosystem. They therefore represent a risk to ecosystem stability and relevant services (Figs. 35.2 and 35.3). (2) Other species produce potent toxins. While many of these species can also be capable of bloom formation, a bloom event is not always necessary for toxicity to become apparent. Members of the dinoflagellate genus *Dinophysis* for instance often occur in very low numbers, but if cells are taken up by filter feeders such as mussels they can accumulate in their tissues [26].

   These above-mentioned effects can have severely negative repercussions with regard to recreational uses of coastal and marine waters. They also affect aquaculture operations. Mussel aquaculture, for instance, is strongly impacted by toxic events because of the potential of mussels and other bivalves filter to accumulate toxins in their tissues. Closures due to high toxic dinoflagellate or toxin concentrations therefore present a clear risk to, in particular, coastal provisioning and regulating services, the latter resulting from their impact on water quality [27].
2. **Food web interactions**: In the case of invasive zooplankton, there are several examples of species that have outcompeted resident feeding on similar food items. Two of these are the cladoceran *Penilia avirostris* [28] and the ctenophore *Mnemiopsis leidyi* [29]. These may cause severe disruption in the entire trophic food web of a marine ecosystem. *Mnemiopsis leidyi*, for instance, competed for food with and preyed on fish larval species, including commercial stocks in the Black Sea, thereby contributing to the decline of local stocks. This was probably only possible, however, because the local fish stocks (the anchovy *Engraulis encrasicolus*) were already overexploited when the ctenophore entered the picture [30]. It has also been argued that the increasing eutrophication in many areas of Black Sea and the accompanying increase in primary production were the true causes of the mass development of gelatinous zooplankton in the first place. It is also noteworthy that in the case of *M. leidyi*, a second invading species of the genus *Beroe*, and not human management actions alone, eventually controlled *M. leidyi*. The example of *M. leidyi* shows that the proliferation of a non-native plankton species is not necessarily driven by a single factor or individual trait; instead, several factors may interact to determine the impact of a species. Having said this, one impact might also affect several services, and to develop good management practices with respect to a new species in an ecosystem it is therefore vital to identify the correct cause-and-effect relationships and to develop intensive co-operation between all relevant stakeholders involved in environmental management (or impacting the environment), as otherwise management actions to protect one service, such as a commercial fishery, might harm another service [31].

Fig. 35.2 Examples of pelagic species in the North Sea classified as invasive/non-native according to the DAISIE online portal [8] and Gollasch et al. [9]. The species are grouped according to their function in the marine food web (including planktonic and non-planktonic species). Photos: *C. wailesii*, *T. punctigera*: by Alexandra Kraberg, *P. avirostris*: by Otto Larink, *M. leidyi*: by Henrike Hamer, from http://planktonnet.awi.de, CC BY 3.0; *A. minutum*: by Alexandra Kraberg, © AWI; *B. ovate*: by APhillips6651 (https://combjellies.wikispaces.com/file/detail/beroe_ovata.jpg), CC BY-SA 3.0; *N. melanostomus*: by Peter van der Sluijs (https://commons.wikimedia.org/wiki/File:A_large_neogobius_melanostomus.jpg#file), CC BY-SA 3.0. Drawing adapted from [25], CC0

Fig. 35.3 Numbers of invasive species (both true invasives and introduced pelagic species) as provided for European waters in the DAISIE portal [8] and for the North Sea in Gollasch et al. [9]



35.1.3 Long-Term Risks of Non-native Species: Manage Them or Leave Them Alone?

Not all introduced or invading species have a long-term negative effect, and may have no effect whatsoever. In the case of the plankton, for instance, there are examples of species that were introduced into the North Sea more than 100 years ago, such as the diatom *Odontella sinensis*. These appear to have had no noticeable effect on the ecosystem or the services it provides. Thus, any changes have probably been dwarfed by the overall environmental fluctuations, as well as by management and regulatory measures, all of which additionally triggered species composition changes over the years. Likewise, a very recent arrival into the North Sea, the diatom *Mediopyxis helysia*, was touted as a new invasive species, as it formed extensive blooms in many North Sea locations [32]. These blooms, however, seem to have abated, and the species has turned into an additional food item for grazers such as copepod. Possibly the most well-known example of a non-native marine phytoplankton species in the North Sea is the diatom *Coscinodiscus wailesii*. This species was introduced into the North Atlantic accidentally in the 1970s and caused severe disruption by clogging the nets of commercial fishing operators [33]. However, this species is now a common component in the North Sea, and autumn and winter plankton, copepod grazers, and parasites all contribute to controlling its numbers [12]. There are, on the other hand, species such as *Prorocentrum minimum* in the Baltic, whose first appearance and subsequent proliferation are well documented, and harmful blooms of which have been a consistent feature of the receiving ecosystem with demonstrated consequences for the local flora, e.g., the displacement of species [10].

The occurrences and abundance of phytoplankton are known to be very patchy. In many cases they are also difficult to identify, making their true native ranges and the trajectories of their invasion into new areas difficult to assess. Reliable, consistent, and internationally comparable long-term data are clearly needed to assess ecosystem changes, whether they are caused by anthropogenic climate change or other factors, before any effective management decisions can be made. Especially where toxicity or adverse effects such as de-oxygenation cannot be demonstrated (e.g., *Mediopyxis helysia*), a cautionary management approach is needed, with careful monitoring of potential harmful effects and their investigation in experimental laboratory studies before taking action. However, similar time series should also be established for socio-economic consequences of climate change, including the long-term effects of non-native species. Ultimately, the problem of how to assess and ultimately deal with the problem of non-native species is not just an issue of hard scientific facts, but is a very emotive issue in the eyes of the public, and different research communities also view and assess the risks very differently [34].

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36 Invasion of the Wadden Sea by the Pacific Oyster (*Magallana gigas*): A Risk to Ecosystem Services?

Lars Gutow and Christian Buschbaum

Which ecosystem services are addressed? Biodiversity, food provision, coastal protection, seawater quality.

What is the research question addressed? Does the introduction of an invasive species affect ecosystem services?

Which method has been applied? Literature review.

What is the main result? The introduced Pacific oyster does not affect ecosystem services of the Wadden Sea. However, the consequences of species invasions are generally unpredictable.

What is concluded, recommended? Careful observations of non-indigenous species and introduction pathways. Development of measures to reduce the risk of species invasions.

Species invasions are a major reason for global changes in biodiversity [1]. Invasions can induce changes in the structure of native populations and communities [2]. A loss of key species may lead to direct or indirect effects on ecosystem functions [3].

The Pacific oyster, *Magallana gigas*, is native to the coastal waters of the western North Pacific. The species was introduced to the North Sea in the 1980s for aquaculture purposes. Soon after the introduction, *Magallana gigas* escaped from the culture plots into the natural environment and spread along the entire Wadden Sea coast [4]. The planktonic oyster larvae preferentially colonized the shells of the native blue mussel, *Mytilus edulis* (Fig. 36.1a), which provided extensive hard substratum for settlement in an ecosystem, which is otherwise dominated by unstable sediments. This induced a transition of large intertidal areas, which were formerly dominated by beds of the blue mussel (Fig. 36.1b), into extensive oyster reefs (Fig. 36.1c). Accordingly, it was feared that the invasive oyster could drive the native blue mussel population to extinction.

Mytilus edulis is a key species of the Wadden Sea ecosystem, which is essential for various ecosystem functions. The structurally complex intertidal and subtidal mussel beds provide habitat and foraging ground for highly diverse associated species communities, thereby substantially enhancing biodiversity [5]. Mussel beds enhance sediment stability, reduce wave action, and can thus, provide natural coastal protection [6]. Humans are extracting mussels from natural populations either for food or as seed mussels to sustain commercial cultures. Mussels are suspension feeders. By filtrating large volumes of seawater, they contribute to the recycling of suspended organic matter, improve seawater quality, reduce eutrophication, and suppress phytoplankton blooms [7, 8]. Consequently, extinction of the blue mussel population in the Wadden Sea is likely to affect various important ecosystem services.

About three decades after the introduction of the invasive oyster, *Mytilus edulis* is still maintaining vital populations in the Wadden Sea. After the first intensive colonization of mussel shells during the initial phase of oyster establishment, the larger, grown-up oysters became more attractive as settlement substratum for the oyster larvae than the smaller mussels. Today, oysters and mussels coexist in mixed beds in the Wadden Sea, where both species show a specific within-bed distribution pattern (Fig. 36.2). The large oysters extend far above the sediment and dominate the uppermost sections of the reef-like aggregates, whereas the mussels mainly occur in deeper sections between the oysters. Today, mussels occur in abundances similar to the time before the oyster invasion. The body size of individual mussels has decreased, indicating intense competition with the oysters for food. However, in the deeper layers of the aggregates, the mussels

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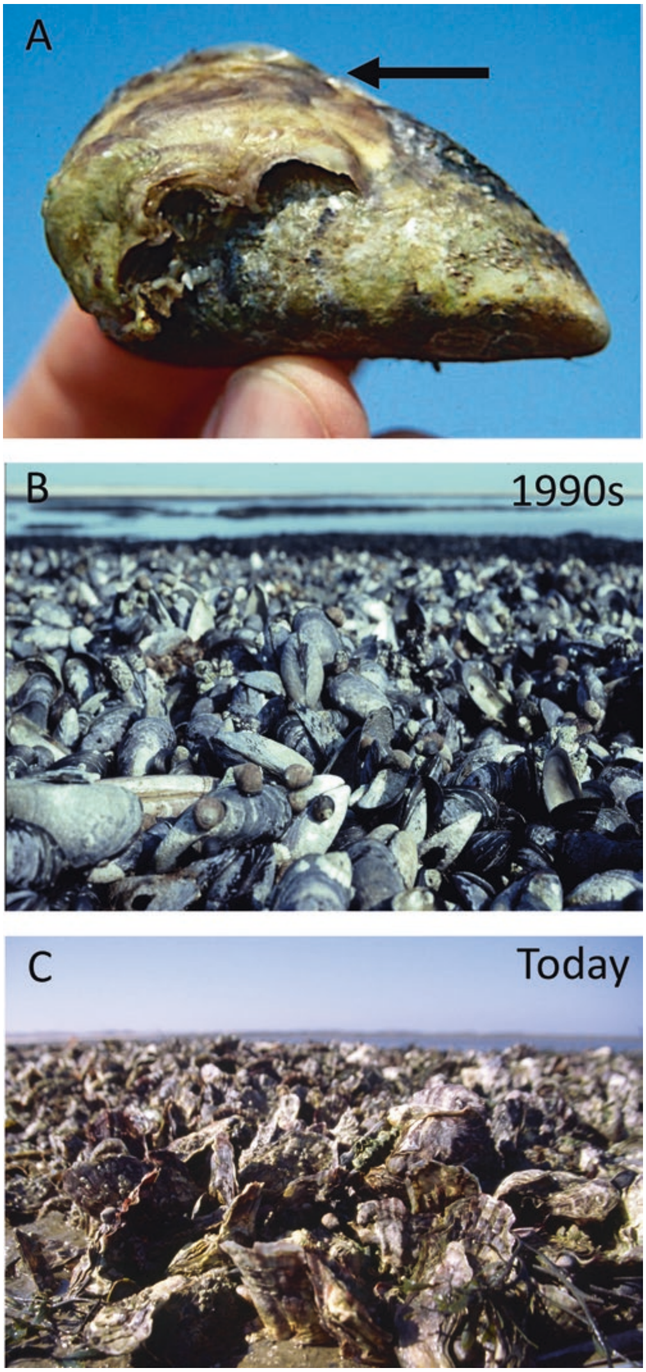

Fig. 36.1 Transition from native blue mussel beds into mixed reefs of mussels and Pacific oysters in the Wadden Sea. (**a**) Shortly after the introduction of *Magallana gigas,* oyster larvae preferentially settled on the shells of blue mussels (arrowhead). (**b**) Until the 1990s, blue mussels formed dense beds on tidal flats. (**c**) Today, most mussel beds in the Wadden Sea have turned into mixed aggregations, which are visually dominated by Pacific oysters

are better protected from predatory birds and crabs as well as from detrimental overgrowth by barnacles [9].

The Pacific oyster is one of the most conspicuous invaders of the North Sea coast. The vast spatial extent of the oyster beds and the intensive use of food resources and competition with native species, such as the blue mussel, have definitely altered the structure and the functioning of the Wadden Sea ecosystem. However, important ecosystem services apparently remained largely unaffected by this invasion, although explicit quantitative comparisons between the pre- and post-invasion period are lacking for some services. The associated species assemblages of the former mussel beds and of the mixed aggregates of oysters and mussels show similar diversity. The overall species inventory remained identical, although the dominance pattern of the associated fauna has changed [10]. Still, the bivalves provide important resources in terms of food and habitat. Oysters are similarly or even more efficient than mussels with regard to sediment stabilization and water clearance [6, 11], which is the reason that efforts are currently under way to re-establish native oysters in various regions worldwide where historical over-exploitation has led to their extinction. Mussel fishery and farming in the Wadden Sea is largely restricted to specific areas and to subtidal mussel beds and is thus not affected by the mostly intertidal distribution of the Pacific oyster.

The list of detrimental effects of species invasions on ecosystem services is extensive and provides examples from all types of environments including marine, freshwater, and terrestrial ecosystems [12]. However, not all species invasions negatively affect ecosystem services and, apparently, it is almost impossible to predict which invader is likely to have negative effects and which ecosystems are likely to be negatively affected by invasions. The species inventory of the Wadden Sea is currently experiencing dramatic changes due to the cumulative effects of global warming and the transport of species by transoceanic shipping, with *Magallana gigas* being a prominent example of how rising seawater temperatures may favor the establishment of foreign species [13]. So far, however, none of the invaders of the Wadden Sea seems to have severe negative effects on the functioning of the local ecosystem. Even efficient ecosystem engineers, such as the Pacific oyster *Magallana gigas*, which have the potential to sculpt the coastal environment, has been smoothly incorporated into the ecosystem of the Wadden Sea. Apparently, comparatively young ecosystems, such as the North Sea, which developed only 8000 years ago after the last ice age, have a capacity to integrate non-native species without profound implications for ecosystem services. However, this may not generally be applicable to other ecosystems and invaders, which makes the consequences of future species invasions unpredictable. Therefore, non-indigenous species, as well as potential immigration routes, should be carefully observed and appropriate measures—such as ballast water treatments and improvement of aquaculture practices—should be taken to reduce the risk of species introductions.

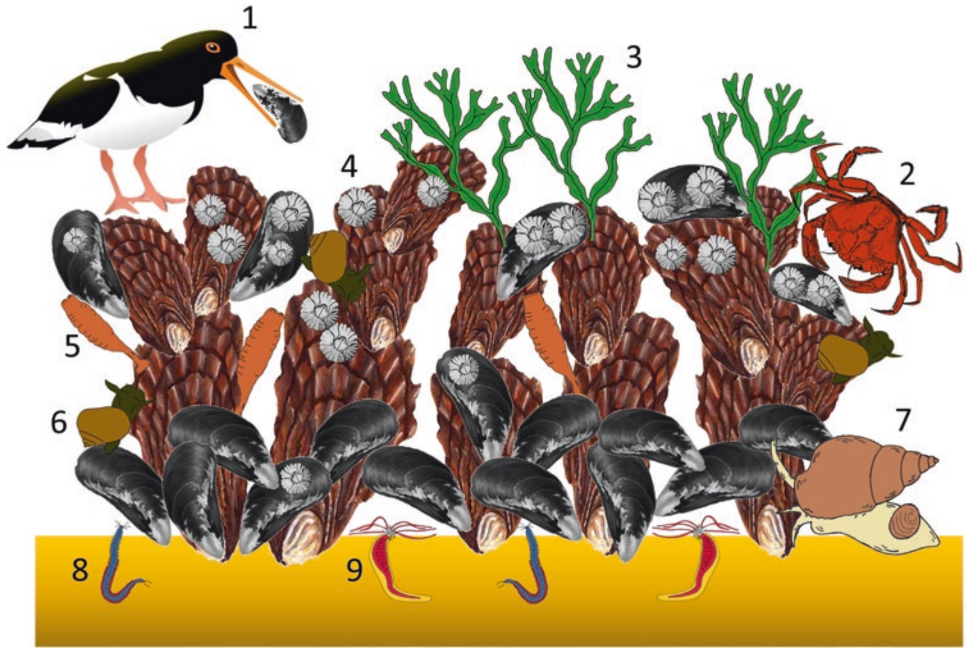

Fig. 36.2 Schematic illustration of a common oyster reef structure and its ecological functions in the Wadden Sea. The smaller native blue mussels mainly occur near the bottom between the large oysters where they experience less predation and barnacle overgrowth. An oyster reef provides food for predatory birds (**1**) and crabs (**2**). Associated sessile species such as seaweeds (**3**), barnacles (**4**), tunicates (**5**), and mobile species such as periwinkles (**6**) and whelks (**7**) use the reefs as habitat. The bottom of a reef is inhabited by endobenthic species such as polychaete worms (**8** and **9**). The reefs represent diversity hotspots and an integral component of the trophic network of the Wadden Sea ecosystem

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37 International Trade and Global Flows of Ecosystem Services

Thomas Koellner, Nikolaus McLachlan, and Sebastian Arnhold

37.1 Trade Theories and Mapping Trade Flows in the Nineteenth Century

Trade of biomass as food, fodder, and fibres has long had a place in human history, and was theoretically underpinned by David Ricardo's book *The Principles of Trade and Taxation* in 1817. His theory of comparative advantage explains that nations mutually benefit from trade, if the trading partners differ with respect to factor endowments, e.g., land or labour. When a nation is able to produce more than is consumed domestically, it is sensible to trade with other nations with lower product-specific endowments. This is even true if in absolute terms the trading partner is better off with endowments to produce all products. Specifically, the nations' differences in provisioning services have led to a global market of commodities from fisheries, forestry, and agriculture. This is because the provisioning services of biogeographical regions to supply such goods show an unequal distribution as a result of favourable climate conditions, soil qualities, rainfall, or biodiversity, but also technologies. France, for example, excels traditionally in the production of wine. In 1864, Minard [15] painted a map of France's export of this nationally important produce from agroecosystems (Fig. 37.1). We can describe this sketch as the first map of global flows of an ecosystem service because this map already depicts three features of today's graphical representation of global flows of trade goods, virtual water, and ecosystem services, which are 1) biophysical realism (width of the flows represent hectolitres); 2) spatial mapping of the flows; and 3) time series information between 1830 and 1864 (as seen in the upper right corner of Fig. 37.1).

Which ecosystem services are addressed? Erosion regulation.

What is the research question addressed? Does import of biomass lead to negative impacts on ecosystem services in the exporting regions?

Which method has been applied? InVEST globally parametrized in order to assess erosion regulation lost due to agricultural commodity production.

What is the main result? An increase of EU soybean imports due to set aside area would lead to net export of erosion.

What is concluded, recommended? The results show high spatial variability of erosion regulation in exporting countries, which needs to be considered when imports' impacts on ES should be minimized.

37.2 Assessing Global Flows of Ecosystem Services

In the modern globalized world, interregional dependencies such as those explained in the previous section became even more important. People living in a specific region depend not only on domestic ecosystem goods and services, but increasingly on imported ones from all over the world (see Fig. 37.2). With that, domestic consumption has global impacts on ecosystems services and biodiversity, which requires an extended consumer responsibility (i.e., consumers are at least partly responsible for environmental impacts along the value chain) [1]. Quantification of such interregional impacts was often based on FAOSTAT or UN Comtrade data in terms of ecological footprints, virtual land, and water flows, carbon footprints, and biodiversity threat exports [2–7]. For example, the amount of virtual water embodied in agricultural exports is captured by the Water Footprint indicator [8]. However, such interregional studies do not yet quantify the impacts on multiple ecosystem services showing interregional synergies and trade-offs. Currently, we see studies that focus mainly on mapping, modelling, and valuing a set of ecosystem services for a specific region or nation. The EU member states, for example, are asked to map ecosystem services nationwide

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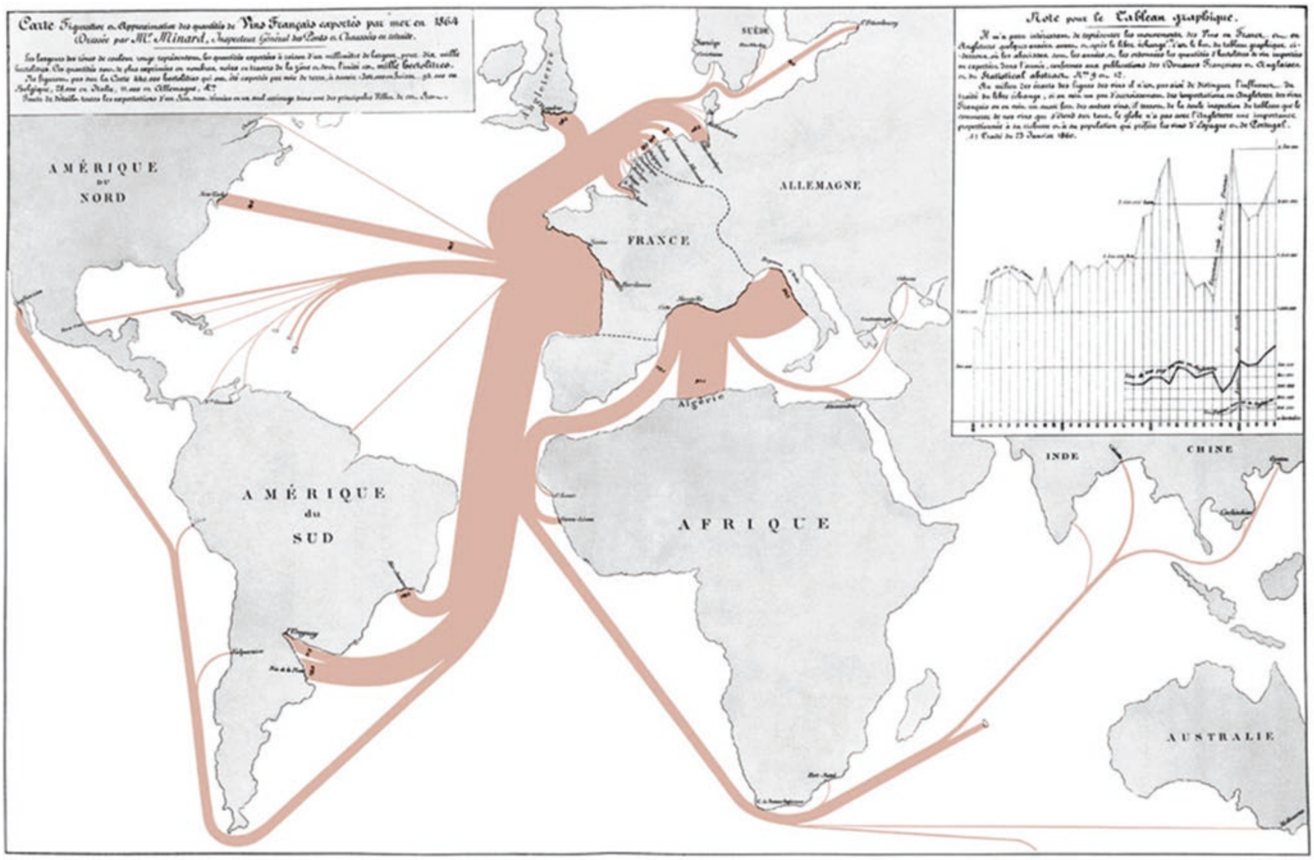

Charles Joseph Minard, *Tableaux Graphiques et Cartes Figuratives de M. Minard*, 1845–1869, a portfolio of his work held by the Bibliothèque de l'École Nationale des Ponts et Chaussées, Paris.

Fig. 37.1 Interregional flows of ecosystem services with EU as an exemplary reporter region [15]

(Target 2, MAES Action 5) in the framework of the Biodiversity Strategy 2020 [9], but this does not require analysis of interregional supply of ecosystem services. The political border is chosen as the system boundary, assuming that the regions are closed [10]. However, in a globalized world, system boundaries are typically open with respect to fluxes of matter, energy, and information. Regional studies wrongly neglect the dependence on "overseas" ecosystem services, the biodiversity and ecosystem service impact of traded commodities, and thus teleconnections between regions [11]. This is also underlined by the EU Biodiversity Strategy (Target 6, Action 17) which requests to "reduce the impacts of EU consumption patterns on [global] biodiversity and make sure that the EU initiative on resource efficiency, our trade negotiations and market signals all reflect this objective" [9].

37.3 European Union's Overseas Flows of Ecosystem Services

Net land imports embodied in commodity trade into European Union EU27 range between 14 and 54 Mha, depending on the type of model applied [5]. The underlying food and feed trade is linked to real nitrogen net imports of roughly 2200 Gigagram Nitrogen per year (GgN/year) into the EU [12], putting pressure on EU nutrient retention services. At the same time, overseas ecosystem services and biodiversity in the production regions are significantly damaged. An example is the loss of erosion regulation and soil degradation. Fig. 37.3 shows the annual EU impact on soil erosion (i.e., the loss of erosion regulation) in the five major EU trading partners, possibly triggered by a 5% increase of today's import of

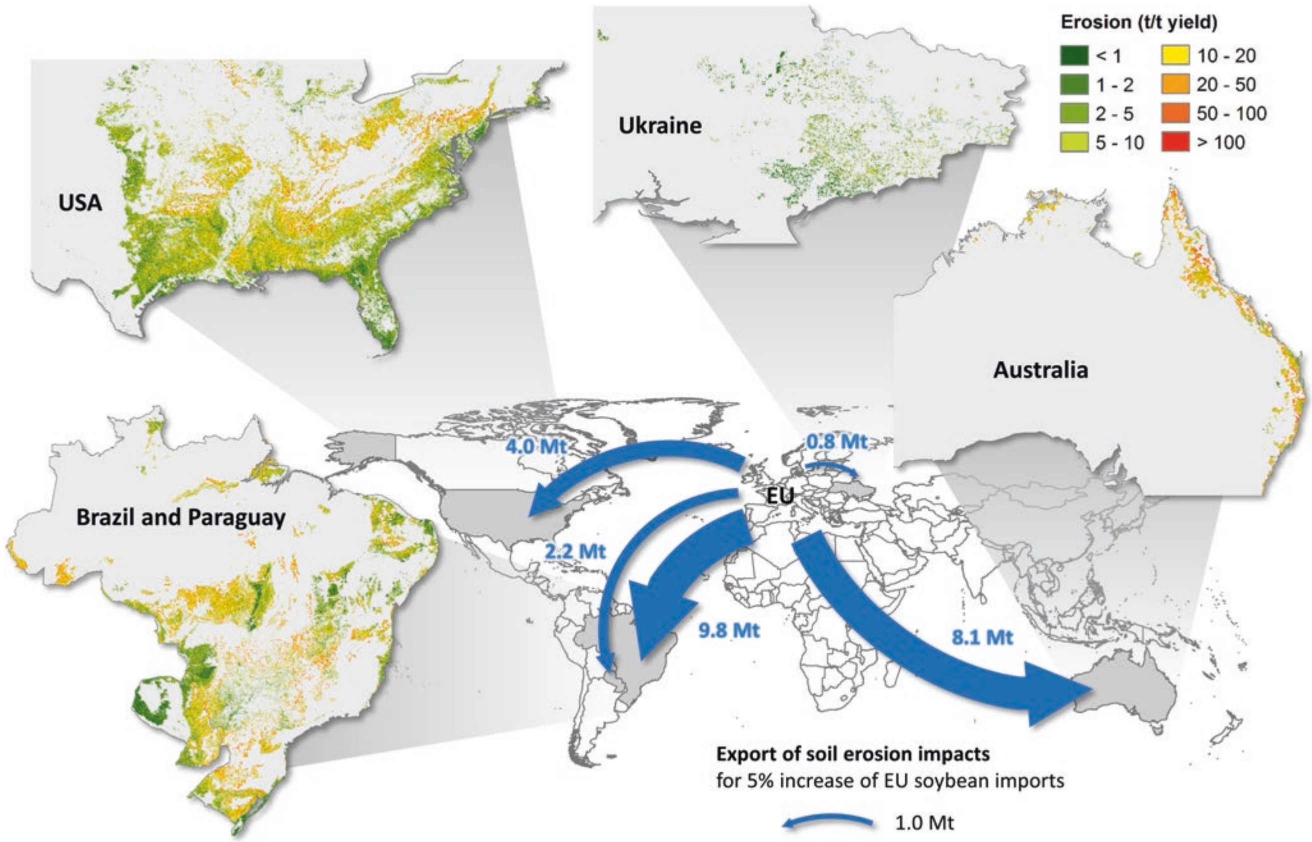

Fig. 37.2 Spatial differences of soil erosion potential for soybean production in tonnes per tonne of yield (raster maps in green to red show t/t yield) and export of impacts on erosion regulation due to EU soybean imports (blue numbers and arrows indicate total amounts of megatons soil eroded per year)

soy products. As seen in the maps coloured from green to yellow to red, the extent of soil loss would be on average most severe in Australia, followed by Brazil and the USA, while imports from Ukraine show more moderate impacts. In analogy to virtual land and water flows [8], such virtual erosion flows refer to the average annual amount of soil loss added through the outsourcing of production. Virtual erosion flows are expressed in total amounts of megatons (Mt) per year (blue arrows and numbers). They are computed based on the additional production area required to compensate potential production losses within the EU, e.g., associated with extensification of EU agriculture via 5% Ecological Focus Areas. From a global perspective, such shifts would be clearly negative if the impact on erosion regulation in the overseas countries were higher compared to the domestic production. Broadly speaking, increasing the trade-offs between biomass production (e.g., soy bean production) and other services (e.g., soil erosion regulation) should be avoided on a global scale and not only in one world region, because in exporting countries this can lead to an increase of ecological risks and vulnerability of the socio-ecological system.

37.4 Towards Interregional Assessments of Ecosystem Services

Ultimately policies aiming at improved ecosystem services in one region should not lead to ecosystem damage and therefore impact on bundles of services elsewhere. Therefore, interregional assessments of ecosystem services' synergies and trade-offs are needed to complement National Ecosystem Assessments (NEAs); the NEA UK was specifically taking first steps towards this goal [13].

Figure 37.2 provides a schematic overview of how to analyse multiple ecosystem services' synergies and trade-offs between two trading countries. Key elements of such an assessment could be 1) goal and scope statements of assessing interregional ES flows; 2) biophysical quantification of inflowing and outflowing multiple ES; 3) their evaluation in terms of dependencies, impacts, and benefits; and 4) an interpretation and recommendation section. Results of mathematical optimization (methods widely used in business and economics to calculate the combination of input factors to

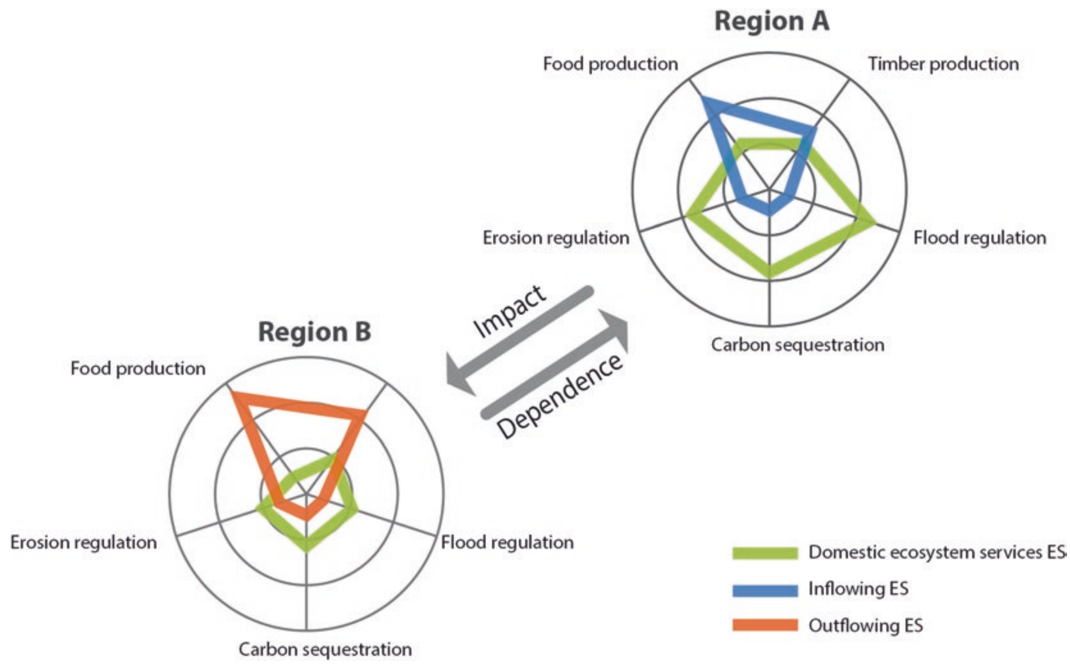

Fig. 37.3 Interregional assessment of ecosystem services' synergies and trade-offs. Region A is depending on food and timber imports (inflowing ecosystem services) from region B (outflowing ecosystem services) and achieves domestically a balanced ecosystem service budget with high levels of erosion regulation, carbon sequestration, and flood regulation. However, region A's imports trigger negative impacts on regulating services in Region B, leading to an unbalanced situation for B. In this example, a global optimization might be achieved by reducing the amount of interregional flows of provisioning services or by reduction of impact of food and timber production on regulating services

maximize or minimize the output) might also be part of such an interregional assessment. An example for this is the spatial optimization of domestic agricultural production versus agricultural imports towards minimizing CO_2 emissions, as it was recently shown for Brazil-Germany in a two-country world [14]. The study showed, under its strong assumptions (e.g., only current cropland used without further deforestation), that CO_2 emissions from transport plus agricultural production are minimized if sugar is produced in Brazil and shipped to Germany compared to using sugar beets in Germany to satisfy the local consumption. The main reason for this outcome is the much higher hectare yield of sugarcane in Brazil. There is also potential *within* countries for optimization due to heterogeneous environmental conditions (see Fig. 37.2), which requests a high spatial resolution beyond assessments on a national scale. Such explorations of the optimization space are an important element in achieving sustainable development in an interregional context and would extend the EU Biodiversity strategy 2020 (Target 6, Action 17), which already requests reducing the impact for biodiversity caused by the EU consumption of imported commodities.

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Part III

Trade-offs and Synergies Among Ecosystem Services

Introduction to Part III: Trade-Offs and Synergies Among Ecosystem Services

38

Anna F. Cord, Nina Schwarz, Ralf Seppelt, Martin Volk, and Matthias Schröter

38.1 Introduction

Efforts are increasing to integrate the sustainable provision of ecosystem services into land management decision-making. These efforts, however, are challenged by (1) the variety of methods to map and quantify ecosystem services, and (2) the scarcity of knowledge on how environmental policies and management decisions affect relationships among ecosystem services. Changes in land management can alter the type of the main services provided (e.g., from regulating to provisioning services in the case of intensification of agricultural management) and the total amount and relative mix of services provided. Unknown relationships among ecosystem services might lead to unintentional effects of management that set the sustained provision of ecosystem services at risk. A better understanding of relationships among ecosystem services is therefore much needed. This chapter introduces the part of the Atlas framework (Fig. 38.1) that focuses on relationships among ecosystem services. It contains a typology and common definitions of different types of relationships (Sect. 38.2), provides a brief overview of the diversity of methods and approaches used (Sect. 38.3), includes a summary of empirical evidence (Sect. 38.4), and, finally, discusses implications for planning and management (Sect. 38.5).

38.2 Typology and Definitions of Ecosystem Service Relationships

The conceptualization of links between ecosystem services—which we term here *relationships*—is characterized by a persistent lack of consensus. The term *trade-off*—the counterpart of *synergy*—has become especially popular in the ecosystem service literature, but still lacks conceptual clarity. Recently, the idea of ecosystem service *bundles* has gained increasing attention as a way of describing ecosystem services co-occurring in space or time. See below for more detailed definitions.

38.2.1 Relevant Mechanisms

Two principle types of mechanisms lead to different forms of relationships among multiple ecosystem services [1]: (1) common drivers such as land use change, fertilization, and expansion of infrastructure; and (2) direct interactions among ecosystem services (Fig. 38.2). While drivers may in some cases affect only a single ecosystem service, they often have positive and/or negative effects on multiple services at once (e.g., fertilization may increase agricultural yield but at the same time decrease water quality). On the contrary, direct uni- or bi-directional interactions among ecosystem services often emerge from the same underlying ecological functions and ecosystem capacity that are relevant to several ecosystem services (e.g., carbon storage and water flow regulation can both be provided by intact forest ecosystems). Neighborhood effects also frequently play an important role in observed relationships (e.g., natural and semi-natural habitats increase numbers of pollinator species, and thus have positive effects on productivity of adjacent coffee plantations). While differentiating between the two mentioned mechanisms is also important when selecting appropriate methods (Sect. 38.3), we also find non-causal co-occurrence of ecosystem services ("no effect relationships") that may happen by chance or as an artifact of ecosystem service mapping techniques [2].

38.2.2 Trade-Offs and Synergies

Common drivers and direct interactions among ecosystem services can both drive service provision in the same or opposite directions, leading to positive (synergies) or negative (trade-offs) relationships [2]. The term trade-off generally describes a situation that involves losing one quality or aspect of something in return for gaining another. Trade-off situations hence require choices to be made between two or more alternatives that cannot be achieved at the same time [3]. Trade-offs and synergies can occur spatially

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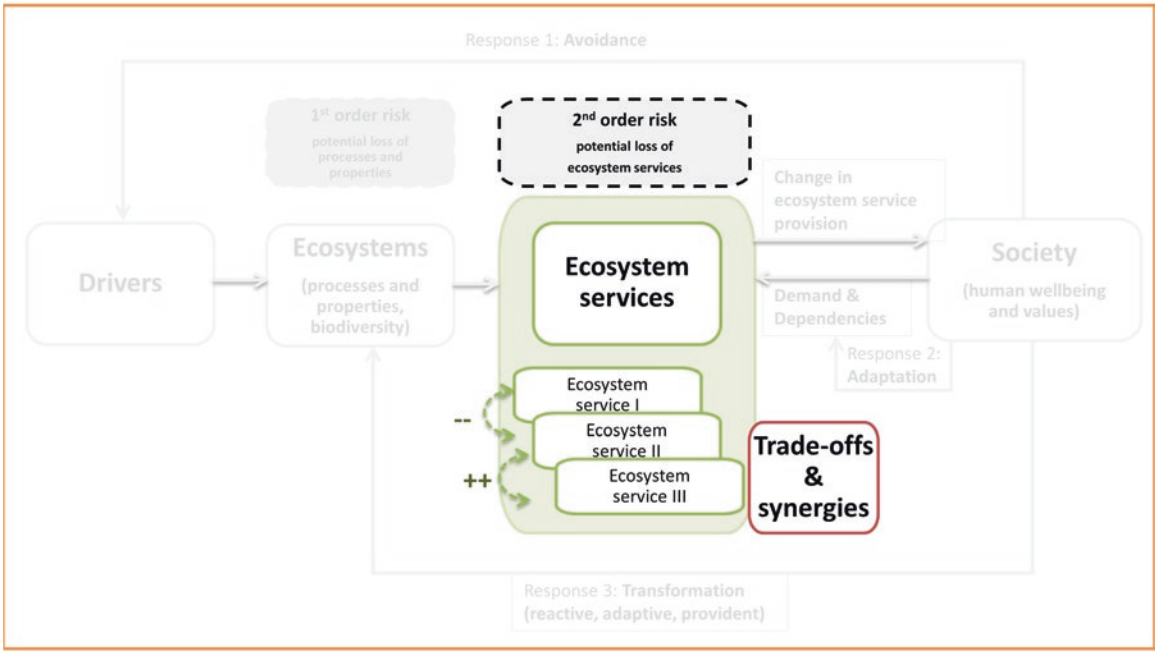

Fig. 38.1 Framework elements of the Atlas of Ecosystem Services addressed in this part

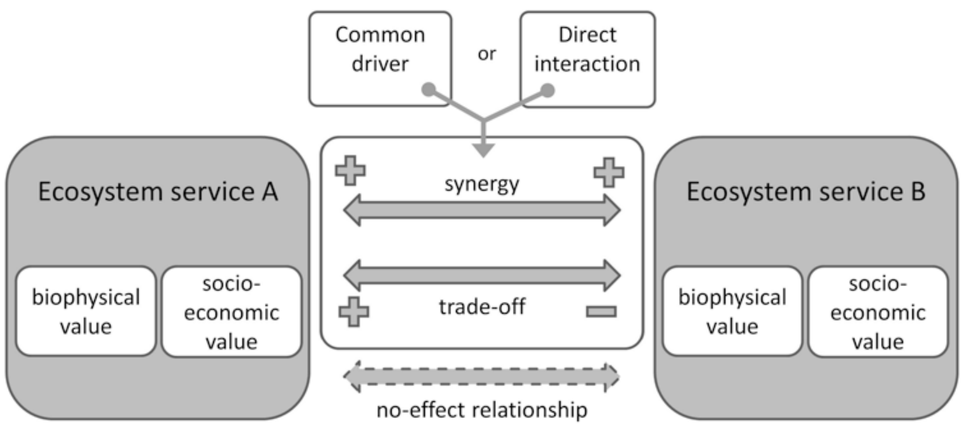

Fig. 38.2 Conceptual overview of possible relationships between ecosystem services

(across locations) or temporally (over time), and ecosystem service perturbations may or may not be reversible [4]. Further, they relate to both the biophysical provision of ecosystem services, as well as to the socio-economic and well-being benefits of different groups of people.

Based on Turkelboom and colleagues [3], we emphasize two major criteria for the occurrence of trade-offs and synergies among ecosystem services:

1. A causal relationship exists (i.e., there is a common driver or direct interaction between services). Hence we ignore non-causal co-occurrences of ecosystem services.
2. Demand for and use of the considered ecosystem services exist. Hence we ignore situations where an ecosystem is not somehow managed, altered, accessed, or experienced.

38.2.3 Ecosystem Service Bundles

Ecosystem service bundles are commonly defined as "sets of services that appear repeatedly together" [5], i.e., they represent patterns of spatially or temporally co-varying types of ecosystem services. As opposed to trade-offs and synergies, ecosystem services occurring within the same bundle are not necessarily causally linked. However, per definition, they are provided at the same time or in the same location/by the same spatial units.

38.3 Methods and Approaches

A range of qualitative and quantitative methods can be used to assess ecosystem service relationships (reviewed in Mouchet et al. [2]). The choice of appropriate method(s) depends on the research objectives [6], specific hypotheses to be tested, and compatibility with data availability and spatio-temporal scale. It may also have effects on the probability of finding trade-offs, synergies, or no-effect relationships [7].

38.3.1 Pairwise Correlations

The most popular quantitative method to assess relationships among continuous ecosystem service indicators are pairwise correlation coefficients. They are used in combination with statistical tests to identify the general direction and strength of ecosystem services relationships (e.g., Raudsepp-Hearne et al. [5]).

38.3.2 Factor Analyses and Clustering Approaches

Factor analyses and clustering approaches represent a better alternative when considering more than two ecosystem services. They identify similar ecosystem services (e.g., ecosystem bundles), are more flexible regarding the formalization of the ecosystem service indicator (i.e., continuous, nominal, or binary), and are able to handle a combination of quantitative and qualitative indicators simultaneously (e.g., Turner et al. and Queiroz et al. [8, 9]).

38.3.3 Regression-Based Methods

Regression-based methods imply causal relationships and emphasize the importance of mechanistic linkages among ecosystem services and of common drivers. Their use hence goes beyond simple detection. Still, they can get at causality only when the methodological framework is set to test for such causal relationships, i.e., by using experimental systems or predictors directly assessing the underlying mechanisms [2].

38.3.4 Multi-objective Optimization

Understanding ecosystem service trade-offs and synergies also plays an important role in studies that aim at exploring the biophysical and socio-economic constraints of landscapes and limitations to their multi-functionality. In this context, a promising approach is the combination of ecosystem service models with multi-criteria optimization methods to simulate a multitude of optimal solutions to land use or management problems. For example, while considering different crop rotations, Lautenbach et al. analyzed the trade-offs between conflicting objectives such as bioenergy production, food and fodder production, and water quantity and water quality [10].

38.3.5 Participatory Methods

Methods from the social sciences can help to understand how stakeholders perceive synergies and trade-offs that result from management decisions. Questionnaires, interviews, workshops, and focus-group discussions can help elucidate trade-offs across value domains, e.g., perceived social importance vs. economic values [11] or instrumental values against other moral values [12].

38.4 Results from Empirical Studies and from the Atlas of Ecosystem Services

The majority of case studies on relationships among ecosystem services focus on agricultural land use systems, particularly in North America and Europe. Such studies explore provisioning ecosystem services much more often than regulating/maintenance or cultural ecosystem services [13]. Typical and often-studied trade-off situations arise between timber production and carbon sequestration, as well as between food production and maintenance of habitats and biodiversity. Further, since research is mostly carried out on plot-to-regional scale, insights regarding potential trade-offs and synergies at larger spatial scales are for the most part missing. A recent global review of pairwise relationships between ecosystem services [7], however, showed that the majority of case studies reported similar relationships for pairs of ecosystem services, independent of the spatial scale considered and the land use system in which they were studied. Whereas the relationship between regulating and provisioning services are dominated by trade-offs, synergistic relationships are commonly observed different regulating services (e.g., flood protection, carbon sequestration, habitat protection) and different cultural services (e.g., spiritual experiences, recreation services). Increases in cultural ecosystem services, however, typically do not significantly influence provisioning services.

In this Atlas, Seppelt et al. (Chap. 39), conceptually synthesize the relationship between agricultural production and biodiversity under changing land composition, configuration, and landscape use intensity. Franko et al. (Chap. 40) show how a trade-off between two ecosystem services from agricultural areas arises in three federal states in Germany. They identify conflict areas in which soil organic matter, important for climate regulation, trades off with biomass for energy production. In the same vein, Majer et al. (Chap. 41) show areas in Germany where the use of straw creates a potential trade-off with the regulation of soil fertility and soil erosion. Haase et al. (Chap. 42) show how synergies and trade-offs among various urban ecosystem services arise under different land use policies.

Bundles of ecosystem services and their relationships with environmental and socio-economic gradients are assessed by Dittrich et al. (Chap. 43) for Germany. De Knegt (Chap. 44) presents bundles of ecosystem services for the Netherlands. This contribution shows the importance of distinguishing capacity to provide ecosystem services and actual use. It also investigates displacement of trade-offs between ecosystem services. Bennett et al. (Chap. 45) study bundles of ecosystem services for the past, present, and future for the Montérégie region in Canada.

Seppelt et al. (Chap. 46), finally, show simultaneous reaching of peaks of extraction of several provisioning ecosystem services on a global level.

38.5 Implications for Planning and Management

There is growing recognition among researches and decision-makers that considering multiple ecosystem services is crucial to inform balanced and sustainable land-use planning decisions [14]. For this purpose, relationships among multiple ecosystem services should be identified and assessed by integrated social-ecological approaches rather than with either social or ecological data alone [1]. So far, however, only a small portion of the literature dealing with ecosystem services in the context of environmental planning and management specifically takes into account synergies and trade-offs [15]. The goal of such studies is typically to find planning or management solutions that minimize conflicts between multiple uses and ecosystem service values. Examples of the evaluation of potential trade-offs and conflicts between multiple ecosystem services given alternative strategies range from managing protected areas [16] to marine spatial planning [17].

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Trade-Offs and Synergies Between Biodiversity Conservation and Productivity in the Context of Increasing Demands on Landscapes

39

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39.1 Introduction

A growing human population coupled with increasing per capita consumption, changing diets, increasing food waste, and ineffective regulation, have led to rising demands on ecosystems for the services they supply [1]. Globally, there have been increases in the amounts of land cleared of natural vegetation, in the intensification of management activities, and in the simplification of landscape structure, for example, through an increase in broad-scale agricultural practices [2, 3]. Areas of high agricultural production, i.e., provisioning ecosystem services, are being increasingly situated in areas of high biodiversity in many regions, especially southern Europe, China, and South America (Fig. 39.1a), and this overlap has grown more pronounced over the last 50 years, most notably in the tropics and subtropics (Fig. 39.1b). The conflicts between biodiversity and the major ecosystem services provided by agricultural production will increase further in the coming decades if, as predicted, tropical and subtropical areas are increasingly converted for agriculture [4]. Suggestions have been made to design agronomic systems shifting from conventional to more closed, regenerative systems, which would reduce energy consumption and emissions [5]. While trade-offs between allocating land to production and biodiversity conservation have resulted in conflict and polarization (e.g. Tscharntke et al. [1]), the scientific understanding of the underlying processes remains limited. These debates have presented an antagonistic set of land-use conditions in which human activities preclude the conservation of biodiversity. Studies that consider land-use gradients have frequently focused on either agricultural production or biodiversity, which limits our knowledge on how to mitigate trade-offs between food production and conservation. There is therefore a need to conceptualize trade-offs between agricultural production and biodiversity conservation, as well as global externalities resulting from the trade in agricultural products in a general, flexible, transferable framework [6].

Which ecosystem services are addressed? Provisioning ecosystem services, agricultural products, supporting ecosystem services (pollination, bio control), biodiversity.

What is the research question addressed? What are possible functional dependencies of biodiversity-production trade-off under changing land composition, configuration, and land use intensity?

Which method has been applied? Connectional and theoretical considerations, review, synthesis.

What is the main result? The framework suggests non-linear relationships caused by the multifaceted impacts of land use (composition, configuration, and intensity).

What is concluded, recommended? We propose solutions for overcoming the apparently dichotomous aims of maximizing either biodiversity conservation or agricultural production and suggest new hypotheses that emerge from our proposed framework.

39.2 Land Use–Production Relationships

Levels of agricultural production depend on a multitude of context-dependent factors including land-use-management practices, land-use history, infrastructure, and access to markets and subsidies, many of which are correlated [3]. Human land use has led to a diversity of land systems worldwide that vary greatly in the amount of land dedicated to agriculture (i.e., landscape composition), the spatial arrangement of natural and agricultural elements in the landscape (i.e., landscape

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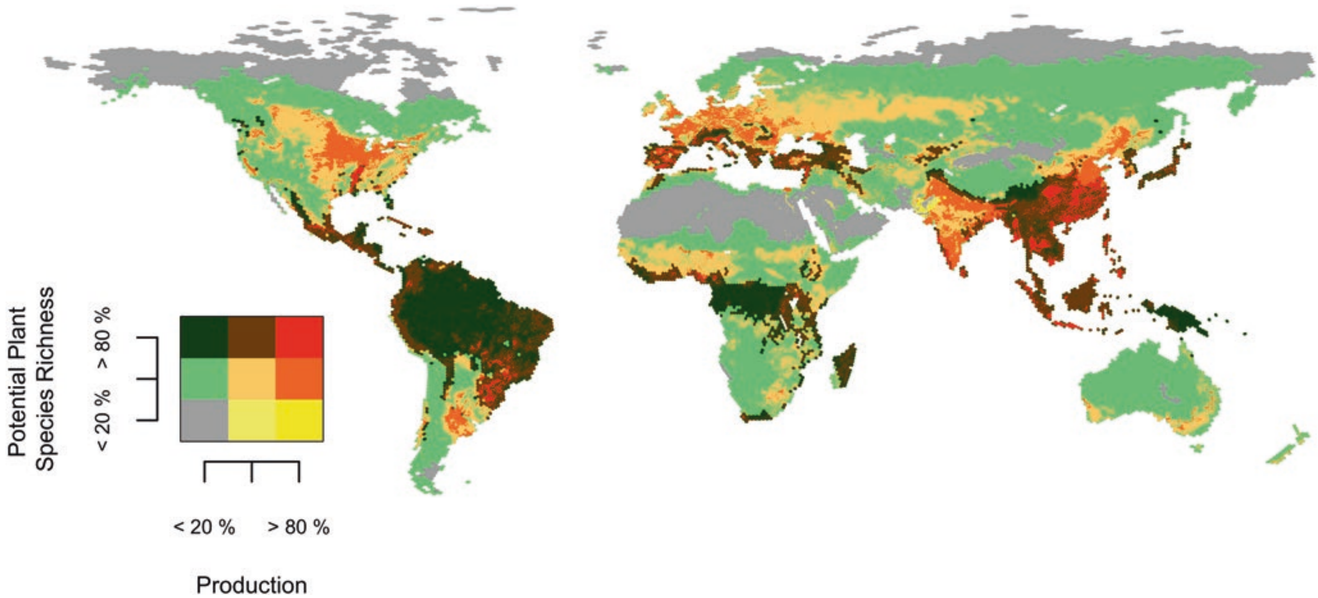

Fig. 39.1 The overlap between agricultural production and plant species richness, based on data on agricultural production and potential species richness of vascular plants. Plant species richness and current crop production were divided into three classes around the 20th and 80th percentiles

configuration), and the kind of management practices applied. The latter is most frequently understood as land-use intensity, characterized by the amount of inputs (chemicals, water, fertilizer, labour) and management aspects (stocking density, tillage regimes).

The most straightforward way to increase production is to increase the proportion of cultivated land. Increased areas of arable land enable a near-linear increase in production (Fig. 39.2a). It also true, however, that once a certain threshold is reached, gains will be reduced by the inclusion of landscape patches that are less suited for agriculture and by the impairment of ecosystem functions in nearby natural habitat. Intensification leads to asymptotically increasing production, with diminishing returns (Fig. 39.2b) owing to limiting factors such as radiation, water availability, and the impairment of important supporting and regulating ecosystem services such as biocontrol or pollination [7]. This pattern of saturation is well known in agricultural economics and is usually referred to as a Cobb-Douglas function [8]. Experimental studies could fully separate the effect of total area from intensity of use, but in real-world landscapes we expect both aspects to interact. The nature of the relationship between production and landscape configuration is less certain (Fig. 39.2c). There might be production benefits to larger farms with more continuous (i.e., less patchy) areas under agriculture, owing to scaling effects or to increased management efficiency [9].

39.3 Land Use–Biodiversity Relationships

Evidence strongly suggests that biodiversity (defined here as the combination of richness and abundance) decreases when the proportion of agricultural land is increased, because this results in the loss and fragmentation of natural habitats (Fig. 39.2d; [10]). The form of this relationship will depend on exactly how landscape composition affects the relative abundances of species [11, but see 12]. Increasing land-use intensity can result in a decelerating decrease in biodiversity (Fig. 39.2e; as shown by, e.g., Gerstner et al. [10]). Small increases in intensity in minimally altered habitat initially lead to large losses of diversity, while further intensification will result in continuing but less dramatic declines (Fig. 39.2e; e.g., Kleijn et al. [13]). The relationship between diversity and landscape configuration, however, is uncertain, and various plausible relationships can be conjectured (Fig. 39.2f). Landscapes of simpler configuration might support a higher diversity of a certain habitat type if the remaining habitats are in larger patches [10], which, however, depend on the surrounding intensity of use and composition. Complex configurations, and a higher proportion of more undisturbed habitats, might support more mobile species. Furthermore, if migration through the agricultural matrix is possible, small-scale extinctions in fragmented landscapes might be reversed through colonization [14].

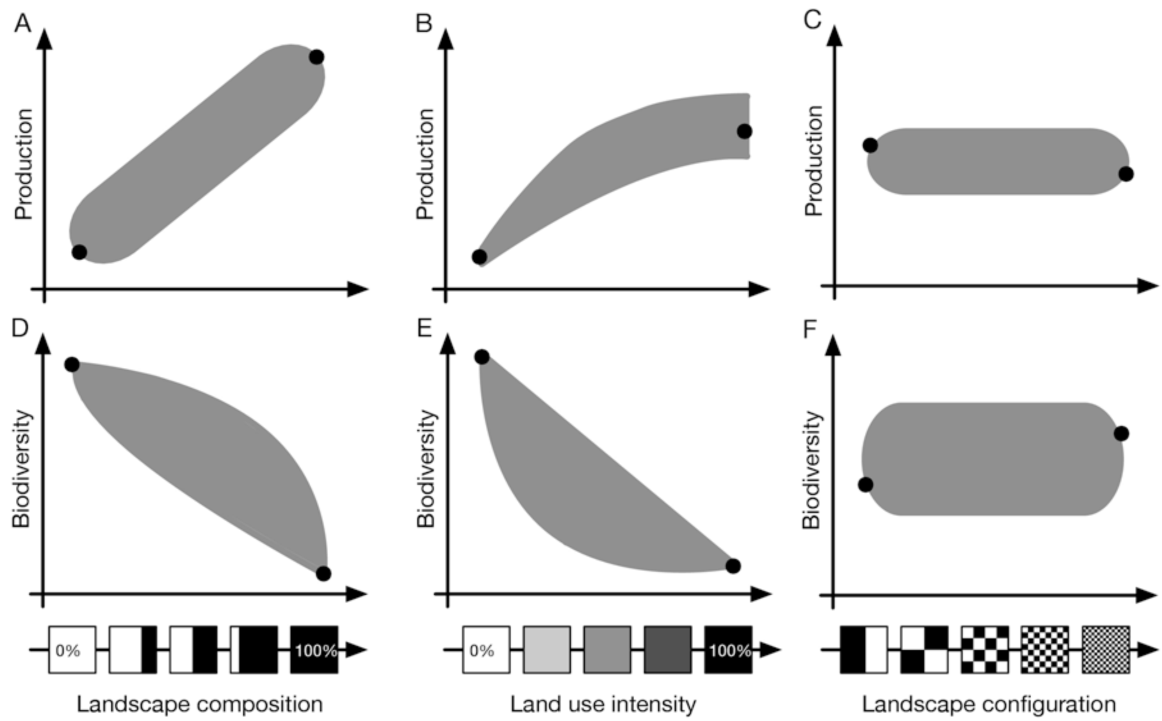

Fig. 39.2 Foundation of the conceptual framework: hypothesized relationships of agricultural production (**a–c**) and biodiversity (measured with abundance-richness metrics; **d–f**) as a function of landscape composition (proportion of agricultural land), land-use intensity, and landscape configuration (reprinted from [6]). Relationships represent a summary of current knowledge as reported in the published literature, with grey shading indicating uncertainty or lack of consensus. Black points illustrate the often-used dichotomous view, comparing just two levels of land use. In the depictions of land use, white colouring indicates areas of natural habitat, and grey or black colouring areas of agriculture (with the intensity of grey indicating land-use intensity)

39.4 Synthesis: Land Use and the Biodiversity–Production Relationship

Figure 39.3a, b show the combined effects of land-use composition, configuration, and intensity on a single axis. The coloured arcs of the smaller upper panels translate directly to the arcs of the same colour in the main panel, and can be associated with different land-use systems. This ranges from best cases, where biodiversity is both maintained within agricultural areas and supports production (upper edge of the grey shaded area in Fig. 39.2c), to worst cases, where agricultural production is at the expense of biodiversity (lower edge of the grey shaded area).

High biodiversity and high agricultural production are possible where biodiversity can provide benefits to agricultural crops, and where agricultural areas are managed to maintain high levels of biodiversity (Fig. 39.3, green arcs). In species that support control of pests, pollination or nutrient cycling contribute to supporting ecosystem services and maintaining higher yields. This requires specific management strategies such as intercropping, agroforestry, and provisioning of nesting habitats (e.g., for pollinators [14]), such as managing complex landscapes that compensate for local high-intensity management by enhancing local biodiversity. This functional relationship could be, e.g., a hump-shaped curve (Fig. 39.3; [15]), although quantitative data along such a complexity gradient are still lacking.

Beyond a certain point, only larger fields, with more efficient production or more energy input and higher land-use intensity, can achieve a further increase of production. Use of chemical inputs is increased, and practices that sterilize, structurally level, and standardize agricultural plots are promoted [1]. The consequences are rapid losses of biodiversity [10] and a comparably slower increase of agricultural yields (Fig. 39.3, blue arcs; [8]).

Where the focus is exclusively on agricultural production, biodiversity is quickly lost. In these cases, increasing production might be less successful if it depends on components of the biodiversity (Fig. 39.2, red arcs). This could lead to a

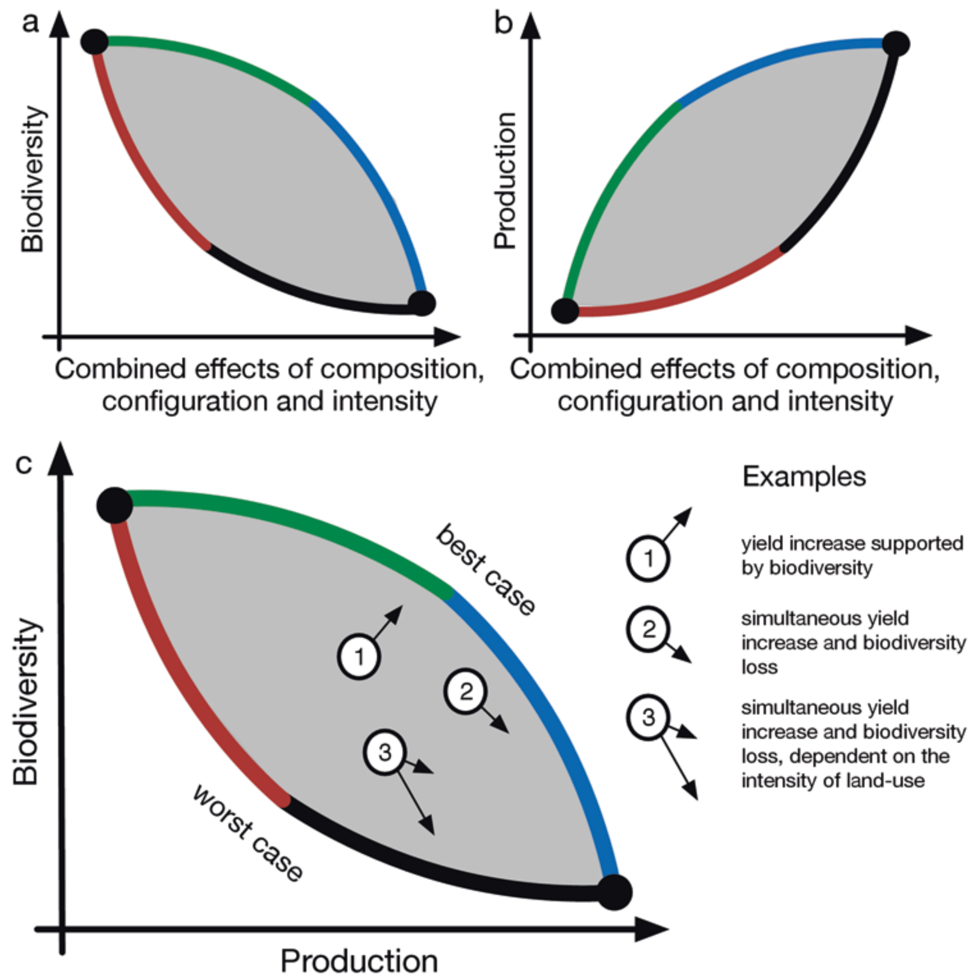

Fig. 39.3 Synthesis of the conceptual framework: Combining the relationships between land use and biodiversity (**a**), and between land use and agricultural production (**b**) leads to hypothesized relationships between agricultural production and biodiversity (**c**) (reprinted from [6]). In the top panels (**a**, **b**) we assume a combined effect of landscape composition, landscape configuration, and land-use intensity, with increased anthropogenic impact to the right. The coloured arcs of the smaller upper panels translate directly to the arcs of the same colour in the main panel and can be associated with different land-use systems. The numbered arrows and corresponding labels in the main panel identify possible options for land management, and correspond to the findings of (1) Finn et al. [17]; (2) Storkey et al. [18]; and (3) Donald et al. [19]

worst-case condition for both biodiversity and production, characterized by antagonistic relationships between wildlife and agricultural production. For example, unsustainable agricultural practices, such as large-scale clearing of vulnerable soils, may cause both large losses of biodiversity and low and declining yields due to soil degradation [16]. On the other hand, there are cases where biodiversity under agricultural production is low, and where agricultural productivity can be achieved only through very high levels of intensification and degradation of the natural area (Fig. 39.2, black arcs). For example, this is the case for highly intense agriculture in the so-called Corn Belt of the US Midwest, with very high soil erosion, depletion of aquifers, water pollution, evolution of herbicide- and pesticide-resistant pests, and so on, leading to a plateauing of agricultural production [3].

39.5 Discussion and Conclusions

The framework helps identify key knowledge gaps and generates hypotheses about trade-offs between agricultural production and biodiversity (Box 39.1). It illustrates how various non-linear relationships in the complex three-dimensional space of land use, biodiversity, and production could be conceptually synthesized into various relationships between production and biodiversity (Fig. 39.3). These relationships encompass the option space for reconciling biodiversity and production. The framework goes beyond the dichotomous views taken in previous discussions, showing that a consideration of gradients in the different facets of land use allows an understanding of the non-linear nature of the relationships.

Box 39.1 Hypotheses Emerging from the Conceptual Framework

Considering the effects of multiple aspects of land use (composition, configuration, intensity) on both agricultural production and biodiversity leads to novel hypotheses about the trade-offs between agricultural production and biodiversity conservation. The following list of hypotheses exemplifies the variety of research questions generated by the conceptual framework and may be extended, especially by considering more landscape contexts and species groups.

1. Landscape **configuration** affects agricultural production less than it does biodiversity. The most pronounced differences in both production and biodiversity are seen in landscapes with intermediate proportions of agricultural land (**composition**).
2. Higher habitat diversity in the landscape (**configuration**) enhances agricultural production, because biodiversity, and thus the ecosystem functions that support production, are supported by a larger number of edge habitats.
3. The higher the habitat diversity in the landscape (**configuration**), the stronger the impact of land-use **intensification** on biodiversity, because of increasing exposure to edge habitats. This will result in land-use intensification being less effective in landscapes with higher habitat diversity because the ecosystem functions supported by biodiversity will decrease more significantly.
4. The larger the fraction of land under agricultural production in the landscape (**composition**), the less effective land-use **intensification** will be for agricultural production (i.e., saturation in Fig. 39.1b appears earlier), because ecosystem functions supported by biodiversity are lacking.
5. Land-use **intensification** can compensate for reduced agricultural productivity caused by lower biodiversity; however, the marginal gain of agricultural production with increasing land-use intensity depends on the crop type(s) and the landscape composition and configuration.
6. Land-use **intensification** negatively affects biodiversity disproportionately more than it increases agricultural production, to different degrees depending on landscape **configuration** and **composition**, and environmental conditions.

(*Reproduced from* Seppelt et al. [6].)

Moving away from a strictly dichotomous view is key to working towards a more complete understanding and more nuanced decision-making. A challenge remains to develop general metrics that combine all aspects of land use (configuration, composition, and intensity), which will allow the application of the proposed framework. It is a high priority for ecologists studying land use–biodiversity relationships to also obtain estimates of agricultural production. We also encourage broadening the set of biodiversity indicators used to include species' abundance information. The framework identifies possible options for reconciling demands for agricultural production with demands for biodiversity conservation. There are multiple unexplored combinations of landscape composition, configuration, and management that might offer the opportunity to manage landscapes optimally to both feed the needs of a growing human population and conserve biodiversity. Conservation of biodiversity needs to be achieved by designing appropriate production systems that contain and benefit from higher biodiversity, rather than focusing only on the protection of pristine habitat.

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40 Climate Change Induced Carbon Competition: Bioenergy Versus Soil Organic Matter Reproduction

Uwe Franko, Felix Witing, and Martin Volk

Abbreviations

AA	Agricultural area within a BPU
BAT	Biologic active time
BPU	Biomass providing unit
C	Carbon
CANDY	Carbon and nitrogen dynamics model
CAP	Capacity index
CCB	Candy carbon balance model
CDI	Carbon demand index
C_{rep}	Carbon reproduction flux
IC	Installed capacity
OM	Organic matter
SOM	Soil organic matter

Which ecosystem services are addressed? Provisioning services (biomass for energy), regulating services (carbon sequestration, climate regulation).

What is the research question addressed? Identify areas with possible conflicts between bioenergy production and reproduction of soil organic matter (SOM).

Which method has been applied? Indicator development, large-scale hot spot analysis.

What is the main result? Due to climate change there is a growing demand (10–40%) of fresh organic carbon from biomass to maintain current levels of soil organic matter reproduction. Within the study region, hot spots of a high carbon demand have been identified, where a high capacity of biogas production may conflict with rising demands for biomass to mitigate climate change effects on soil organic matter storage.

What is concluded, recommended? The developed indicators are widely applicable and transferable to other large-scale studies and help to identify hot spots with a need for adaptation measures. The development of specific mitigation measures in the hot spot areas require a further quantification of the actual carbon demand on a local to farm scale.

40.1 Introduction

Ecosystems regulate the global climate by storing and emitting greenhouse gases. Soils store at least three times as much carbon (C) in soil organic matter (SOM) as can be found in either the atmosphere or in living plants [1]. Besides climate regulation, soil organic matter also positively affects the regulation of other ecosystem services like soil fertility, soil erosion, and soil biodiversity. Depending on the type of land use, soils may be turned into a source or a sink of C. Climate change is expected to lead to an accelerated turnover of SOM in many cases. Additional C sources may be required to sustain the current level of SOM [2], but many C sources that are suitable for SOM reproduction could as well be used for bioenergy production to reduce the trace gas emissions from fossil fuels. The production of bioenergy consumes biomass from crops on arable fields as well as organic matter (OM) from agricultural residuals such as slurry. It can be considered as a provisioning service, but also represents a climate change mitigation strategy. If a bioenergy system successfully contributes to a mitigation of global warming, it consequently reduces the general risk of potential harms to global ecosystems. But whenever C from the agricultural system is used to produce biogas, it is removed from the agricultural matter cycling, thus reducing available C for the reproduction of SOM.

To investigate this potential conflict, we analyzed the expected changes in SOM turnover together with current regional biogas production capacities in Central Germany. An indicator-based assessment scheme is used to classify sub-regions according to their conflict potential between matter demand for SOM reproduction and production of bioenergy. The results presented here are an extract from

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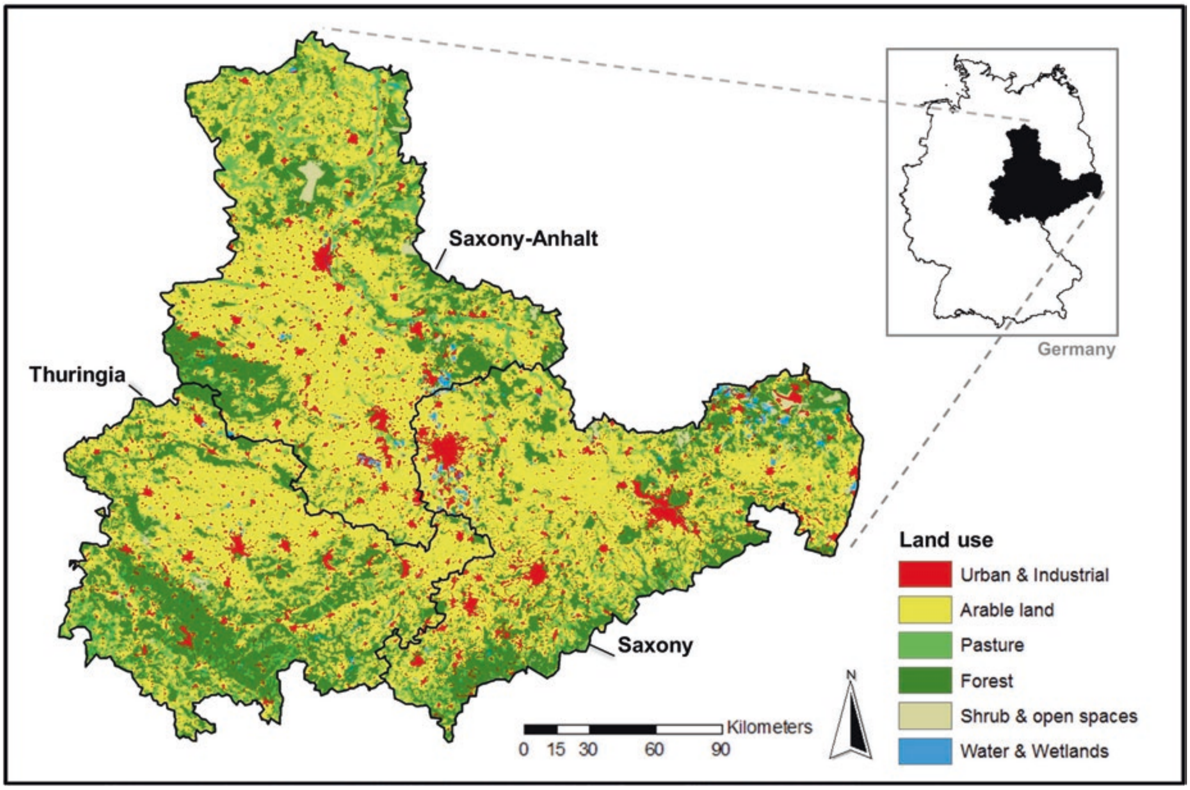

Fig. 40.1 Location and land use map of Central Germany. (*Modified from* Franko et al. [3], *with permission*; land use information is based on CORINE Land Cover data [10, 11])

the study of Franko et al. [3]. The study area included the Federal States Saxony, Saxony-Anhalt, and Thuringia (Central Germany), covering around 55,000 km^2 (Fig. 40.1). Except for the mountainous areas, which reach altitudes of >1100 m, the region is dominated by arable land-use (around 57%). The climate conditions are moderate, with mean annual air temperatures of 6–10 °C in the lowlands, and about 4 °C in the mid-range mountains. Large parts of the lowland region are protected by the mountain ranges and show (sub-)continental climate conditions with an annual precipitation of 300–500 mm. The non-protected landscape has precipitation values of around 700 mm in the lowlands, and increasing values (>1000 mm) in the mountains.

40.2 Soil Organic Matter Turnover Under Climate Change

Turnover of organic matter in soil is controlled by land management and site conditions. Climate change will lead to changes in rainfall and air temperature and will therefore influence the turnover conditions. The Carbon and Nitrogen Dynamics (CANDY) model quantifies the site conditions using the "Biologic Active Time" (BAT in days), which aggregates the time of microbial activity for SOM turnover of a specific location. It allows the comparison of turnover conditions of different environments whereby high BAT values indicate a high potential for biological mineralization of SOM into CO_2. Following the simplified meta-model introduced by Franko and Oelschlägel [4], the annual sum of BAT is calculated depending on rainfall, air temperature, and soil texture.

Biologic Active Time calculations for Central Germany were made for the period 1961–2000 to characterize the current climate conditions and, based on the average climate data over the IPCC scenarios, A1B, A2, and B1 from 2000 to 2100 to represent the future climate. The regional pattern of soil texture is based on the Soil Map of Germany (BÜK1000).

Following the theoretical concept of the Candy Carbon Balance (CCB) model [5], we assume that the soil organic matter level of a soil at steady state is proportional to the carbon reproduction flux (C_{rep}) from supplied fresh organic matter related to the biological activity for SOM turnover expressed by BAT:

$$SOM \sim C_{rep}/BAT$$

This relation characterizes the combined effects of soil management (i.e., organic crop residues and fertilizers), local climate, and soil texture on SOM. It has no direct connection to the current level of SOM storage, but indicates changes in the SOM reproduction due to changes in management and/or climate. If the current SOM level shall be kept under conditions of climate change, the management has to be altered to make sure that C_{rep}/BAT remains constant. We defined a Carbon Demand Index (CDI) as a factor which relates the required C demand for a sustainable SOM reproduction in future (C_{rep}(future)) to the C reproduction flux in the past (C_{rep}(past)):

$$C_{rep}\left(\text{future}\right) = \text{CDI}^{*}C_{rep}\left(\text{past}\right).$$

CDI can also be calculated from the turnover condition values in future (BAT_{future}) and past (BAT_{past}):

$$\text{CDI} = BAT_{future}/BAT_{past}.$$

The CDI calculation provides a simple tool to assess the effect of climate change on SOM reproduction for a given soil type without consideration of the actual SOM level. Site-specific CDI values for Central Germany vary between 1.04 and 1.46, which indicates that due to climate change the C demand generally increases between 4% and 46% until 2100. Aggregated on the level of Biomass-Providing-Units (BPUs), the range is between 7% and 32% (Fig. 40.2). The CDI values show a pattern caused by the distribution and occurrence of soil, landscape, and climate conditions. The lowest CDI values can be observed in (sandy) soils with low clay and silt content. In contrast, soils in bogs, in river valleys, and in loess areas show medium-to-high CDI values, but also indicate partly high variance. The extremely dry conditions in parts of the loess region favor medium CDI values. In contrast, the loess areas with currently higher precipitation amounts show high carbon demand for SOM reproduction. The CDI values corresponding to mountain and hill soils have no clear pattern; they

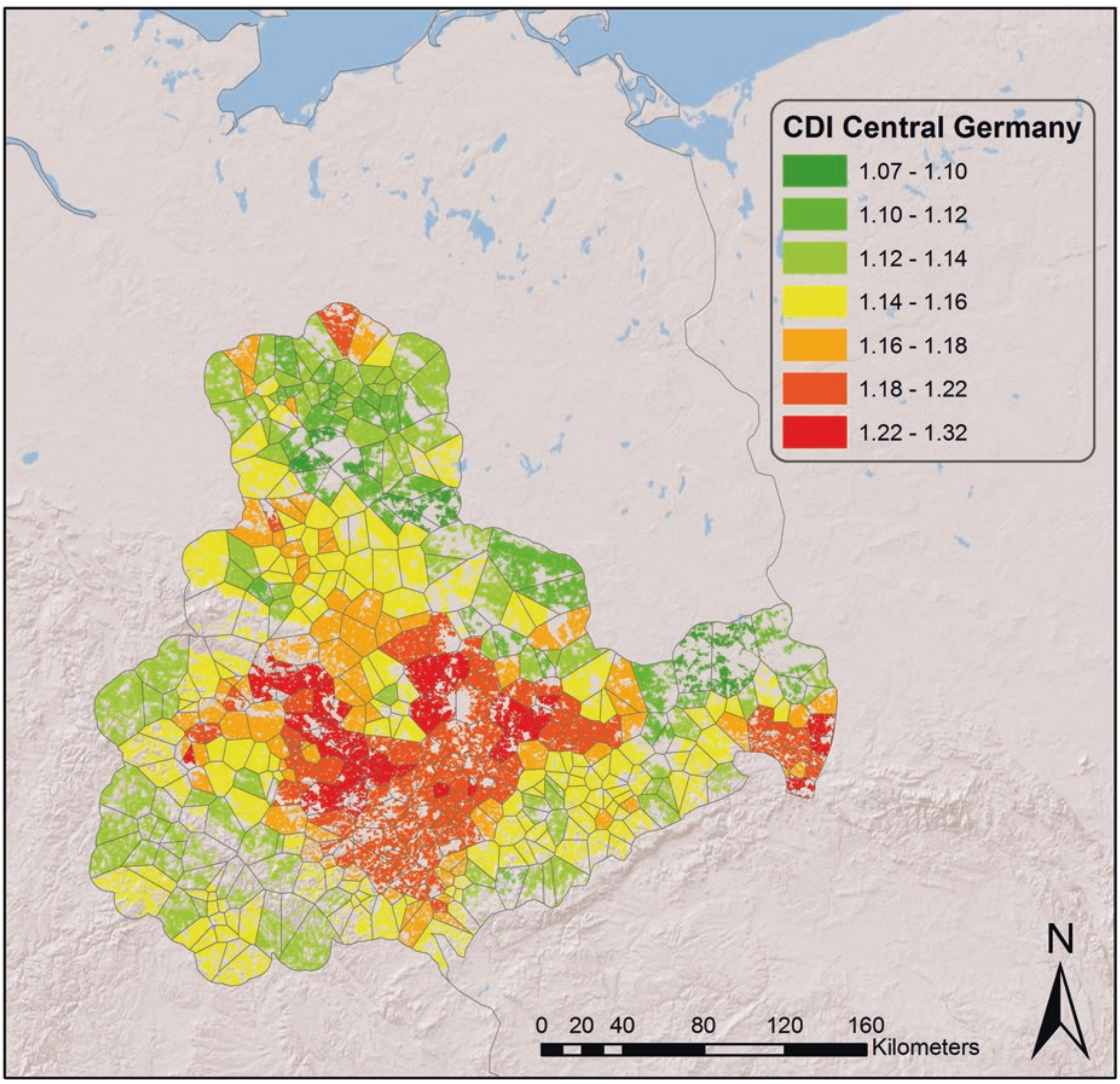

Fig. 40.2 Regional distribution of the Carbon Demand Index (CDI) showing the site-specific increase factor for organic C required sustaining the current SOM level on the BPU scale. (*Modified from* Franko et al. [3]; *with permission*)

show both low and medium results. This can be explained by the great variety of the parent material from different rock types and weathering products, as well as by spatial differences in regional climate change.

The CDI concept does not take into account changes of the seasonal pattern, which may have an effect on the SOM dynamics as well. Thus, the proposed scheme must be seen as a first step that is very useful to identify possible hot spots for analyses on a large scale.

40.3 Regional Biomass Demand Intensity for Biogas Production

When the size and number of biogas plants in a region increases, more agricultural biomass is required, which leads to a rise in competition between soil organic matter reproduction and bioenergy production for fresh organic matter—especially if the available agricultural area is limited. Central Germany has a high density of biogas plants with different capacities. Information on the location and capacity of the biogas plants [6] was used to identify their Biomass-Providing Units. A Voronoi map was generated in ArcGIS to develop separate catchment areas (i.e., for agricultural substrates) for each competing biogas plant. The seeds for the Voronoi diagram had the positions of the biogas plants. Due to potential transport costs, there is a high motivation to get the substrate within short distances. Within its BPU, a biogas plant has the shortest distance to the corresponding agricultural land and therefore the lowest cost for substrate transport.

The demand of a biogas plant regarding agricultural substrates and the consumption of biomass-C can be directly related to the installed capacity of the plant [7]. To regionally differentiate the intensity of matter demand and the possible effects on SOM reproduction, the agricultural area available for biomass provision also must be considered. When preparing a regional analysis, we defined a capacity index (CAP) for each BPU:

$$CAP = IC/AA.$$

CAP represents the installed capacity (IC) in kW per available agricultural area (AA) in ha within a BPU. The available agricultural area is based on CORINE Land Cover data, considering all forms of arable land and pasture.

CAP is used as an indicator describing the regional intensity of matter demand for biogas production. Next to the installed capacity, the density of the biogas plants is especially important. With growing distance to the neighboring plants, more agricultural area is available to mitigate negative effects. CAP varies over a wide range of values (from 0.0055 to 2.5100 kW ha^{-1}) for the individual BPUs in Central Germany having a rather equal distribution in the region. After differentiating BPUs according to their installed capacity into five classes of typical biogas plant sizes (<150 kW, 150–300 kW, 301–500 kW, 501–1000 kW, >1000 kW), we found a constant increase of CAP with growing installed capacity (Fig. 40.3)—which means that higher installed capacities are not compensated by a larger area. Increasing amounts of (co)substrates for larger biogas plants have to be provided from a constant agricultural area. The production and burning of bio-methane takes C out of the agricultural system, which is then no longer available for SOM reproduction regardless of the type of agricultural biomass.

The results are affected by the individual size of the BPUs and thus from the method used for the identification of catchments for agricultural substrates. The use of a Voronoi diagram is a simplified approach, adapted to scale of the study. A detailed discussion of this concept is presented by Franko et al. [3].

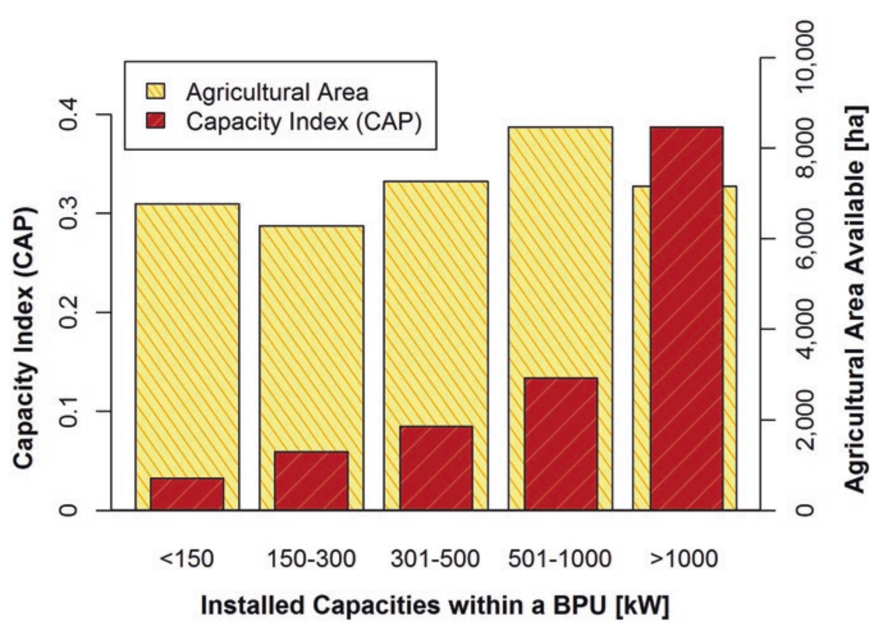

Fig. 40.3 Average values of capacity index (CAP) and available agricultural area represented in typical classes of installed capacities within a bioenergy producing unit (BPU). (*Modified from* Franko et al. [3]; *with permission*)

40.4 Regional Assessment of Carbon Competition

Carbon Demand Index and capacity index indicators are calculated on Biomass-Providing-Unit-level, which enables a spatially comprehensive assessment of the competition for fresh organic matter. To assess the level of conflict between both applications, the indicators were used to differentiate three types of regions (hot spot, warning, and low alert). A possible competition for the fresh OM could most likely be expected in BPUs, which already have a high demand of biomass for energy production and require a high increase of soil organic matter reproduction due to climate change. Therefore, BPUs having both indicators classified as "high" are hot spots. Combinations of "high" in any category with "low" or "medium" in the second category are considered as warning level regions. Remaining areas are classified as low alert regions, as no "high" matter demand for any of the categories could be observed.

Classified hot spots with high demand for fresh OM cover about 5% of Central Germany. They are mostly scattered within the center and the very east of the study region (Fig. 40.4), which is known for its high agricultural productivity. Hot spots are seldom directly connected to each other, but are most commonly connected or even surrounded by regions classified as warning level. Warning level regions can be found in 30% of the area. Within the center and the very east of the study region, they create large transitional areas between the hot spots and low alert regions. Low alert regions, where neither indicator is at high level, cover the main part of the area (65%).

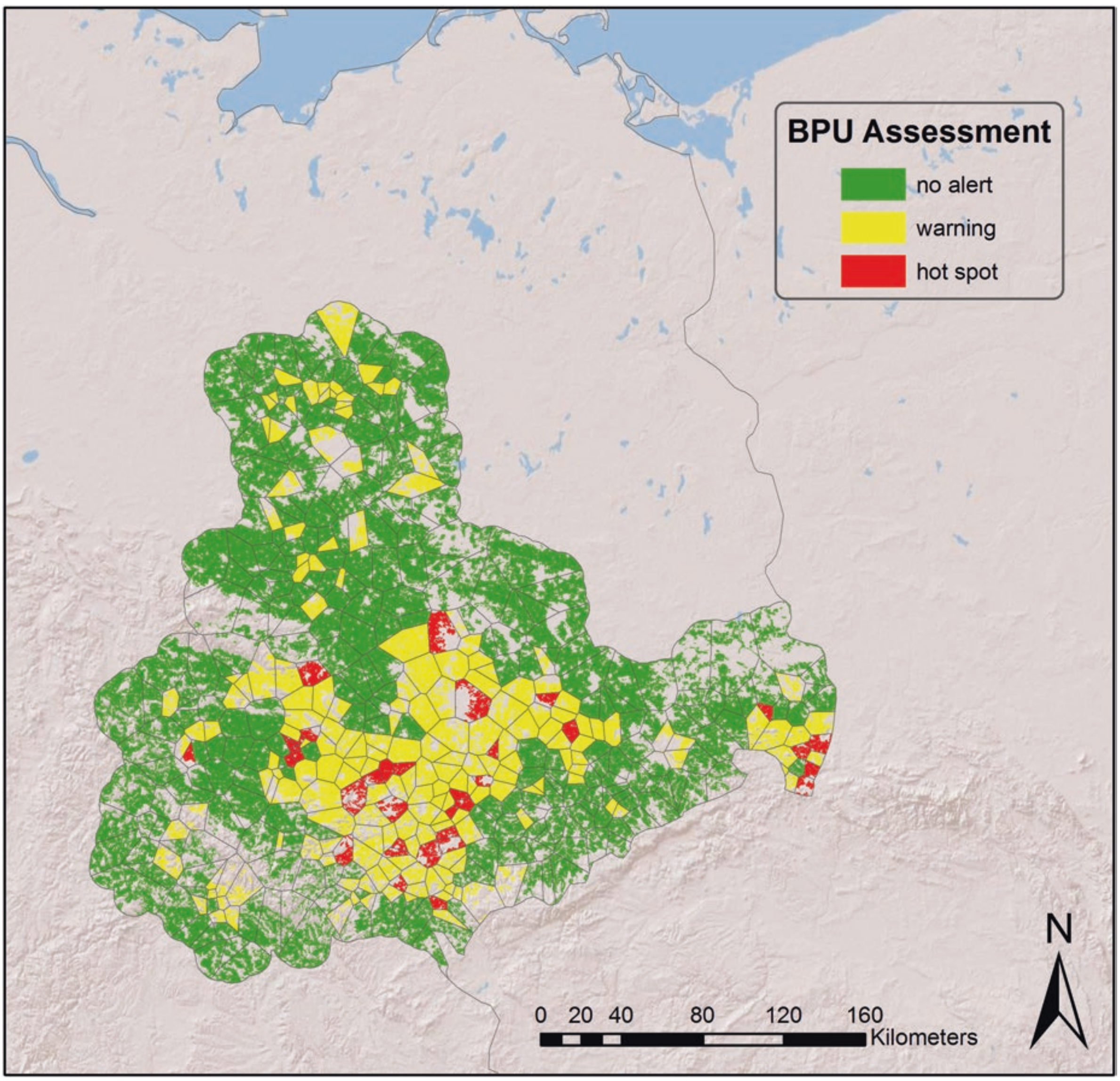

Fig. 40.4 Assessment of the competition level for biomass C on the Biomass-Providing-Unit (BPU) scale showing hot spots with high C demand for biogas production as well as for SOM reproduction and warning on BPU's where only one of the two C demand items is high. (*Modified from* Franko et al. [3]; *with permission*)

The combined assessment of CDI and CAP allows a BPU classification in terms of a potential C competition. The identified hot spots have the highest priority to introduce adaptation measures. Potential measures to stabilize SOM reproduction include, for instance, improved cropping systems with a major fraction of crops that leave higher amounts of OM in the soil. Following a scale-specific procedure as suggested by Volk et al. [8], individual hot spots should subsequently be analyzed on a smaller scale using regional-specific crop rotations and crop management, as well as the substrate and digestate management of the relevant biogas plants. Such further development requires a close collaboration between scientists, state authorities, and farmers who provide the necessary knowledge, experience, and data for the scale.

For future studies, other potential impacts are also worth considering: (1) An inappropriate agricultural management can have an additional negative impact on SOM. (2) The use of more stabilized OM like biochar may help to solve the problem of OM loss. However, it remains unclear if soil functions are supported by biochar in the same way as conventional agricultural carbon sources. (3) Conservation tillage may be a measure to save SOM stocks, but the actual impact must be verified [9]. (4) Afforestation programs could increase C sinks of these areas and mitigate the impact of biogas production and climate change. (5) Demographic changes and related decrease of the population can lead to a lower demand for agricultural products and energy.

To include ecological and spatial effects of biogas plants and energy cropping into planning decisions, legal requirements different from those applied to traditional agricultural production need to be implemented into legislation. Within this context, regulations on water, soil, nature conservation, agriculture, and regional planning could be adjusted, as well as measures for the promotion of renewable energies.

40.5 Conclusion

Current climate change projections for Central Germany indicate that additional carbon sources may be required to sustain the current level of soil organic matter to preserve the soil functions and regulation services. Otherwise, soils may become an important source for carbon emissions if SOM levels decrease. But carbon sources that are suitable for SOM reproduction could as well be used to produce bioenergy that would reduce emissions of CO_2 from energy production based on fossil carbon. The presented study developed two indicators (CDI and CAP) for the large-scale identification of hot spots of high carbon demand for SOM reproduction due to climate change and bioenergy production. The application of the indicators for Central Germany shows a general trend of an increasing carbon demand for SOM reproduction (+4% to +46%). About 5% of the agricultural area was labeled as hot spots of carbon demand. Within these hot spots an intensive biogas production leads to a high risk for sustainable soil management due to additional climate-change-driven pressures on SOM storage. The results reveal that the areas with the highest agricultural productivity overlap with regions of high C demand. Potential measures mitigating this conflict require the quantification of the actual C demand within these hot spots and the development of measures on farm scale. If there are changes in the pattern of bioenergy production, it might be worthwhile to consider the potential conflict with SOM reproduction and concentrate bioenergy production within regions where the climate driven impact on SOM loss is lower.

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41 Removal of Agricultural Residues from Conventional Cropping Systems

Stefan Majer, Daniela Thrän, and André Brosowski

41.1 Introduction

Biogenic residues and waste materials from agriculture are often referred to as part of a globally accessible, unused potential of biomass to produce fibres, biomaterials, biochemicals, and bioenergy, and are thus important ecosystem goods from provisioning services. Even though these residues do not require additional land resources, such as energy crops, and thereby do not have an impact on risks for food security or effects from indirect land use change, a number of sustainability issues (e.g., maintenance of soil organic carbon contents, etc.) have to be considered when assessing a sustainable potential use of these feedstocks. This is important if risks related to the loss or the substantial destruction of agricultural resources or ecosystem services are to be avoided. Furthermore, trade-offs between the opportunity to provide potentially new resources for energy or a bio-based economy, and the loss of ecosystem services due to an unsustainable management of land, must also be avoided.

Amongst the huge variety of biogenic waste materials and residues, straw is often described as one of the most interesting future feedstock options for the production of bioenergy. This is especially the case for countries such as Germany that have large areas of agricultural production.

The vision of agricultural residues as sustainable feedstocks is frequently supported by examples of established utilisations of straw for the production of energy, especially in Denmark. In Germany, the market introduction of technologies for the conversion of straw to heat, power, biogas, or liquid biofuels is not equally far advanced.

Which ecosystem services are addressed? 1. Provision of straw as a resource
2. Regulation of soil fertility and soil erosion

What is the research question addressed? What is the potential of surplus straw as a currently unused agricultural resource without increasing risks for soil degradation and nutrient losses, and thus introducing additional corresponding risks for biodiversity losses?

Which method has been applied? Methods from agro-environmental soil assessments to evaluate the sustainable potential for straw removal rates at regional levels

What is the main result? There is a sustainable potential of straw as a new renewable resource and ecosystem good that could be harnessed without potentially new negative risks or impacts on other ecosystem services

What is concluded, recommended? The approach presented here is suitable (a) to estimate the overall sustainable and site specific potential production of surplus straw as a provisioning service; (b) to identify sustainable hot spot regions for a future intensification of straw utilization and (c) to develop medium- and long-term strategies to unlock the identified potential for the construction of straw conversion facilities without additional risks for biodiversity and other ecosystem services

41.2 Assessing the Availability of Cereal Straw

The amount of cereal straw that could be removed from agricultural land for the production of energy can be evaluated by means of a biomass potential analysis. According to Thrän and Pfeiffer [1] and Kaltschmitt et al. [2], different levels of biomass potentials, including a sustainable potential, can be distinguished (for more details please see Kaltschmitt et al. [2]). For the definition of a sustainable[1]

[1]The term *sustainability* might be a typical example of a concept whose abstract definition usually receives broad consensus, while its actual operationalisation is strongly driven by individual values and conceptions. For the issues described in this chapter, we define the term *sustainable potential* as part of the technical biomass potential, which is defined by the difference of the overall technical potential and the amount of straw necessary for the supply of soil organic carbon and

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straw potential, a number of parameters, such as the yield of straw, cutting height and the dry matter, and the contribution of straw to maintain the soil organic carbon, have to be considered at a regional level. Furthermore, straw that is already utilized (e.g., for animal husbandry or mushroom production) must be evaluated. Thus, the amount of straw necessary for these purposes is determined and subtracted from the technical potential.

The amount of straw for animal husbandry and straw-based housing systems for livestock (e.g., cattle, pig, sheep, horses, chicken, etc.) as well as the amount of straw used for other purposes (e.g., the production of mushrooms) in Germany can be calculated on a regional level based on statistical data and information [3–12].

An additional ecosystem service is the contribution of straw left on the field, which is an important source of organic carbon and can be crucial to maintain the soil organic matter. The amount of straw necessary for this purpose is assessed in various ways in the literature. While some authors tend to use average values for a straw removal rate (e.g., 33–50% in Valin et al. [13], 37–52% described in a review of available literature conducted by Zeller et al. [14]) other authors argue that, owing to strong differences between agricultural regions and existing interlinkages to other agricultural sectors (e.g., carbon import due to fodder import into areas with intense livestock production), the removal rates need to be calculated on a regional level [15].

A regionally specific removal rate for cereal straw can be calculated based on a humus-balance that considers the specific characteristics of a location. The humus-balance of agricultural production areas is often calculated based on the standard humus-balance method in Germany, which has been developed by Körschens et al. [4] and which is typically referred to as VDLUFA method, or "lower (VLV) and upper (VUV) values." The VLV method is usually applied in calculations for soils in good condition and with a sufficient and appropriately managed nitrogen supply. VUV, on the other hand, is typically used for soils missing a proper management of the soil organic carbon supply in the past [4]. In addition to this method, the Dynamic Humus Unit method (DHU) is an approach that originates from organic farming [16, 17].

Following these approaches, Weiser et al. [15] calculated the straw potential (including a site-specific humus balance as well as a consideration of existing utilisations of cereals straw) for every NUTS 3 region in Germany in a stepwise district-by-district analysis.

Firstly, the annual average amount of straw from grain production has been quantified (see Fig. 41.1).

41.3 Regional Distribution of Straw Potentials in Germany

As result of the assessment, Weiser et al. [15] have described a theoretical potential of 29.8 Tg yr^{-1} of straw. As this potential is limited by a number of technical constraints (e.g., not harvestable stubbles, straw utilized for livestock, etc.), a technical potential of 15 Tg yr^{-1} was assessed. Further, soil restrictions were considered to maintain soil fertility, and a sustainable straw potential of 7.97–13.25 Tg yr^{-1} was determined for Germany based on the methods for humus-balancing described above. The resulting bandwidth show significant differences, not only for the three methods applied, but also for the different regions considered (Fig. 41.2).

The VLV method shows generally positive humus-balance results except for two districts, the Lüneburg Heath and the Northern-Upper Rhine Plain, while peak values are reported in the North German Plain and the Rheinish Massif (Fig. 41.1) [14, 15].

The results of the VUV method show 31 districts in humus supply deficit, while the distribution of districts with maximum and minimum supply rates is generally similar to the first method applied.

Based on the results of these humus balances, regionally specific sustainable straw removal rates and associated sustainable cereal straw potentials have been calculated by Weiser et al. [15] to maximise sustainable production and to minimise risks for biodiversity and other ecosystem services, such as soil and water regulations (Fig. 41.3).

According to the first VLV method applied (VLV), 44% of the theoretical potential production can be considered as sustainable surplus straw production (13.25 Tg yr^{-1}, Fig. 41.2). Altogether, 9% of the districts show restrictions regarding the humus balance. The highest total potential at the district level occurs in the Loess Belt and in the Schleswig-Holstein Uplands, while no surplus straw is obtainable, based on the analysis, for 52 districts of Germany [15].

The surplus straw calculated according to the second VUV method (VUV) amounts to slightly less potential of 9.89 Tg yr^{-1}. Overall, 38% of the investigated regions, or 81 districts, show limitations regarding the humus carbon supply. Due to these limitations, 81 districts cannot sustainably provide straw for energy purposes without posing risks to other ecosystem services. Peaks are shown for the Mecklenburg Coastal land. The third DHU approach shows the lowest sustainable straw potential of 7.96 Tg yr^{-1}, which correlates to 27% of the overall theoretical potential. With no sustainable straw potential, 32% of the districts show

additional, already established utilisations such as animal husbandry. Since we aim to estimate the sustainable potential of cereal straw from agricultural production systems, we apply this definition to the whole of the agricultural production in Germany. This includes both conventional and organic farming.

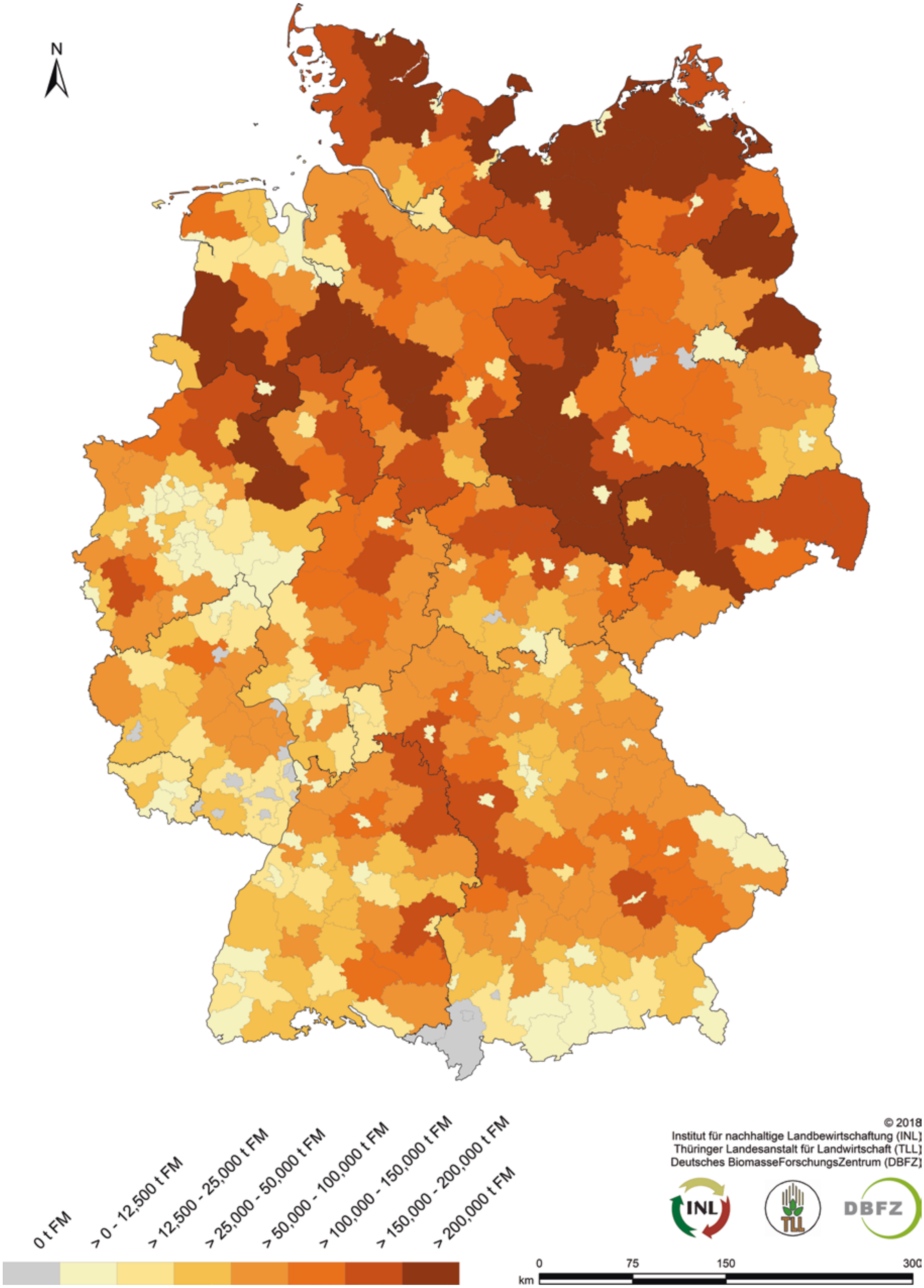

Fig. 41.1 Straw production on county level in t fresh matter (FM) [14]

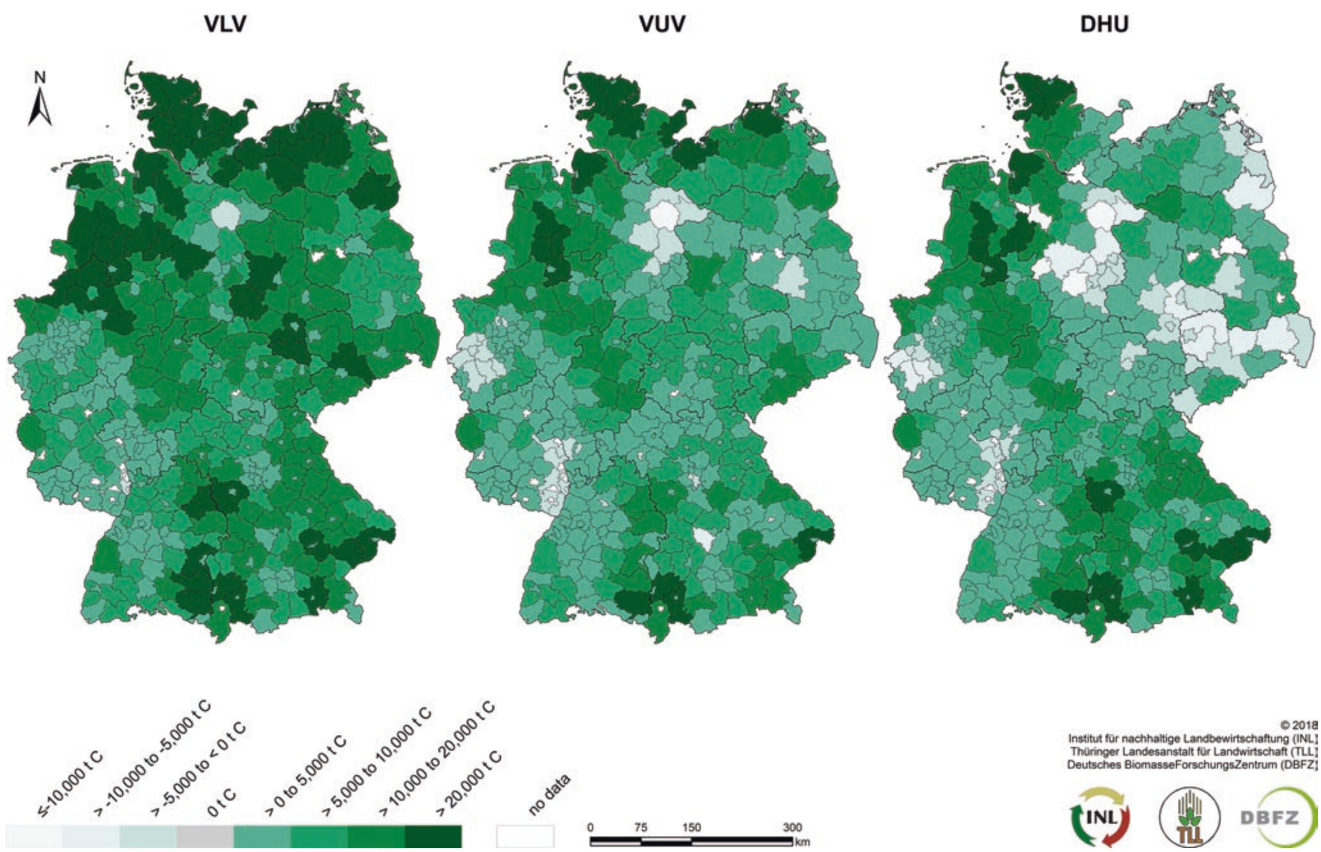

Fig. 41.2 Soil carbon balance on county level in t C according to VDLUFA method “lower (VLV) and upper (VUV) values” as well as to Dynamic Humus Unit method (DHU) [14]

limitations. Peaks are again shown for the Central North German Plain or in the Loess Belt.

Brosowski [18] has worked on a greater spatial resolution of the potential identified by Weiser et al. [15] on a square kilometre basis (Fig. 41.4). These results can be used for additional investigations, such as identification of suitable locations for biomass conversion plants.

41.4 Discussion

Straw is an agricultural residue that contributes to the humus balance as an important element for the maintenance of soil fertility and health. The specific demand depends on different soil and climate conditions and differs widely from place to place. On the other hand, straw is a resource for bioenergy and biomaterials, which does not need additional land for cultivation. To unlock this potential in a sustainable way, strategies for the exploitation of their potential have to be based on a spatially explicit approach that respects local soil conditions and existing contributions of straw to the humus balance and thus to soil fertility and health. This is important if trade-offs are to be avoided between the use of a potentially interesting resource for energy or bio-based products and the risks to ecosystem services through soil degradation caused by unsustainable management of agricultural resources. This requires, amongst other things, site-specific knowledge of the carbon and humus-balance of a region. Even though straw is often highlighted as a sus-

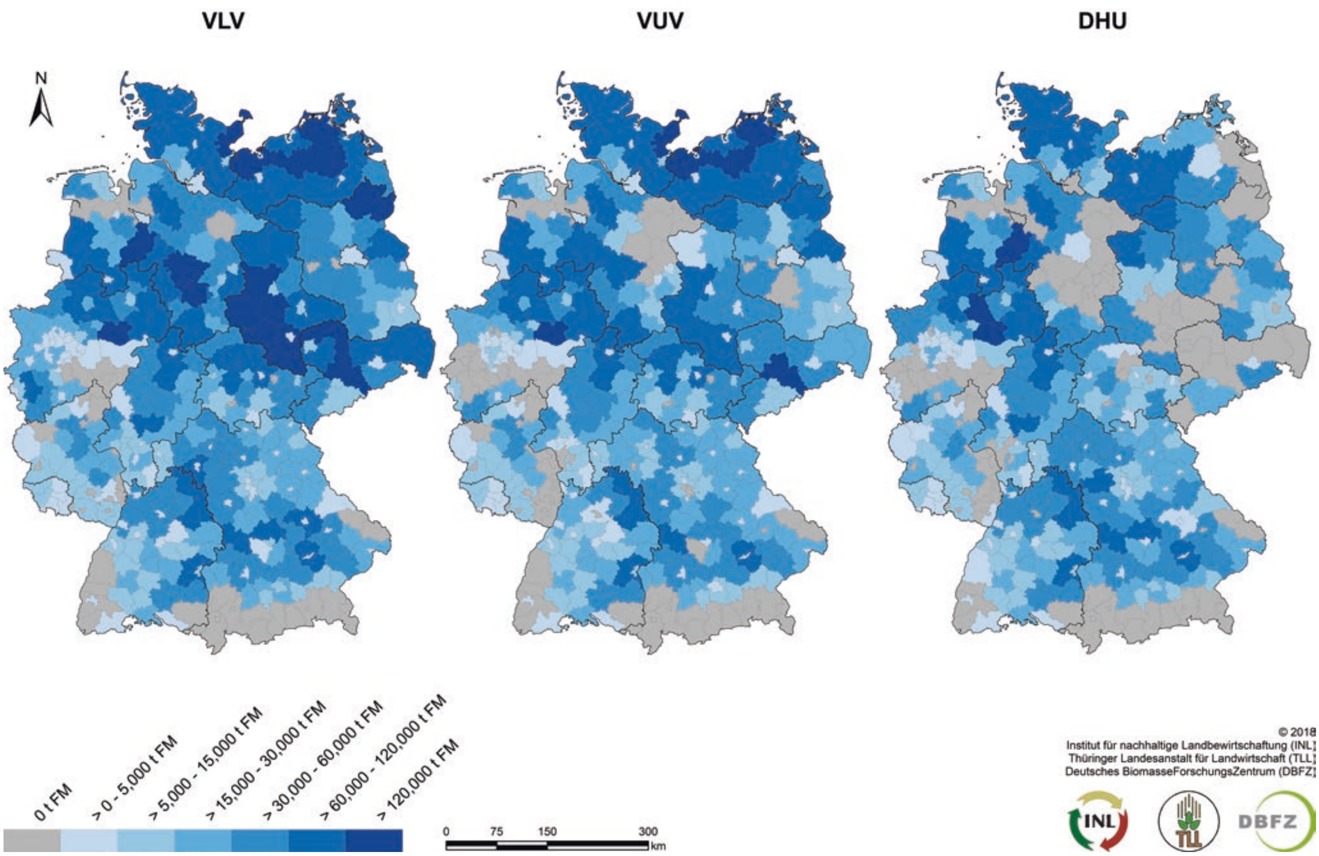

Fig. 41.3 Straw potential on county level in t fresh matter (FM) according to VDLUFA method "lower (VLV) and upper (VUV) values" as well as to Dynamic Humus Unit method (DHU) [14]

tainable feedstock for the production of energy, an assessment of the actual potential of this agricultural residue under consideration of a regional specific humus balance and other existing utilisations of straw is currently only available for Germany.

To estimate the necessary contribution of straw to maintain the soil organic carbon within a specific region, the results available for Germany have been calculated with consideration of different approaches. Consequently, the calculated potential of surplus straw ranges from 7.96 to 13.25 Tg yr^{-1}. The information available can help to develop strategies for the utilisation of the surplus straw identified without increasing risks for soil degradation and thus introducing additional corresponding risks for biodiversity losses. The approach presented here is suitable (1) to estimate the overall sustainable and site-specific potential of surplus straw, which can serve as a resource or provisioning ecosystem good; (2) to identify hot spot regions for a future intensification of straw utilization; and (3) to develop medium- and long-term strategies to unlock the identified potential. On the other hand, it should be noted that future activities for the implementation of utilisation strategies for straw on a regional level need to be accompanied by additional investigations to include a greater level of detail, regional data, and characteristics to identify locations suitable for the construction of straw conversion facilities and to exclude additional risks for biodiversity and ecosystem services.

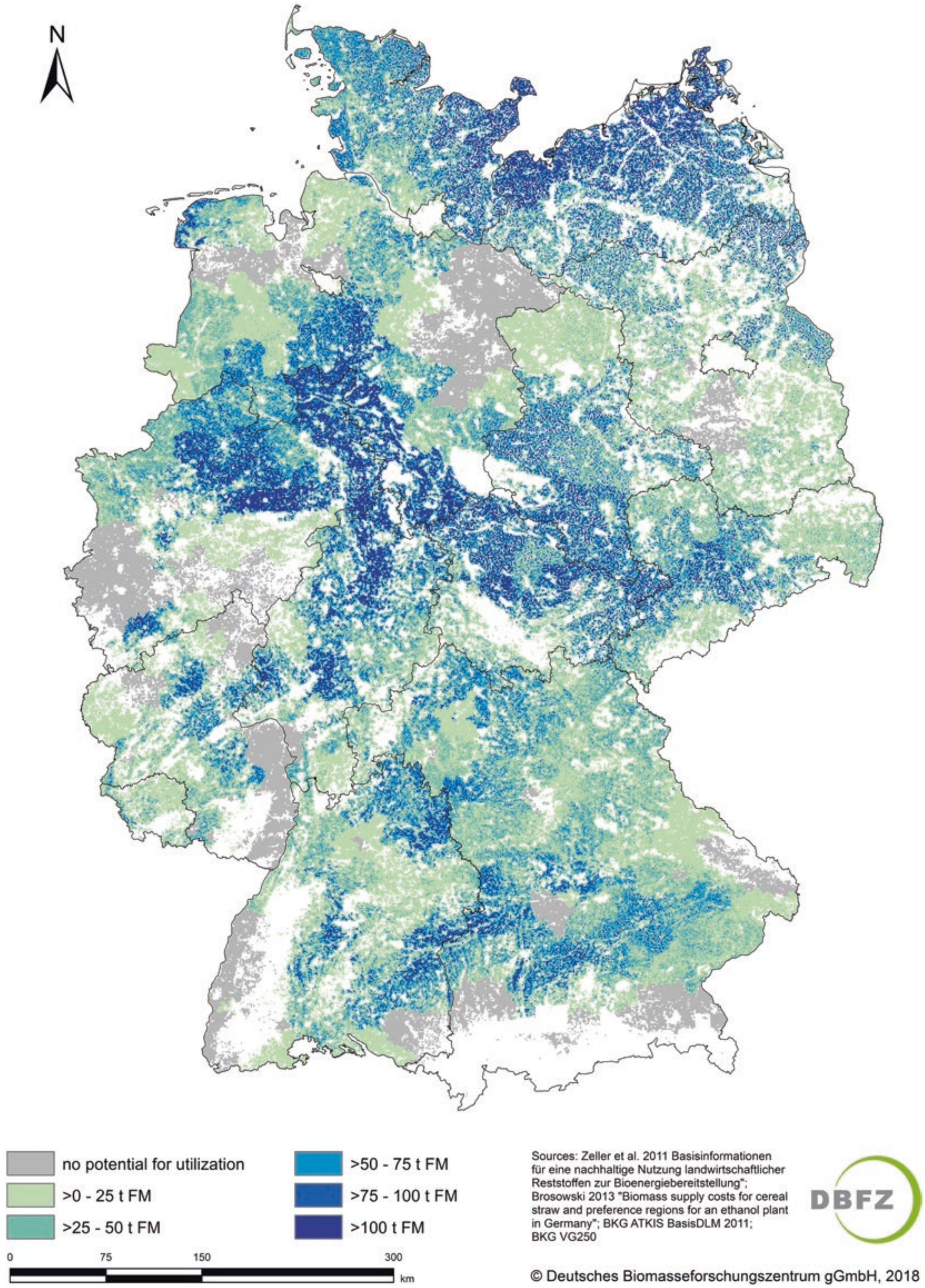

Fig. 41.4 Sustainable straw potential in Germany [14, 18]

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Shrinking Cities and Ecosystem Services: Opportunities, Planning, Challenges, and Risks

42

Dagmar Haase, Annegret Haase, Dieter Rink, and Justus Quanz

42.1 Conceptualizing and Operationalizing the Nexus

Urban shrinkage has become an issue for urban planning and policy in Europe. Shrinkage implies dramatic land-use impacts, including under-utilisation, vacancy, demolition, emerging brownfield sites, and de-densification. However, shrinkage also offers great potential to "re-create"—that is, to enhance and implement—urban green space, including the ecosystem services it provides: Local climate and air quality regulation by trees that grow on abandoned land, carbon sequestration and storage by vegetation on vacant lots, preservation or enhancement of urban biodiversity, and recreational facilities that support the mental and physical health of the inhabitants through the enlargement of parks and woodlands. This contribution argues that there is a linkage—a nexus—between shrinkage and ecosystem services provisioning, and we present a way to frame it. Using the example of Leipzig, we have developed a matrix approach that links the potentials of land use (change) related to urban shrinkage with ecosystem services provisioning in cities. Through a discussion of these potentials, challenges, and the relevant strategies of urban planning such as interim uses, urban afforestation, and community gardens, and using a quantitative model study for the whole city and single show cases, we show how planning policy in shrinking cities could benefit from considering the nexus between shrinkage and urban ecosystem services provision (Fig. 42.1) [1]. Our evidence comes from a Central European geographical context which determines the explanatory power of the data and conclusions to be valid first and foremost to cities from this area.

The concept of the "shrinkage-ecosystem services nexus" sets typical shrinkage-related land uses into a connection with ecosystem services provided by these respective land covers as seen in Table 42.1. The following example demonstrates how to read Table 42.1: characterising the type of a "lower impact nexus," vacant abandoned buildings reduce the heat production in the city and thus (to a limited extent) support the cooling function of surrounding green spaces. Vegetated

Which ecosystem services are addressed? Air filtration, air cooling, physical and mental recreation, food production, flood regulation. 2. Regulation of soil fertility and soil erosion

What is the research question addressed? How can we make use of the "shrinkage dilemma," which means the under- and non-use of space, and change it for something positive and sustainable? How can we improve human health and well-being in our cities by employing low-cost green infrastructure approaches? Where are suitable spaces to do so?

Which method has been applied? Field and survey data analysis, land use/cover data analysis, field mapping, 11interviews, policy document analysis

What is the major result? We developed a linkage matrix for land use, five ecosystem services, and human health/quality of life benefits under shrinkage conditions and a respective empirical database

What is concluded, recommended? Open spaces and brownfields emerging under shrinkage provide—when greened—a multitude of ecosystem services for urban residents. Under regrowth, these new green spaces are running the risk of being replaced by land uses with higher return rates, such as housing or commercial sites, in the course of a possible turn of a shrinking city towards new growth

soil and lawns at demolished sites contribute to almost all ecosystem services listed due to their biochemical potential to emit moisture and essentials, to store CO_2, and to serve as ground for vegetable/fruit production. Trees at such sites additionally provide shade and cool the air by 1.5–3 K [2].

Based on data and empirical evidence from a full range of studies including our own modelling work and field research (see Table 42.2, [1]), we developed a linkage matrix for the

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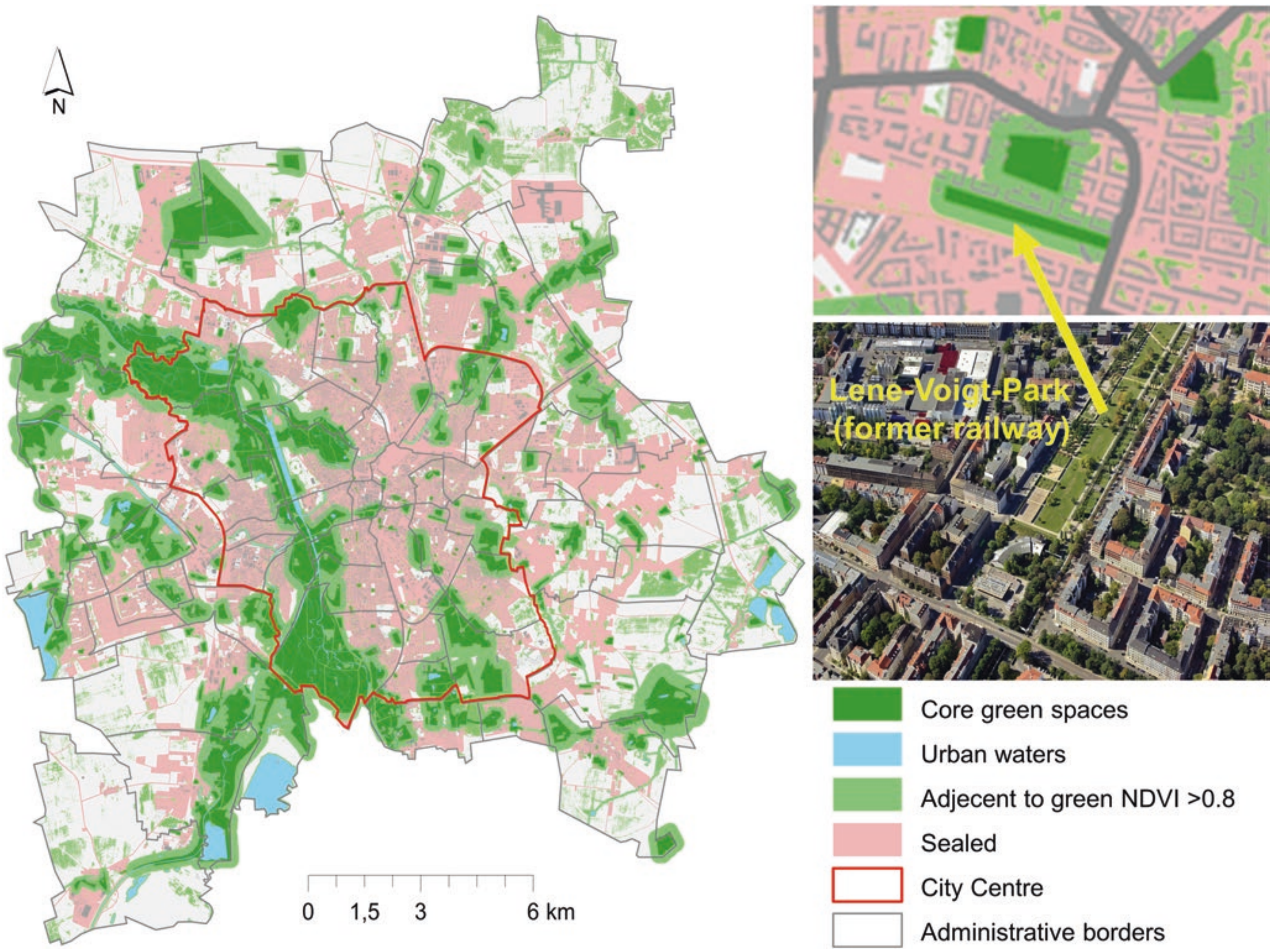

Fig. 42.1 Cooling map for Leipzig including existing (dark green colour) and potential (light green colour) green spaces as a consequence of demolition and non-use due to shrinkage. (Data sources: Strohbach et al. [6] and cadastral map by the Saxon Landesvermessungsamt Leipzig 2014)

Table 42.1 Shrinkage-related land use/cover patterns (processes) and their impact on urban ecosystem services[a]

	Urban ecosystem services					
Land use/cover	Air filtration	Carbon storage	Cooling	Rainwater and flood regulation	Food production, community gardens	Biodiversity
Vacant built land	0	0	X	0	0	XX
Vacant sealed land	0	0	0	0	0	0
Bare soil	X	0	0	XXX	0	X
Vegetated soil, lawn	XX	X	X	XXX	X	XX
Trees	XXX	XXX	XXX	XX	0	XXX

[a]XXX = highly suitable; XX = suitable; X = less suitable (ambivalent); 0 = not suitable/hindering. The assessment is based on a series of previous studies by the authors where models were used to determine ecosystem functions and performances for the exact land uses/covers listed in column 1. (*From* Haase et al. [1]; *with permission*)

same ecosystem services listed in Table 42.1 and human health/quality of life benefits listed in Table 42.3. From Table 42.3, it becomes especially obvious that each of the listed services has large benefits for several aspects of human quality of life, such as health, recreation and leisure, and food availability, but also for good housing or safety (Tables 42.3 and 42.4; [1]). These examples demonstrate how to read Table 42.3: If rainwater is captured by open soil or vegetated surfaces it cannot contribute to street flooding and makes urban residents safe from flood risk. A reduced particle number filtered by trees lowers the risk of negatively affecting the human respiratory system [3, 4]. Food production and the realization of the “edible city” positively contribute to mental well-being and social cohesion in cities [5]; we can state the same for biodiversity [6]. These benefits to urban residents result from urban shrinkage and population loss.

Table 42.2 Empirical data values that support the findings and the argumentation of this section[a]

Land use class	Subclasses	Air filtration (PM_{10} mean ($\mu g/m^3$))	Carbon storage as above-ground carbon stock [$Mg\ C\ ha^{-1}$]	Cooling (temperature reduction in K)	Rainwater and flood regulation (in mm surface runoff)	Food production (binary 0/1)	Biodiversity neighborhood density of birds)	Safety (binary 0 not safe … 1 safe)	Health LDEN Averaged noise level (dB (A))	Recreation (binary 0/1)	Home (binary 0/1)
Vacant built land	Dense urban fabric	37.79	20.01	0	325	0	52	0	63.89	0	1
	Low-density urban fabric	24.75	13.70	1.2	125	1	40	1	58.45	1	1
	Industry, commercial land	36.52	8.52	0	425	0	14	0	63.81	0	0
Mean		33.02	14.08	0.40	291.67	–	35.33	–	62.05	–	–
Standard dev		7.19	5.75	0.69	152.75	–	19.43	–	3.12	–	–
Vacant sealed land	Roads	71.01	0	0	400	0	7	0	80.02	0	0
	Railway	62.99	0	0	400	0	5	0	85.11	0	0
	Airport	75.01	0	0	250	0	11	0	90.15	0	0
Mean		69.67	–	–	350	–	7.67	–	85.09	–	–
Standard dev		6.12	–	–	86.60	–	3.06	–	5.07	–	–
Brownfields		26.25	0	0	200	1	35	0	61.69	1	0
Bare soil		25.10	0	0	0	1	32	0	59.25	0	0
Vegetated soil	Urban green spaces	19.39	29.38	2.9	10	0	71	1	57.70	1	0
	Allotment gardens	22.07	13.48	1.3	35	1	58	1	59.07	1	0
	Urban forests	16.91	75.71	3.3	25	0	69	0	56.75	1	0
Mean		19.46	39.52	2.50	23.33	–	66	–	57.84	–	–
Standard dev		2.58	32.33	1.06	12.58	–	7	–	1.17	–	–
Trees		0.38	–	2.5	0	–	–	–	8.00	1	0

[a]Data source: data and underlying literature are compiled in Haase et al. [1]; *with permission*

Table 42.3 Linking quality of life benefits to urban ecosystem services[a]

Quality of life benefits	Urban ecosystem services					
	Air filtration	Carbon storage	Cooling	Rainwater and flood regulation	Food production, community gardens	Biodiversity
Safety	0	0	0	XX	0	0
Health	XX	XXX	XXX	0	XX	XXX
Home, place to stay	0	0	0	XX	0	0
Recreation, leisure	XXX	XXX	XXX	0	XX	XXX
Food	XX	0	XX	0	XXX	0

[a]XXX = large benefit; XX = benefit; 0 = no benefit. (*From* Haase et al. [1]; *with permission*)

Table 42.4 Synergies and trade-offs between land use policies/instruments and urban ecosystem services performance[a]

Land use policy	Urban ecosystem services			
	Air filtration, carbon storage, cooling	Rainwater and flood regulation	Food production	Biodiversity
Newly built housing HD[1]	T_{QoL}	$T_{econ/QoL}$	T_{econ}	$M_{econ/QoL}$
Newly built housing LD[2]	M_{QoL}	$M/T_{econ/QoL}$	M_{econ}	$M_{econ/QoL}$
Interim use (urban gardening)	S_{QoL}	$S_{QoL/econ}$	M_{econ}/S_{QoL}	M_{econ}/S_{QoL}
Interim use (parking lots)	S_{QoL}	S_{QoL}	$M_{econ/QoL}$	T_{econ}
Water	S_{QoL}	S_{QoL}	Arbitrary[3]	T_{econ}
Green spaces (parks, meadows, afforestation)	M_{econ}/S_{QoL}	S_{econ}/S_{QoL}	T_{econ}	M_{econ}/S_{QoL}
"Wait and see"	Arbitrary[4]	Arbitrary[4]	Arbitrary[4]	Arbitrary[4]

[1]High density; [2]low density; [3]water: can be positive if there is fish production and negative if there is no other food production possible because of the water surface; [4]wait and see: could be positive, e.g., niches for semi-legal use, and negative, e.g., decrease in ground value or housing prices because of the bad image of derelict land. (*From* Haase et al. [1]; *with permission*)
[a]Both the synergy and trade-off assessments were based on the data of Table 42.1 but also include field experience from other cities, 'non-quantitative evidence' (e.g., communications concerning the impact of land transformation in cities by planners, visitors or house owners) and literature evidence. T_{QoL} = trade-off with quality of life, S_{QoL} = synergy with quality of life, M_{QoL} = minor impact on quality of life, T_{econ} = trade-off with economic values, S_{econ} = synergy with economic values, M_{econ} = minor impact on economic values

Quantifying our nexus-approach to the city of Leipzig, we found that all areas of bare soil resulting from de-sealing and the demolition of houses offer potential gardening space of about >3000 ha. Considering the annual production of fruits and vegetables of an allotment or community garden in Leipzig, bar soil areas can "produce" about 10,000 kg/ha of organic food per year. If planting trees along the street and in backyards, implementing lawns, and, to a limited extent, installing green roofs at newly built townhouses at all demolished sites and sites with high housing vacancy that might get demolished, we calculated some 54,958,000 particles per m^3 air that are annually filtered/captured by the respective vegetation. Converting all demolished former residential sites into parks, for example, would increase the public recreation area to almost 4000 ha. In addition, 71,445,400 kg CO_2 could be annually taken up from the urban atmosphere in Leipzig. Figure 42.1 shows the cooling space of the city, including not only existing green spaces but also those virtually implemented as shrinkage-following land use (dark green) and, last but not least, showing in light green those neighbourhoods that might be positively impacted by cool air due to being in the vicinity of the green spaces (Fig. 42.1). The example shown refers to the city of Leipzig and the Lene-Voigt-Park.

42.2 Synergies, Trade-Offs: A Heuristic Model and Examples for Potentials and Risks

Table 42.4 shows the multiple relations of the nexus between land cover types and land use in shrinking cities and their potential to provide (or to restrict provision of) ecosystem services, including an assessment of existing synergies, trade-offs, and conflicts (risks). The assessment is based on different types of knowledge: from empirical and modelling data [1], field experience and qualitative data (e.g., based on interviews and other communications with involved people and stakeholders), as well as literature evidence.

Two examples will be expanded on in more detail here: low-density housing on vacant or post-demolition sites, and interim/temporary uses of vacant land as urban or neighbourhood gardens.

First, *low-density housing*, the second policy from Table 42.4 (see also Fig. 42.2), has minor positive/negative effects on air filtration, carbon storage, and air cooling because low-density housing is normally accompanied by some greenery around the building—shrubs and flower beds, for example—but not large trees that can store substantial

	Air filtration, Carbon storage	Rainwater regulation	Food production	Biodiversity
New built housing (LD)	M_{QoL}	$M/T_{QoL\ econ}$	M_{econ}	$M_{QoL\ econ}$

Fig. 42.2 Synergies (S), trade-offs (T) and marginal effects (M) for and between selected ecosystem services for the land use of newly built housing (LD = low density) as a re-use of shrinkage-driven brownfields

amounts of carbon [6]. This type of housing provides some open spaces between the buildings (except in the case of rows of low-density buildings) that allow rainwater to flow off more easily during extreme rainfall events or floods. This decreases the danger of economic damage for the houses and their inhabitants. Studies of the city of Leipzig state that high- and medium-density housing surfaces lead to more rainwater runoff (urban areas 60–80% [7]), whereas low-density housing surfaces only produce 40–60% rainwater surface runoff. Nevertheless, compared to forest areas with "only" 13% direct rainwater runoff, low-density housing has an impact on storm water retention, but not as much as medium- and high-density housing. Surfaces that are not completely but only partly sealed (e.g., grass paving blocks and half-pervious surfaces) allow a runoff of at least 60% [8]. Those surfaces are more frequently found around townhouse developments than in high-density housing areas. Using these empirical values, the respective water balance for other shrinking cities (precipitation, evapotranspiration, topography, filtration) can be produced, and the stormwater risk can be estimated.

Low-density housing implies the use of green spaces close to buildings as gardens, which contribute to food production (non-commercial, subsistence) and reduce living expenses. In terms of health, Ferrante and Mihalakakou [9] demonstrated that planting just one tree per one-story house can produce energy savings for cooling between 12% and 24%. Low-density housing and the accompanying gardens/green spaces can contribute to biodiversity because they provide habitats [10]. This increases the (perceived) quality of life for the inhabitants, which in turn might increase the economic value of this housing (for the owners/sellers); see Gruehn et al. [11] for Germany and, for more general considerations, see Gómez-Baggethum and Ruiz-Perez [12] and Köhler and Clements [13]. Furthermore, green roofs, which are easy to implement on new flat-roofed single houses or urban villas, can retain 25–100% of rainfall, depending on the rooting depth, roof slope, and the amount of rainfall [13]. Green roofs, which absorb CO_2 during the daytime, are also able to reduce atmospheric levels of CO_2 in the nearby area by as much as 2% on a sunny day [14].

Second, *interim use* (e.g., community or neighbourhood gardening, Fig. 42.3), the third policy from above in Table 42.4, has positive effects on the quality of life with respect to air filtration, carbon storage, and cooling through various types of vegetation, including trees. It also offers airflow corridors [15]. This form of land use also facilitates

	Air filtration, Carbon storage	Rainwater Regulation	Food production	Biodiversity
Interim use (urban garden)	S_{QoL}	$S_{QoLecon}$	$M_{econ} S_{QoL}$	$M_{econ} S_{QoL}$

Fig. 42.3 Synergies (S), trade-offs (T) and marginal effects (M) for and between selected ecosystem services for the land use of the interim use form of community gardens as a re-use of shrinkage-driven brownfields

rainwater runoff and provides potential protection in case of flooding. Urban gardening on interim use sites has clear synergies with respect to the quality of life because many urban gardeners are activists and deliberately engage in this type of work [4]. It also improves their health. In economic terms, urban gardening has only minor effects because, although it produces food, it results in quantities that are far from marketable, in contrast to urban farming, which must be treated as another type of land use practice [1]. With respect to biodiversity, urban gardens offer numerous niches for urban flora and fauna and increase the quality of life in the sense that users of the gardens can experience/enjoy biodiversity and wildlife [2]. Urban gardens may also increase the rent or price of adjacent housing because they are seen as a pull/quality factor of the residential environment [11].

42.3 Challenges for Urban Planning in the Context of Urban Shrinkage

Brownfields and open spaces emerging because of shrinkage do not deliver the above-mentioned ecosystem services *per se*: They have to be opened to the public, designed for specific purposes and uses, and they have to be promoted. Moreover, as they often stock on low quality porous substrates, they need irrigation and sometimes melioration, in case they are situated in topographic depressions. These maintenance tasks thus arising for urban planning and governance are often neglected or simply not recognized by public authorities or other stakeholders. Whereas the value of established green spaces for ecosystem services provisioning is not questioned, the case of brownfields and newly emerging green and open spaces is different: Here, the skepticism towards their value for recreation, regulation, and health is obvious. It needs a different, novel perspective to perceive and identify the potential of this “legacy of shrinkage.” It is not by accident that neighbourhoods, groups, and initiatives are establishing new or interim uses on brownfield sites and, in doing so, exploring their value for urban ecosystem services. Only afterward does planning or policy come into play to promote or save these spaces and open them to the broader public. Typically, informal agreements or structures with limited scope are the outcome of these first engagements of policy and planning in shrinking neighbourhoods. As a result, different forms of interim uses like urban gardening, urban forest, and—sometimes—urban wilderness emerge. To fully deploy and protect the ecosystem services provided by these sites in the long run, they have to be somehow institutionalized. This could be done through creating public property or by the conversion into a public (municipal) green space. Other solutions might be the acquisition by neighbourhood or local civic associations or long-term contracts such as lease-in-perpetuity. In addition, the interim use sites must be codified within municipal zoning plans as new types of green infrastructure. Otherwise, these new green spaces and ecosystem service providing units run the risk of being replaced by land uses with higher return rates, such as housing or commercial sites, in the course of a possible turn of a shrinking city towards new growth.

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43

Spatial Patterns of Ecosystem Service Bundles in Germany

Andreas Dittrich, Ralf Seppelt, Tomáš Václavík, and Anna F. Cord

43.1 Introduction

The EU Biodiversity Strategy to 2020 [1] requests all member states to map and assess the state of their ecosystems and related services to halt biodiversity loss and habitat degradation. Against this background, a scoping study for implementing a national ecosystem assessment in Germany was recently conducted [2] and a first set of suitable indicators for mapping ecosystem services was published [3, 4]. Our study builds on these efforts by analyzing ecosystem service bundles, defined as "sets of services that appear repeatedly together" [5], to compile and synthesize information on ecosystem services for decision-makers in Germany. Ecosystem service bundles allow a systematic and synoptic description of landscapes based on the importance and co-occurrence of different ecosystem services. This provides insights regarding differences in ecosystem services provision and use across space [6] and time [7]. Considering multiple ecosystem services is also essential to obtain a better understanding of how ecosystem service trade-offs and synergies may vary among different regions [8].

Which ecosystem services are addressed? Timber production, crop production, crops for bioenergy use, livestock production, water quality regulation, recreation, erosion control, pollination, nitrogen retention, flood regulation

What is the research question addressed? How are landscapes in Germany characterised by their ecosystem service trade-offs and bundles, and how does this help to support land management in Germany?

How is the spatial configuration of ecosystem service bundles (ESB) characterised by underlying socio-environmental gradients?

Which method has been applied? Self-organising maps applied to twelve ecosystem service indicators to determine ESB. Likewise socio-environmental cluster (SEC) based on 18 covariates have been delineated and a spatial overlap analysis with the ESB was finally conducted

What is the major result? Eight ESB have been identified, allocated in different regions and dominated by different ecosystem services. ESB dominated by provisioning services spatially co-occurred mainly with SEC characterized by environmental variables; environment still highly important for provisioning services beside technical progress

What is concluded, recommended? Spatially explicit analyses of ecosystem service associations can support strategic planning and prioritization of environmental issues at the national and international level. This can aid understanding which ecosystem services are most important for specific regions

43.2 Data and Methods

We used the method of self-organizing maps [9] to identify and map eight ecosystem service bundles (Fig. 43.1a). To this end, we first collected and synthesized quantitative spatial data of 12 provisioning, regulating/maintenance, and cultural ecosystem services in Germany (see Fig. 43.2). The analysed indicators, which referred to ecosystem service potential, supply, or demand, resulted from either own calculations based on public statistics and geospatial data, or modelled results from other authors [10–15]. Input variables had been standardized prior to self-organizing maps calculation, and differences in ecosystem service values per ecosystem service bundle could hence be interpreted as deviation from the German national average represented by zero (see Fig. 43.2). Using these standardized ecosystem service values, the absolute values for all services belonging to the same ecosystem service section (provisioning, regulating/maintenance, and cultural) were summed up to determine how important the three sections of ecosystem services were for each ecosystem service bundle. This approach allowed us

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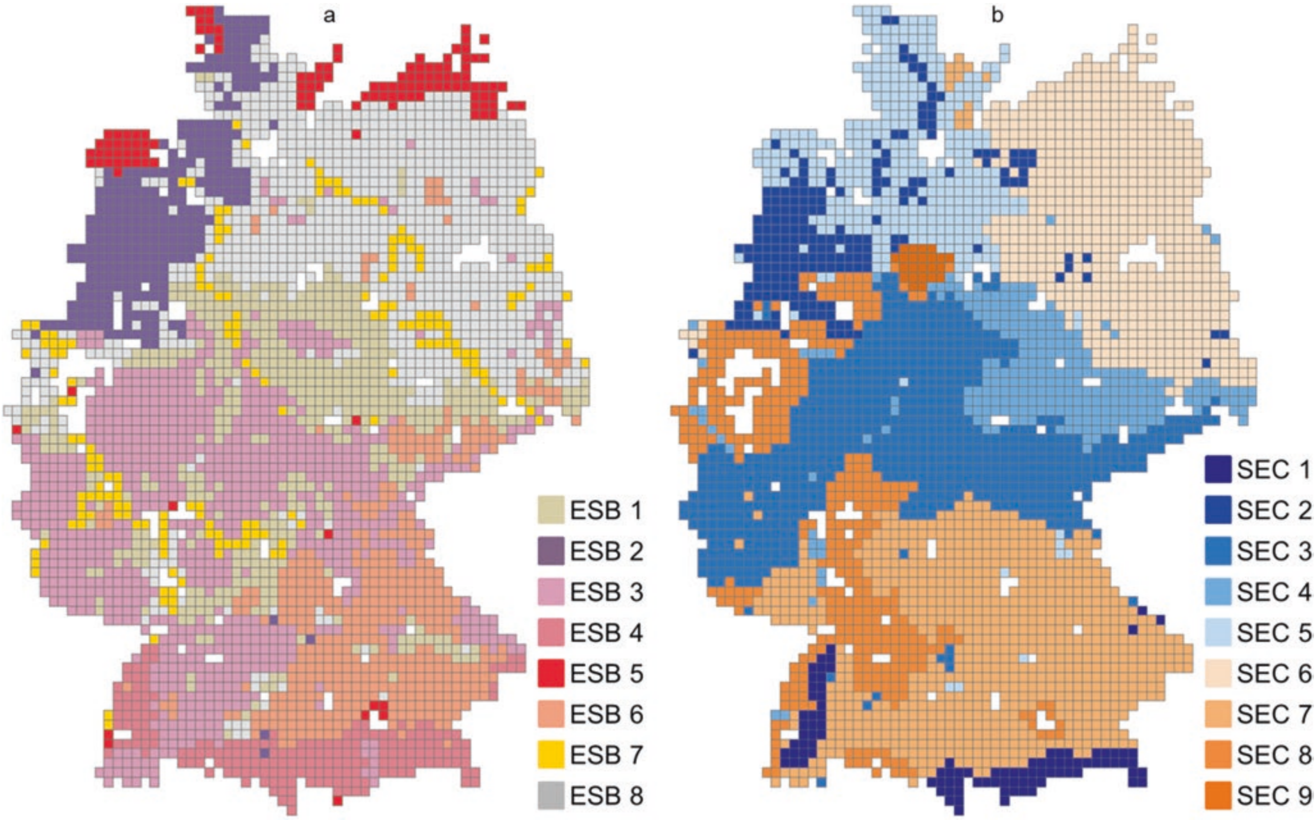
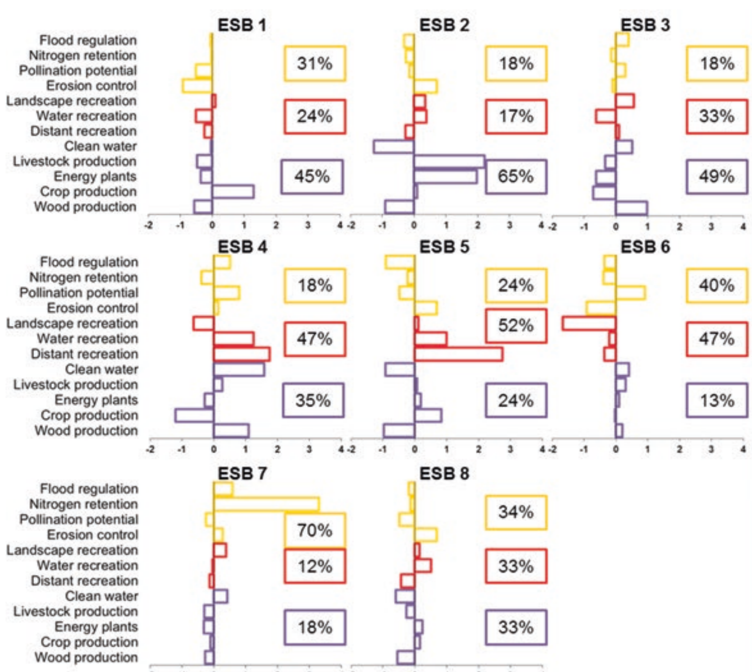


Fig. 43.1 (**a**) Ecosystem service bundles (ESB) and (**b**) socio-environmental cluster (SEC) mapped in Germany. Ecosystem service bundles are dominated by provisioning (purple), regulating/maintenance (yellow) or cultural services (red) (see Fig. 43.2), whereas socio-environmental clusters are either dominated by environmental (blue) or socio-economic (orange) covariates (see Fig. 43.3). Colour intensity illustrates the degree of dominance from dark (strong) to light (week). (*Adapted from* Dittrich et al. [16]; *with permission*)



ES name (category)	Indicator (unit)
Provisioning services	
Wood production (P)	Harvest likelihood combined with production statistics (m³/km²/yr)
Crop production (S)	Energy produced by main crops (GJ)
Energy plants (S)	Amount of methane provided by crops for biogas production (hm³)
Livestock production (S)	Number of livestock units (LU)
Clean water (P)	Nitrogen concentration in rivers (mg N/l)
Cultural services	
Distant recreation (D)	Number of overnight stays (No./km²)
Water recreation (P)	Lakes within 6 km radius of settlements (% area)
Landscape recreation (P)	Protected areas within 6 km radius of settlements (% area)
Regulating/maintenance services	
Erosion control (S)	Management-dependent soil loss due to water erosion (dt/ha/yr)
Pollination potential (P)	Habitat suitable for pollinators (% area)
Nitrogen retention (S)	Ability of rivers to remove nitrogen
Flood regulation (S)	Biophysical dependent flood regulation by catchments (0 – 1)

Fig. 43.2 Bar charts showing deviations of the standardized ecosystem service indicators from the German national average (equal 0 at the x-axis) per ecosystem service bundle (ESB); colors of bars indicate affiliation of ecosystem services (ES) to one of the main service sections (purple: provisioning, red: cultural, yellow: regulating/maintenance). The relative importance of these sections in characterizing each ecosystem service bundle is indicated by the percentages next to the bar charts, based on the absolute values of the ecosystem service indicators. The table on the right hand side provides an overview about the ecosystem service indicators used in the study; abbreviations of the ecosystem service category in brackets in the first column refer to: S—supply, P—potential and D—demand. (*Adapted from* Dittrich et al. [16]; *with permission*)

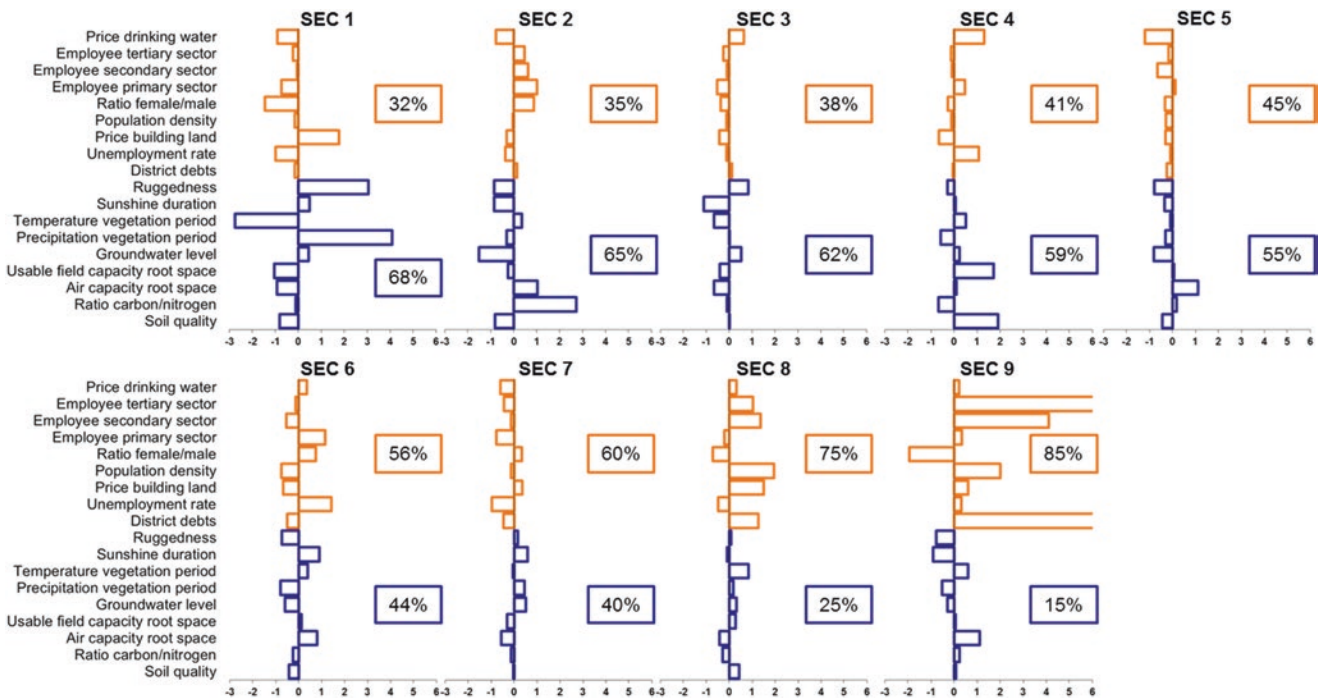


Fig. 43.3 Bar charts showing deviations of the standardized socio-economic (orange) and environmental (blue) covariates from the German national average (equal 0 at the x-axis) per socio-environmental cluster (SEC). The relative importance of these two sections in characterizing each socio-environmental cluster is indicated by the percentage next to the bar charts, based on the absolute values of the covariates. The bar chart of socio-environmental cluster 9 is cut off for the variables "Employee tertiary sector" and "District debts" as the values were more than 6 standard deviations over the national average and while aiming at a constant extend of the x-axis this would have decreased clarity of the remaining graphs. (*Adapted from* Dittrich et al. [16]; *with permission*)

to classify ecosystem service bundles as provisioning, regulating/maintenance, cultural, or mixed bundles. Likewise, we collated data on 18 covariates (see y-axis in Fig. 43.3) to delineate nine socio-environmental clusters (see Fig. 43.1b) again using the self-organizing maps technique. To characterize socio-environmental clusters as being characterized by mainly environmental or socio-economic conditions, we again summed up the absolute values of environmental vs. socio-economic covariates per socio-environmental cluster and thereby determined their relative importance (see Fig. 43.3). Finally, we used overlap analysis to characterise the relationship between the spatial configuration of ecosystem service bundles and the underlying environmental and socio-economic gradients.

43.3 Results and Discussion

43.3.1 Spatial Distribution and Characteristics of Ecosystem Service Bundles

We found eight bundles for Germany, which are characterized by pronounced differences in the type, amount (deviation from the national average), and combination of ecosystem services provided (see Fig. 43.2).

Ecosystem service bundle 1 (central loess plain around Harz mountain range; Fig. 43.1): This bundle is characterized by the most important agricultural production areas, while the potential for pollination and water-related recreation is very low (Fig. 43.2).

Ecosystem service bundle 2 (north-western lowlands): This bundle is the hotspot of provisioning services and depicts a region specialized in the production of energy plants and the raising of livestock, which at the same time suffers from low water quality.

Ecosystem service bundle 3 (low mountain ranges in Germany): These regions have a strong focus on wood production accompanied by a high potential for local landscape recreation. This regional concentration of specific provisioning services (bundles 1–3) likely reflects the ongoing specialization in land use, especially in agricultural production, which accelerated around 1970 [17].

Ecosystem service bundle 4 (German Alps and Black Forest): This region, which is dominated by cultural services, is very attractive for winter sports and hiking during summer. In addition, strong focus is given to wood production.

Ecosystem service bundle 5 (North Sea and Baltic Sea shorelines): This bundle is a hotspot for cultural services—or more precisely for distant recreation—and is nearly exclusively located at the shoreline of the North Sea and the Baltic Sea. In addition to the scenic beauty of the sea, large areas along the coastline have been designated as national parks and protected areas that provide infrastructure for nature appreciation and protection of resting places for migratory birds.

Ecosystem service bundle 6 (Alpine foothills and Bavarian low mountain ranges): These regions have a pronounced potential for pollination and are important for the raising of livestock. At the same time, the potential for landscape recreation is limited and the estimated soil loss due to water erosion is high.

Ecosystem service bundle 7 (Rhine, Elbe, Weser, and Oder Rivers): In these, the main rivers of Germany, nitrogen retention and flood regulation are particularly important, making this the only bundle dominated by regulating/maintenance services.

Ecosystem service bundle 8 (north-eastern lowlands; Fig. 43.1): This bundle has an equal share of all three ecosystem service sections, indicating multifunctional use of the landscape; while energy plants are widely cultivated and the potential for water recreation is exceptionally good, the potential for pollination and landscape recreation is intermediate (Fig. 43.2).

43.3.2 Characterisation of Ecosystem Service Bundles by Socio-Environmental Covariates

Ecosystem service bundles that are dominated by provisioning ecosystem services (ecosystem service bundles 1–3) spatially co-occurred mainly with socio-environmental clusters determined by environmental variables (Fig. 43.1). This indicates the persistent importance of local environmental conditions for the provisioning of these services, despite the technical progress during the last centuries. For ecosystem service bundles 4–6, which were classified as mostly cultural bundles, the two mainly associated socio-environmental clusters belong to both the environmental and socio-economic section, as the cultural services assessed require attractive environmental settings (e.g., sunshine duration) and at the same time likely affect local society (e.g., in terms of unemployment rate and price of building land). The ecosystem service bundle 8, characterised by multifunctionality, mainly overlapped with an intermediate socio-environmental cluster representing both socio-economic and environmental variables. The absence of pronounced environmental gradients may in this case have hindered a specialisation in certain provisioning services and, in turn, also prevented known trade-offs with regulating/maintenance services [18].

43.4 Conclusion and Outlook

The methods applied in this study represent a straightforward approach to assess ecosystem service bundles at a national scale, which can be repeated in other EU member states to support reporting requirements for the EU Biodiversity Strategy. The conducted spatially explicit analysis of ecosystem service associations can support strategic planning and prioritization of environmental issues at the European level as it helps to understand which ecosystem services are most important for which region. Further, regional responsibilities for certain services, as well as specific environmental issues (e.g., low water quality), may be identified while taking into account differences in the environmental and societal settings. The presented framework allows regular mapping and analysis of ecosystem services to track changes in time. We plan to also integrate biodiversity indicators in our future research to better understand linkages between ecosystem services potential/supply and biodiversity patterns.

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44 Indicators of Ecosystem Services for Policy Makers in the Netherlands

Bart de Knegt

44.1 Introduction

44.1.1 Ecosystem Services in the Netherlands

Nature supplies many useful goods and services to our society. These goods and services can be categorised into the capacity of ecosystems to (1) supply goods; (2) regulate processes; and (3) supply cultural services [1]. We made an infographic with a schematic overview of the Dutch ecosystems and expressed with pictograms the ecosystem services they provide (Fig. 44.1). The infographic creates awareness on the array of ecosystem services Dutch ecosystems provide, and was adopted by many other institutions within and outside the Netherlands.

44.1.2 Policies Are Setting Goals for Natural Capital

There is still insufficient awareness of the current state of ecosystems, the trends they demonstrate, and the benefits they provide to our society. In the European Biodiversity Strategy, targets are formulated on the sustainable use of natural capital [2]. Methods and tools are developed to help member states to assess the status of ecosystems and the services they provide [2]. The Dutch government has established comparable goals [3]. The Dutch Government wants to survey all Dutch ecosystem services by 2020, assign each one a place in the economic system, and incorporate them into the decision-making processes of the government, industry, and other stakeholders. Furthermore, the Dutch government wants to preserve and use natural capital in a sustainable manner by 2020.

A first step in the process of incorporating natural capital in decision-making is to assess the current status and the historic trend of ecosystem services in the Netherlands. Therefore, we developed an indicator that provides information to Dutch policymakers on the national level regarding the current state and historic trend of supply, demand, and deficit of seventeen ecosystem services, categorised according to the Common International Classification of Ecosystem Services [1]. State of the art knowledge in the form of maps, models, statistics, and expert judgement was used to construct these indicators and establish its uncertainties [4].

Which ecosystem services are addressed? Seventeen ecosystem services. Provisioning services: food, non-drinking water, drinking water, wood, fuel. Regulating services: soil fertility, erosion prevention, water retention, coastal protection, climate control cities, water purification, pest control, pollination, carbon sequestration. Cultural services: outdoor recreation, natural heritage, symbolic value of nature

What is the research question addressed? What is the current status and the historic trend of ecosystem services in the Netherlands?

Which method has been applied? Mapping, literature research, expert judgement

What is the main result? Status and trends of seventeen ecosystem services

What is concluded, recommended? Key message is that for many ecosystem services part of the demand remains unmet despite the use of technological alternatives and imports to meet the current demand. Trends show an increased mismatch in the supply and demand of ecosystem services. The results imply that the Dutch policy goals to use natural capital in a sustainable way are not yet met

44.2 Results

44.2.1 Increased Mismatch in Supply and Demand

The presented indicator shows an increase of the mismatch between the supply and demand of most ecosystem services over the last 20–25 years; the supply of services provided by

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Fig. 44.1 Examples of goods and services provided by Dutch ecosystems

Dutch ecosystems is declining or has stabilized, while their demand is growing (Fig. 44.2). None of the studied ecosystem services in Dutch ecosystems are sufficient to meet the country's entire current demand.

Climate change is found to be a major cause of the growing demand for ecosystem services such as water retention to prevent flooding, coastal protection, climate control in cities, carbon sequestration, and erosion prevention. The demand for the prevention of erosion has also increased as a result of the further intensification of agriculture. At the same time, population growth and altered consumption patterns have increased the demand for food. The demand for outdoor recreation has also increased as the population has grown and aged (the ageing of the population results in people having more leisure time). The demand for water purification has dropped because of the increased capacity of waste water treatment plants. The

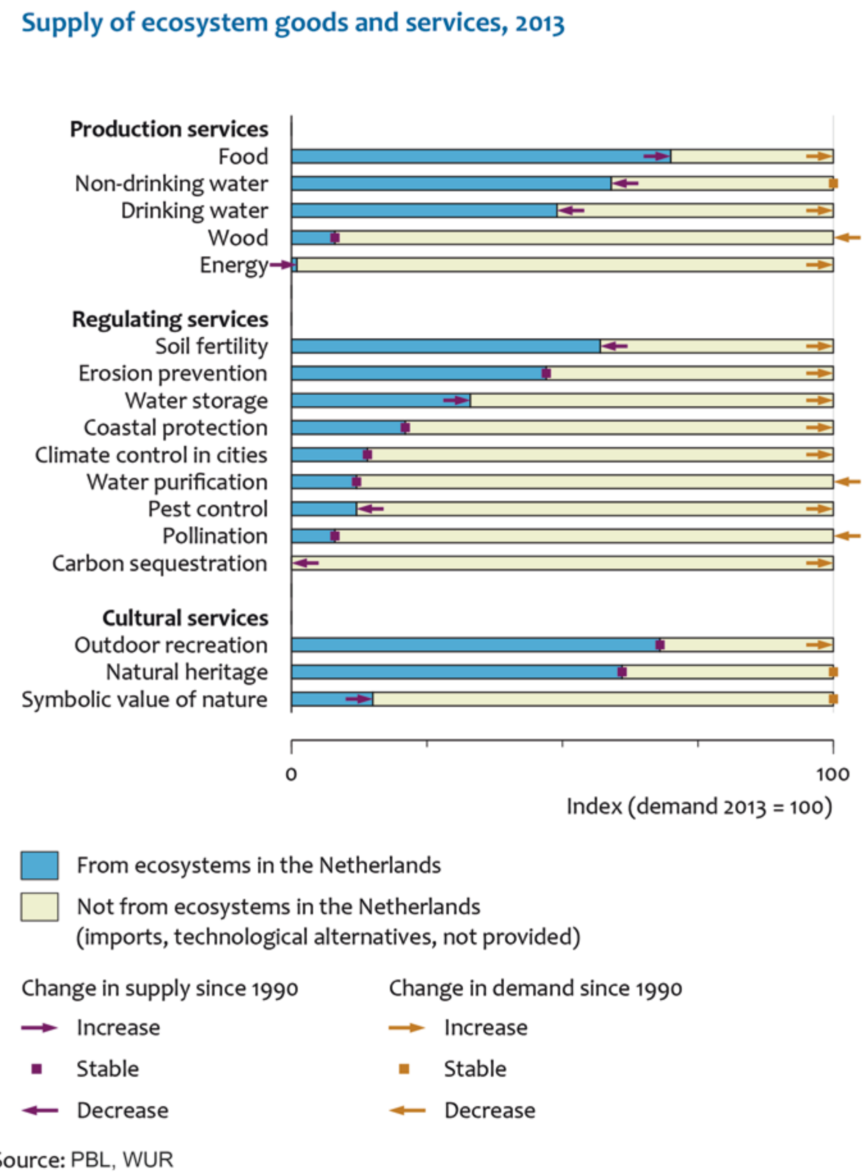

Fig. 44.2 For none of these types of service is the entire Dutch demand being met by Dutch ecosystems. For some services, the supply is declining since 1990–1995. In addition, the demand for several services is increasing faster than the supply from ecosystems

demand for pollination decreased because the surface area of fruit crops that are dependent on pollinators decreased.

The supply of goods and services has increased in the category of provisioning services for food and energy, whereas it has declined for drinking water and non-drinking water (used for washing, farm irrigation, and industry). Decreases are also seen for regulating services: soil fertility, carbon sequestration, and pest control. The decreases are partly related to the intensification of agriculture.

44.2.2 Services Can Be Supplied by Imports or Technological Alternatives

Despite the use of technological alternatives and imports to meet the current demand, part of the demand upon many ecosystem services remains unmet. The demand for the services mentioned here can also be met by imports or the use of technological alternatives (Fig. 44.3). Food, wood, and biomass for power generation are goods that can be trans-

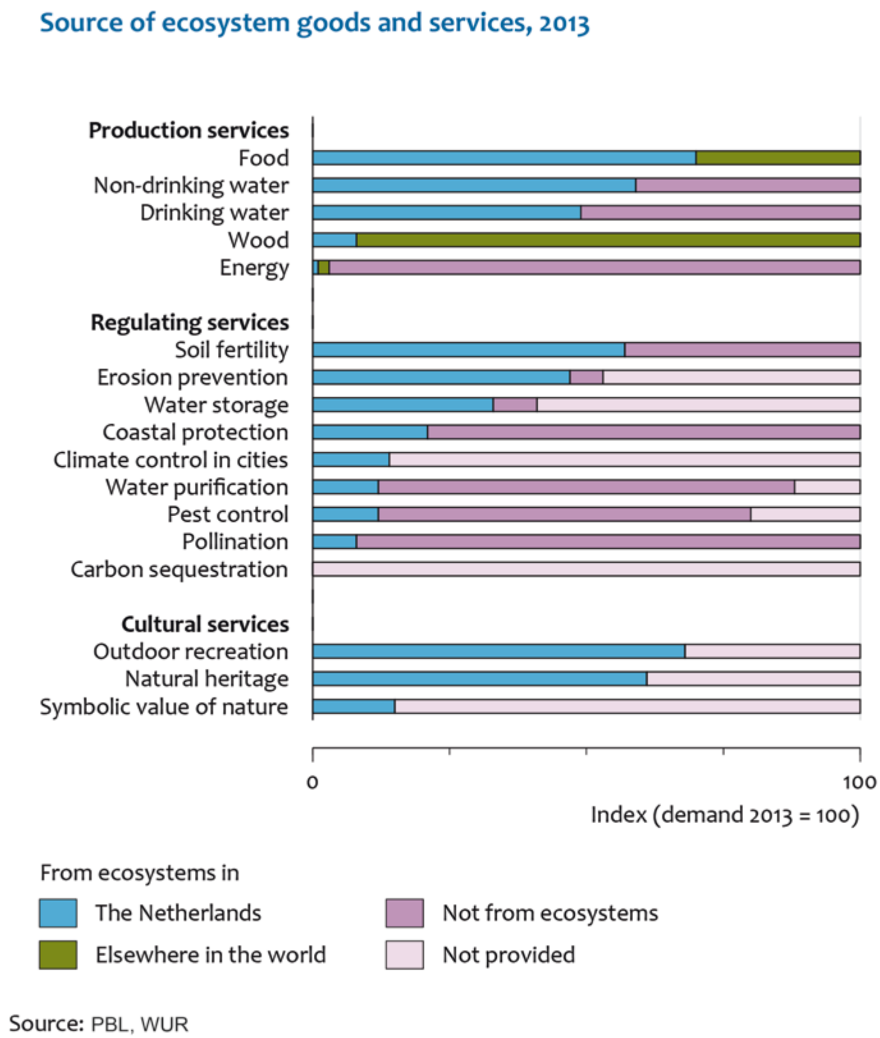

Fig. 44.3 The supply of services by Dutch ecosystems is supplemented by imports from ecosystems in other countries or by using technological solutions, such as dikes or chemical pesticides. Some types of demand remain unmet

ported and are being imported to meet the Dutch national demand. About 30% of food for human consumption in the Netherlands is imported, as are about 90% of the wood and approximately 75% of the biomass for power generation. Importation is usually not possible for regulating and cultural services; they need to be supplied at the location where the demand for services occurs.

The supply of various regulating services can also be maintained by technological means. Examples include dikes, which can replace dunes in protecting the coast, pesticides for pest control (replacing natural enemies), and domesticated honeybees to pollinate crops (replacing wild pollinators). If imports or technology are insufficient as alternatives for ecosystems services, part of the demand for such services will remain unmet. This is especially the case for regulating and cultural services.

44.2.3 Natural Areas Contribute Greatly to Ecosystem Service Provision

Natural areas, farmlands, and urban areas differ in the amount they contribute to the total supply of ecosystem services. Natural areas provide the largest share and the greatest diversity of ecosystem services (Fig. 44.4), even though the surface area of natural areas is many times smaller than that covered by farmlands. Present-day farmland is relatively mono-functional and provides only a few ecosystem

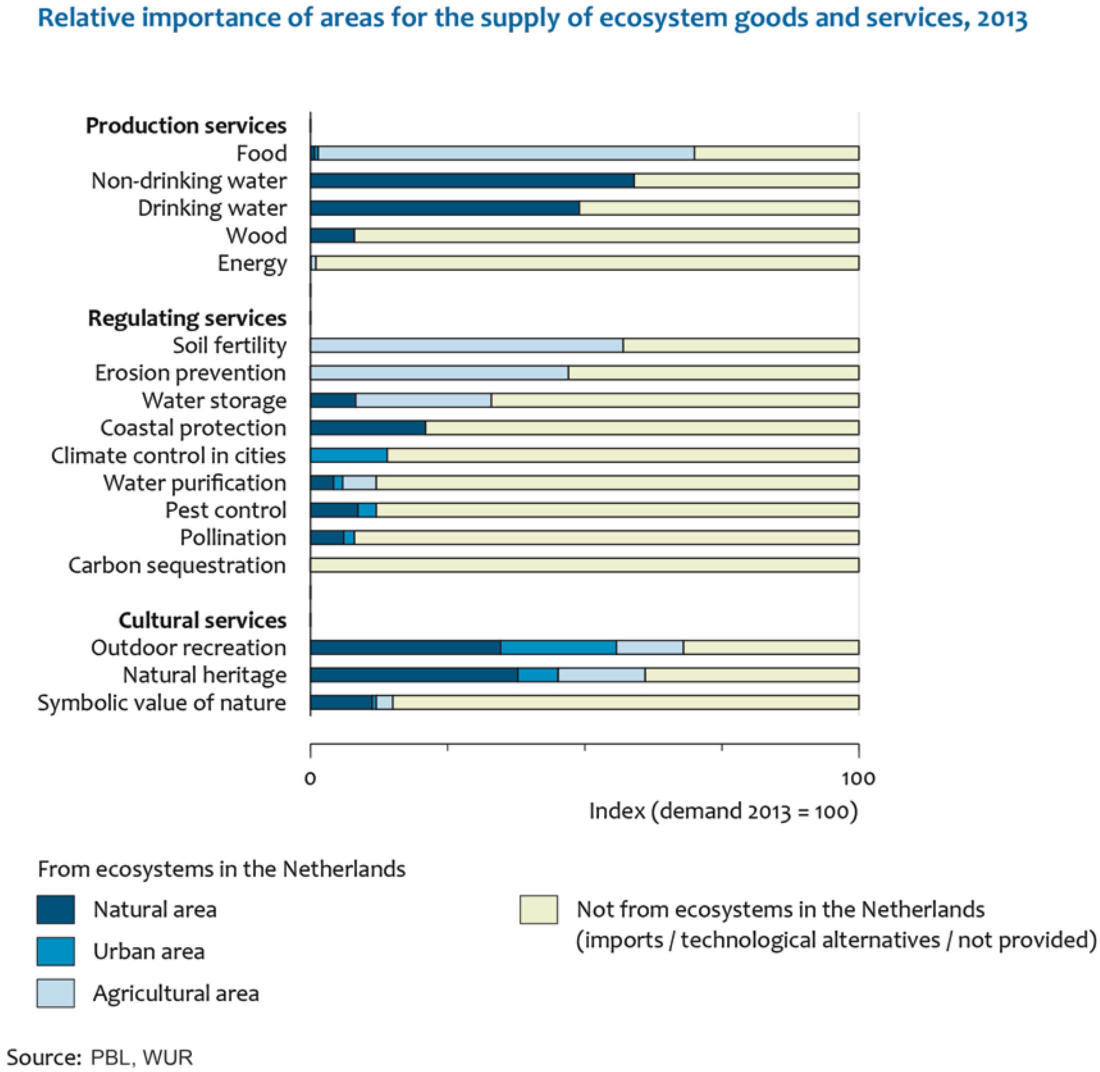

Fig. 44.4 Different types of areas differ in their contributions to the supply of ecosystem services in the Netherlands. Natural areas offer the widest range and the largest share of most ecosystem services, while urban areas hardly contribute to the total supply of such services

services. Urban areas have the smallest share in the total supply of ecosystem services in the Netherlands.

44.3 Discussion and Recommendations for the Way Forward

The ecosystems service indicators described here have been identified as essential for monitoring ecosystem services in the Netherlands [5]. The key message is that for many ecosystem services, part of the demand remains unmet despite the use of technological alternatives and imports to meet the current demand. Trends show an increased mismatch in the supply and demand of ecosystem services.

The results imply that the Dutch policy goals to use natural capital in a sustainable way are not yet met. This can affect human well-being negatively, especially in situations where society relies heavily on ecosystem services. This is the case when imports are impossible (regulating and cultural ecosystem services) or where technological alternatives entail higher costs. Technological alternatives may also have unfavourable side-effects (e.g., pesticides decrease ecological quality) or practices are not sustainable in the long run (e.g., use of fossil fuels). Importing goods from abroad imposes effects on the natural capital outside the Netherlands (international ecological footprint). Imports also increase dependence on ecosystems elsewhere, in the context of global trends that include a growing population and increasing food demands. This leads to increased competition between world regions. Demands that remain unmet it will affect negatively human well-being. For instance, in the case of water retention, it means that areas will suffer from water stress or desiccation. In the case of carbon sequestration, it will lead to higher CO_2 concentrations in the atmosphere, causing further global warming. For natural heritage, it implies that more species will be threatened with extinction.

Implementing more nature-based solutions may play a vital role in counteracting the above-mentioned issues. These solutions are especially effective if they are implemented in a holistic approach that takes into account bundles of services and can be applied to multiple actors and across spatial and temporal scales. The question at hand is, What are the options to increase the sustainable use of natural capital to avert a loss in human well-being? To answer this question, we need to develop other indicators and tools. For example, in Chap. 56 (de Knegt et al.), the information provided by these indicators was used to make opportunity maps and to subsequently formulate options for policymakers and other relevant stakeholders to use our natural capital more sustainably. In order to seize these opportunities, supply and demand could be spatially optimized, demand could be decreased, and/or the supply could be increased. The supply could be increased by increasing the area of natural ecosystems or increasing the production capacity of existing ecosystems by, for instance, by decreasing limiting environmental pressures. Multifunctional agricultural landscapes especially seem to offer chances for a more sustainable use of natural capital in the Netherlands.

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45 The Montérégie Connection: Understanding How Ecosystems Can Provide Resilience to the Risk of Ecosystem Service Change

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45.1 Introduction

Communities manage their landscape and the biodiversity within it to provide various ecosystem services (ES) such as high-quality water, food, and recreation. Provision of these services is a sensitive indicator of ecosystem health, and is also critical to human well-being. To successfully manage both ES and biodiversity, communities need reliable, objective information and effective tools to evaluate how human activities and social-ecological dynamics will alter the landscape structure that affects the maintenance of biodiversity and ES provision.

The Montérégie Connection project aimed to help the community of this agricultural region just southeast of Montreal to improve management of multiple ecosystem services by developing and empirically testing a modeling framework that quantitatively linked landscape connectivity, biodiversity, and ecosystem services in this region. We will use this framework to build scenarios and other practical decision-support tools with communities to help them grapple with the challenges of environmental management in the face of local, regional, and global change. We focused especially on how forest connectivity and forest corridors might help the local landscape maintain biodiversity and provide the desired ES in the face of these changes.

The Montérégie Connection project study area is located to the southeast of Montreal, Canada (Fig. 45.1). It is in the Mixedwood Plains ecozone and St. Lawrence Lowlands ecoregion of southern Québec, Canada. This region has warm summers and cold, snowy winters with a mean annual temperature of ~5 °C and average seasonal temperatures that range from 16.5 °C in the summer to −7 °C in the winter. Mean annual precipitation varies between 800 and 1000 mm. The terrain is

Which ecosystem services are addressed? Provisioning: agricultural production (crops, pork), provision of clean water, maple syrup production, milk production

Regulating: flood control, regulation of nutrient cycling, carbon storage for climate regulation, soil organic matter, pollination

Cultural: hunting opportunities, nature appreciation, tourism, forest recreation

What is the research question addressed? The overarching questions that drove the research project were as follows:

(1) How can the Vallée-du-Richelieu Municipalité Régionale de Comté (VR-MRC) manage the local landscape to maintain biodiversity and provide desired ecosystem services (ES) in the face of regional and global change?
(2) How would development of a network of forest corridors linking natural areas in the VR-MRC affect current and future biodiversity and the provision of ES?
(3) How does the location and size of these corridors alter the provision of biodiversity and ES?

Which method has been applied? We used a variety of methods, including the use of existing government records to map ecosystem services at larger (municipal and county) scales, a broad set of fieldwork methods to measure ecosystem services directly at smaller scales, and a variety of modelling methods to link these field methods with government data to consider possible future provision of services

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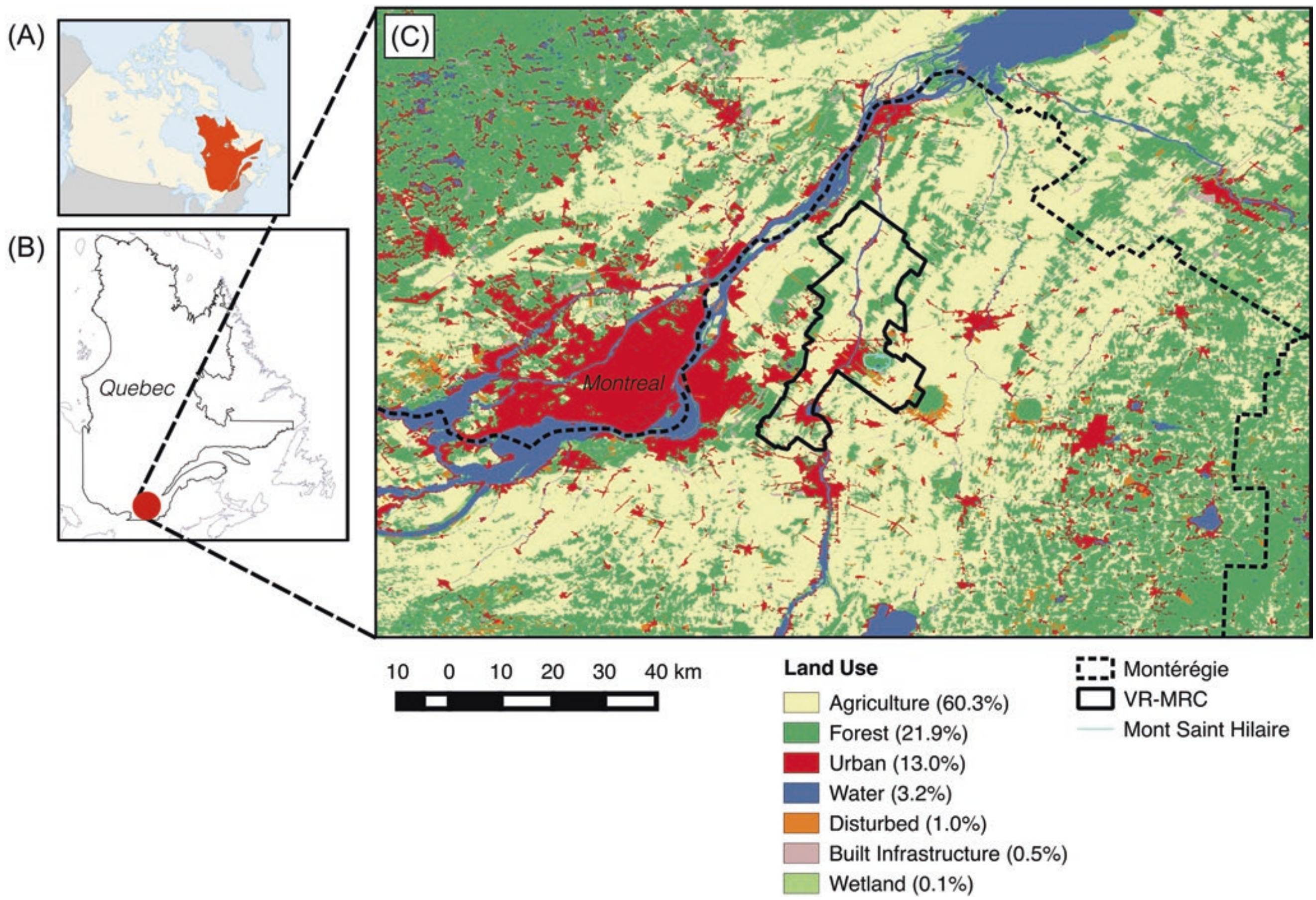

Fig. 45.1 Land use/land cover in the Montérégie and surrounding area. (**a**) The location of Quebec within Canada. (**b**) The location of the Montérégie in southern Quebec. (**c**) Southeastern Quebec, including Montreal and the Montérégie. The Montérégie Connection project fieldwork, models, and scenarios have focused on the Vallée du Richelieu Municipalité Regionale CComté (VR-MRC, outlined in solid black). Proportions of land use within the VR-MRCC are shown in the legend. (*Reproduced from* Mitchell et al. [4])

quite flat and, below 150 m, dominated by poorly drained clay deposits upon which gleysolic soils have developed [1]. In this generally flat milieu, several Montérégian hills, isolated plutonic intrusions that range from ~200–400 m in height, are also present [2]. These hills are forested, often reserved and highly valued for recreation, and are important reservoirs of plant and animal biodiversity in the region.

The administrative unit of the Montérégie is ~11,000 km^2 and has a population of approximately 1.4 million people (~18% of Québec's population). It contains ecosystems with the highest levels of biodiversity in the province [3] and consists of a mosaic of urban, periurban, rural, and agricultural areas (Fig. 45.1) [4]. Rural communities and agriculture have historically dominated this landscape, but the type of agriculture has changed through time. A shift from dairy farming to intensively managed corn and soybean fields has occurred in recent decades, leading to more annual crops, fewer farms, and increases in pesticide and fertilizer use [1, 5]. Apple orchards are concentrated on the well-drained gravel slopes of the Montérégian hills. Numerous, mostly small, residual deciduous and mixedwood forest fragments are present and provide important ES, including maple syrup production, but very few municipalities have more than 30% forest cover. Because of its proximity to Montreal, the region is currently undergoing significant residential development and expansion of periurban areas [6], which is causing significant loss of bio-

What is the main result? Ecosystem service provision is the outcome of a complex set of interacting social and ecological factors that are difficult to understand and manage. Understanding basic principles and working to anticipate the unexpected can help societies move forward protecting the long-term security of the many ecosystem services they desire

What is concluded, recommended? Working with local stakeholders in active listening can improve research and its ultimate uptake in policy- and decision-making diversity [1, 5] and is putting pressure on local land managers to achieve provision of multiple ES across the landscape.

The intensity of human use in this region—and demand for ES—requires effective management to maintain the provision of multiple ES into the future. Because land use and management are key drivers in this system, at issue is the need to move from landscape management that focuses on one service at a time (typically, for this region, either food production, recreation, or biodiversity) to a management that focuses on maintaining multi-functional landscapes. This type of decision-making can be improved through development of theory, data, and models that link landscapes, their biodiversity and functions, and ecosystem services at the scales at which decisions are made [7]. Our overarching goal as a project was to work with the community to facilitate improved decision-making by discussing theory of, collecting data about, and developing models on the linkages between landscapes, biodiversity, and ecosystem services.

45.2 Current State of the Montérégie

Many ecosystem services are provided across this region, including food, water, opportunities for recreation and tourism, carbon sequestration, pollination, pest regulation, water quality regulation, hunting, maple syrup production, and aesthetic and spiritual connections to nature [8–11]. These services are provided across the landscape in a variety of unique patterns (Fig. 45.2) which, in turn, are driven by a combination of factors, including where it is possible to produce services, biophysically and ecologically; human desire for services, and interactions between services [8].

The strongly linked spatial distributions of multiple ecosystem services translates into an emergent pattern of municipalities with similar sets of ecosystem services, which we call *bundles*. Different bundles exist on the landscape in relation to the social-ecological system (Fig. 45.3) [8]; for example, municipalities on the landscape that are known to be destinations for cottagers were grouped together in what we call the Country Homes bundle type, which has high provision of forest recreation, carbon sequestration, high-quality water, phosphorus retention, and soil organic matter. Other municipalities known for agriculture might be grouped in either the Corn-Soy Ag bundle (high soil P retention and crop production, along with good water quality) or the Feedlot Ag bundle (low water quality and low provision of most services besides pork and crops).

The locations of provision of these services, and the services themselves, have shifted over time [12]. At the earliest dates for which we could obtain data on multiple ecosystem services, most municipalities provided a broad mix of services (e.g., providing crops, animal products, and flood control). Through time, most municipalities began to specialize in provision of one or two services (e.g., on recreation alone). Over the entire region, provision of most services remained, and even increased, but this increase was spatially specialized by municipality (Fig. 45.4) [12].

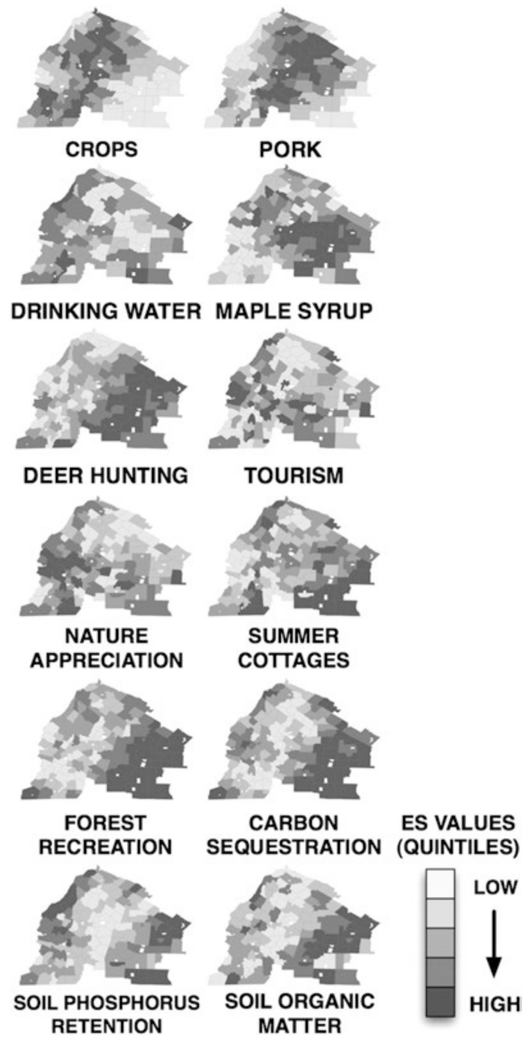

Fig. 45.2 Distributions of 12 ecosystem services, shown in quintiles. The gradient of light-to-dark grey corresponds with low-to-high values of ecosystem service provision. The area of this map corresponds to the dashed-line outline of the administrative boundary of the Montérégie in Fig. 45.1, but extends slightly further to the east to incorporate two entire watersheds. (*Reproduced from* Raudsepp-Hearne et al. [8])

Correlation analysis also revealed that the interactions among some services changed [12]. For example, some relationships, such as that between carbon storage and hunting, increased in strength through time, and others (carbon storage and other recreational activities) became weaker. Other relationships changed entirely, such as the relationship between hunting and livestock production—negative at the start of the study, and positive by the end of it. While we do not know for sure the causes of these changes, hypotheses can easily be developed. For example, it may be that hunting and livestock production were negatively correlated in the early part of the study because animals were kept in pastures and competed with deer for food. By the end of our study,

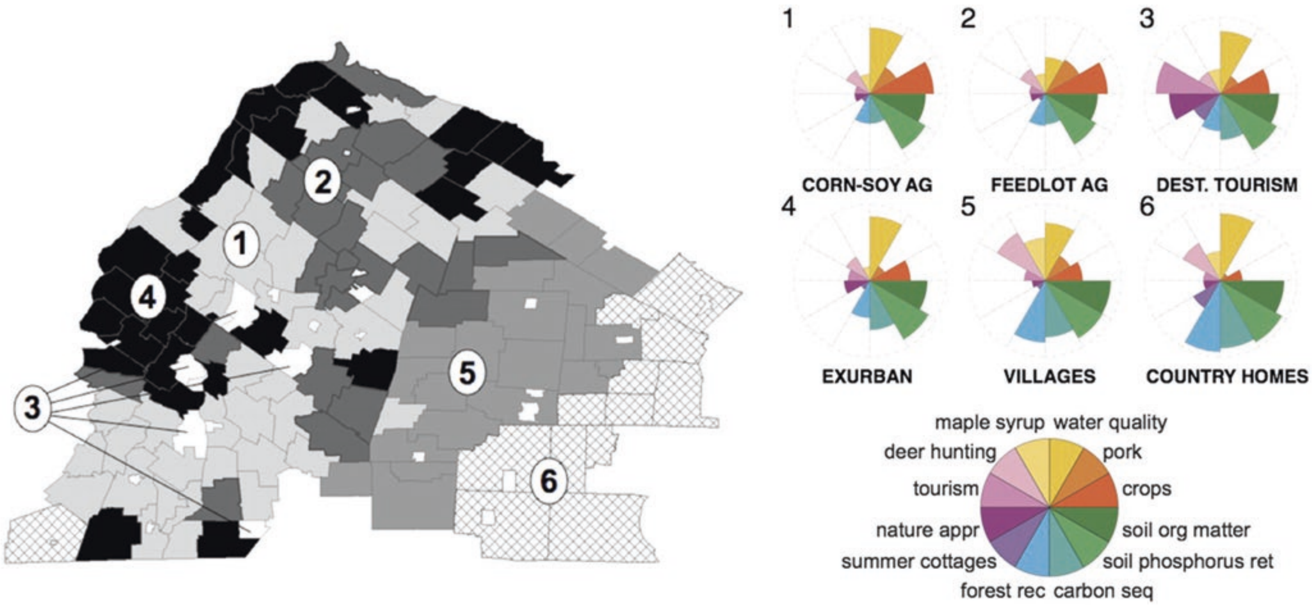

Fig. 45.3 Ecosystem service bundle types represent the average values of ecosystem services found within each cluster. Clusters in the data were also found to be clustered in space, and each ecosystem service bundle type maps onto an area of the region characterized by distinct social-ecological dynamics, represented by the bundle names. The area depicted is the same as in Fig. 45.2. (*Reproduced from* Raudsepp-Hearne et al. [8])

most animals were kept indoors or in feedlots, reducing any competition with deer for food.

In addition to these large-scale studies of ecosystem service provision that relied primarily on existing data and historical records, we also undertook finer-scale studies, in which we collected primary data on ecosystem services along a series of high resolution transects. Here, our goal was to investigate the role of forest connectivity on the provision of various services in both agricultural and forest settings. Thus, these transects were situated in agricultural and forested settings near forest patches of a variety of sizes and levels of connectivity to other forest patches. We found that landscape structure has an important role—especially forest fragment connectivity—in a number of ecosystem services.

In particular, increased forest connectivity reduced insect herbivory and aphid numbers in nearby soybean fields (Fig. 45.5a, b) [13] as well as arthropod pest control within maple tree stands (Fig. 45.5b) [13], but had little effect on aboveground carbon storage [9]. The relative locations of riparian buffers and nutrient sources to agricultural fields and watercourses also play a critical role in water quality regulation in the region [14]. We found that ES provision varies according to distance-dependent relationships within single land-use categories. For example, soybean yield increases asymptotically with distance from forest (Fig. 45.5c), whereas seed set in apple orchards declines linearly as distance to meadow increases (Fig. 45.5d) [11]. It is important to note that the nature of the relationships between landscape structure and ES provision varies widely across different ES. This is significant because it means that the consequences of a single change in landscape structure will vary substantially for different ES, significantly increasing the difficulty for managers who wish to manage for multiple services (Fig. 45.5) [4].

45.3 Potential Futures

In addition to exploring current provision across the region through existing data, and collecting data on fine resolution provision of services, we also worked with local stakeholders to develop four scenarios about the potential future of the region. These scenarios were not intended to be predictions, but to encourage thinking about the long-term future, to anticipate that surprises of some sort would likely happen, and to understand the important connections between parts of the system [14].

Such scenario planning is useful in conditions of high uncertainty, and in complex social-ecological systems such as the Montérégie [15].

The four scenarios developed by local stakeholders explore urban sprawl, the effects of changing demands for energy (and of different means to meet those demands), of economic crisis, ecological crisis (insect pests), and increased interest in green development on the set of ecosystem services provided in the region (Fig. 45.6). We explored these scenarios qualitatively and quantitatively through prototype models developed using data collected in other parts of the study.

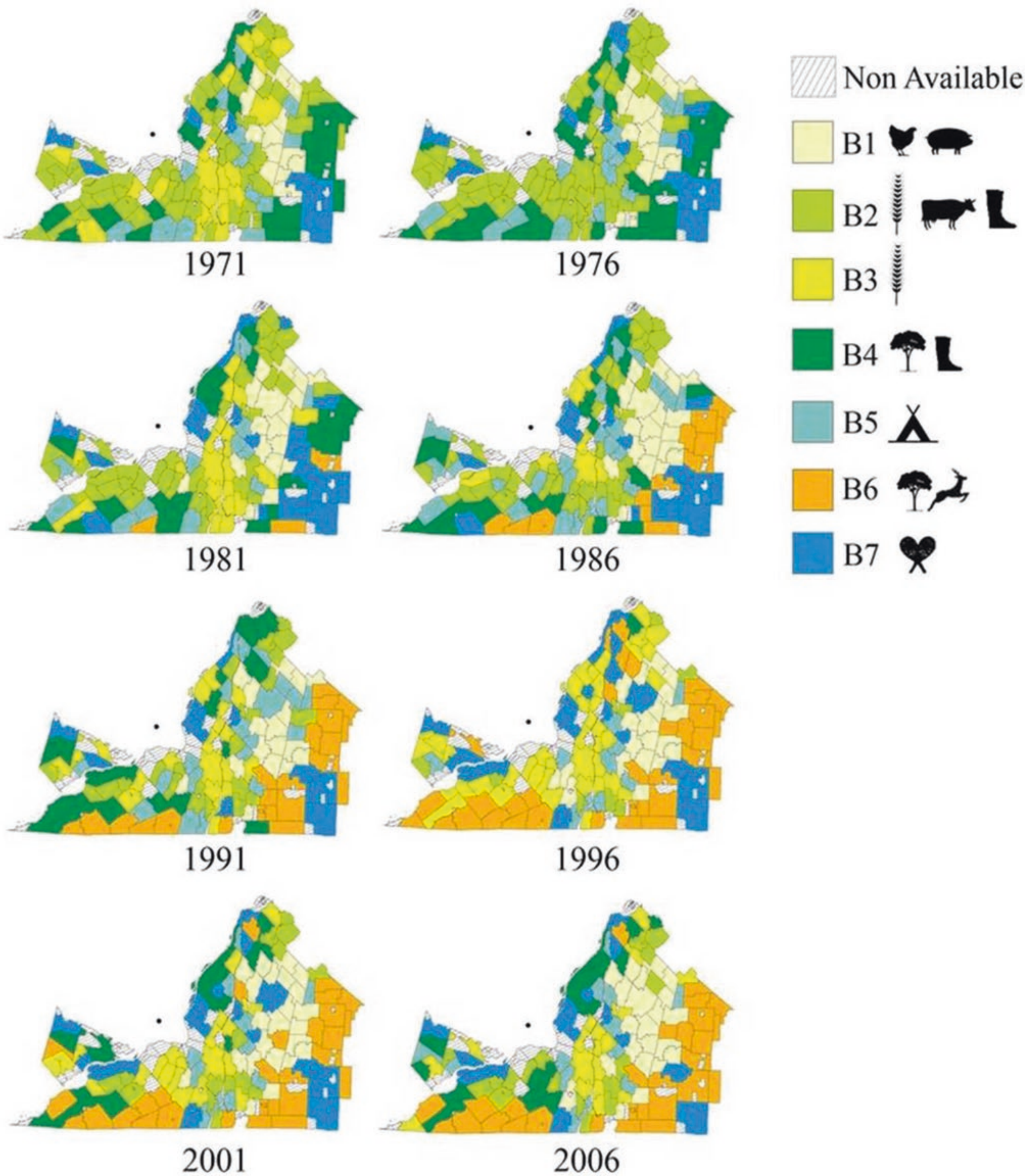

Fig. 45.4 ES bundle dynamics across space at each time step in the Montérégie. This map largely overlaps with the area depicted in the dashed line in Fig. 45.1, but extends slightly further to the west. The small dot in the middle shows the location of Montreal. Bundle 1 (B1) consists primarily of farm animals. Bundle 2 (B2) includes crops, milk production, and flood control. Bundle 3 (B3) consists primarily of crop production. Bundle 4 (B4) includes carbon storage and flood control. Bundle 5 (B5) is campsites, alongside some food production. Bundle 6 (B6) is carbon storage and game animals. Bundle 7 (B7) is recreational activities. (*Reproduced from* Renard et al. [12])

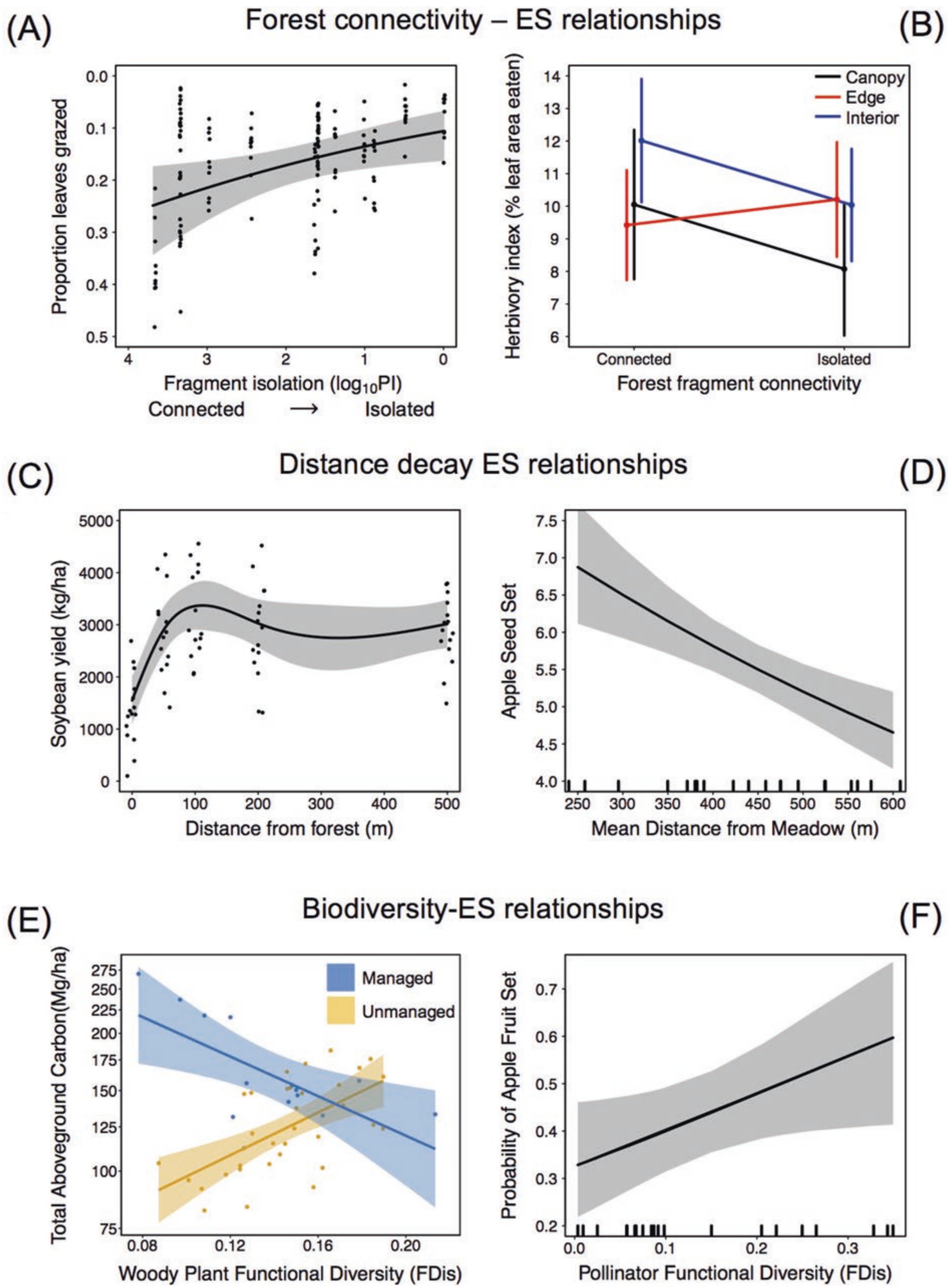

Fig. 45.5 Selected results from the Montérégie Connection project. (**a**) Relationships between Forest fragment isolation (PI indicates proximity index) and herbivory regulation in adjacent soybean fields; (**b**) Forest fragment connectivity and herbivory regulation in different types of maple tree stands; (**c**) Distance from forest fragment and crop yield; (**d**) Distance from meadow and pollination services; (**e**) Woody plant functional diversity (functional dispersion) and carbon storage; and (**f**) Native pollinator functional diversity (functional dispersion) and pollination services. In each, shaded areas or error bars indicate 95% confidence intervals. In panels (**a**), (**c**), and (**e**), we show individual data points; in panels (**d**) and (**f**), the small lines along the x-axis indicate sampled distances from the meadow and pollinator functional diversity, respectively. Reproduced from [11] (panels **a** and **c**); [12] (panels **d** and **f**) and [9] (panel **e**). (*Reproduced from* Mitchell et al. [4])

Fig. 45.6 Artist's depiction of the four VR-MRC scenarios. All illustrations are used with the permission of the artist, Denis Bainville. (**a**) Periurban Development scenario, in which residential growth drives loss of farmland and forests, resulting in loss of landscape connectivity and agricultural production in certain areas. (**b**) Demand for Energy scenario, in which shale gas development expands, resulting in farmland loss, while urban and residential development is limited. Wind power installations are also developed. (**c**) Systemic Crisis scenario, in which an economic crisis drives residential densification, while the Asian Long-Horned Beetle (*Anoplophora glabripennis*) invades, decimating the maple tree populations. Agricultural production shifts to pasture-based livestock, while marginal farmlands are converted to agro-forestry. (**d**) Green Development scenario, in which there is a shift toward sustainable development, with renewable energy, green corridors, protected areas, and agriculture that incorporates principles of agroforestry and agroecology

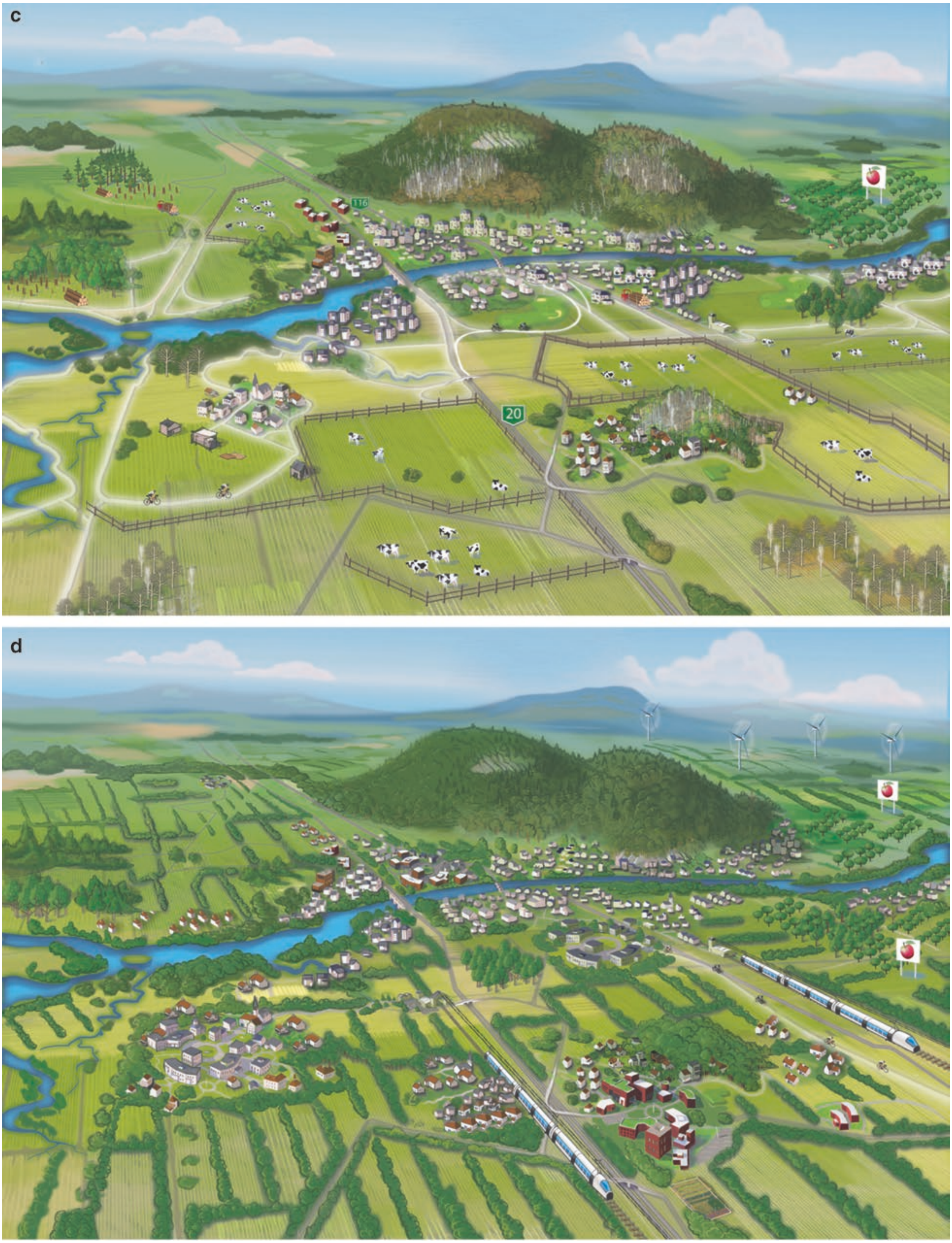

Fig. 45.6 (continued)

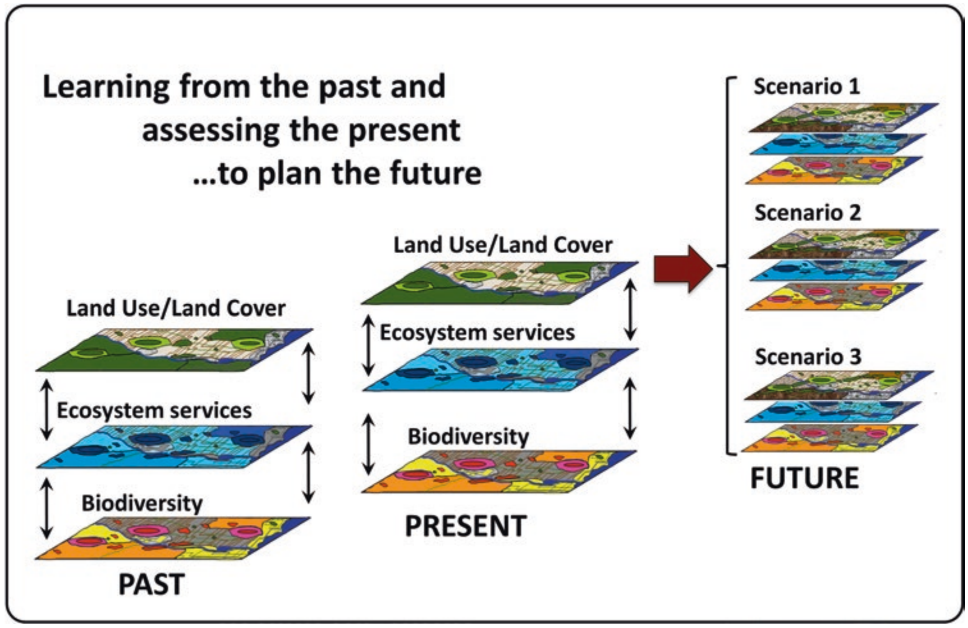

Fig. 45.7 The provision of ecosystem services depends on relationships between land use/cover biodiversity, and services. We investigated these relationships in the present, past, and future through a combination of existing data, new data collection, and future scenarios development

45.4 Conclusions

The project was designed to develop approaches, data, and tools that would be useful for ensuring management of healthy ecosystems in settled landscapes across Quebec and Canada. At a variety of scales, using a variety of methods, and through analysis of past, present, and future, we investigated the relationships between land use/cover, biodiversity, and ecosystem services (Fig. 45.7). We learned that the relationships between biodiversity, services, and land use/cover are highly variable among services and through time. The framework and our new understanding are a significant advance significant advance in our understanding of how human actions alter ES provision at landscape scales across social-ecological systems. Our use of scenarios, and focus on working with stakeholders to develop a true collaboration in all stages of the project, are also advances, as the ultimate value of projects like this are not just the development of new science but our ability to translate the new knowledge into tools that managers, policy makers, and other actors can comprehend and use to make better decisions.

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46 Synchronized Peak Rate Years of Global Resources Use Imply Critical Trade-Offs in Appropriation of Natural Resources and Ecosystem Services

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46.1 Introduction

Four decades ago, the limits to growth model of Donella and Dennis Meadows reignited the old Malthusian debate about the limits of the world's resources, many of are now denoted and understood as *ecosystem services* [1–4]. Limits to growth of specific resources such as oil [5] or fossil water [6] have been analysed separately, by estimating the peak (or maximum) rate year, defined as the year of maximum resource appropriation rate (Fig. 46.1). Exploring the relation among peak rate years for multiple resources or ecosystem services raises an important question: Are global peak rate years synchronized, i.e., occurring at approximately the same time in the long history of human civilization?

Twenty-seven non-renewable and renewable resources and ecosystem services essential for human well-being and daily needs (e.g., energy and food) were used for analysis [7]. These resources are also the focus of global policy bodies such as the United Nations and the World Bank. Non-renewables include the fossil fuels (coal, gas, oil) supplying 87% of the energy consumed by the 50 wealthiest nations [8]. Renewables include staple crops (cassava, maize, rice, soybeans, wheat) [9] that the UN Food and Agriculture Organization identifies as providing 45% of global caloric intake. Combined with data on the consumption of animal products, the main sources of food are included in the analysis. We also evaluate resources with a long history of use (crop land, domesticated species) and renewable energy sources that may be increasingly important in the future. Furthermore, we consider two global drivers of resource use: population and economic activity (world GDP). The raw data and smoothed times series of the bootstrap resample are plotted in Fig. 46.2 [7].

Which ecosystem services are addressed? Most relevant renewable food resources, i.e., provisioning ecosystem services from agriculture (crops, meat, dairy products) and aqua culture

What is the research question addressed? Is there a global limitation of continued ecosystem service provision?

Which method has been applied? Statistical analysis of long-term time series data

What is the main result? 18 of the 20 ecosystem services have passed the point in time at which a further increase of production per year is attained, i.e., production keeps increasing but its growth rates decline

What is concluded, recommended? The synchrony of peak rate years of multiple food resources poses a major adaptation challenge for society, suggesting the need for a paradigm shift in resource use towards a sustainable path

46.2 Results

In Table 46.1, we observe that for 21 of the 27 global resources and for the two global drivers of resource use, there was a peak rate year. For the 21 resources that had a peak rate year, all but one (cropland expansion) lay between 1960 and 2010 (Fig. 46.3). Given the extent of human history, this is a very narrow time window. The available data suggest that peak rate years for several non-renewable

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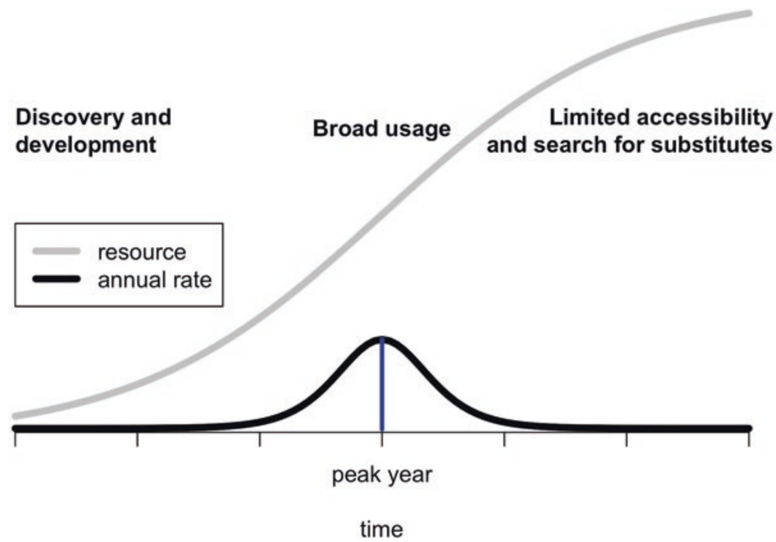

Fig. 46.1 Conceptually, the peak rate year of resource use is the point in time with the maximum appropriation rate (blue line). The appropriation of an individual resource proceeds through three phases. Initially, the resource is discovered and its use developed. Then, the resource is broadly used. Finally the resource is less accessible or scarce and substitutes are explored. The three phases may overlap over time. We analyse renewable resources (regeneration at human life scale) and non-renewable resources (regeneration on geologic time scale). For drivers of resource use, the peak rate year is the year of maximum growth rate. (*Reprinted from* Seppelt et al. [7]; *with permission*)

resources (coal, gas, oil, and phosphorus) has not yet occurred. This implies a continued acceleration of extraction, which is in accordance with earlier analysis for oil [5] and phosphorus [10].

Individual countries have detectable impacts on the global non-renewable resource extraction rate. For example, in 2011 the rate of coal extraction for China was 7.2% (5.7–7.4), while the rate for the world without China was 3.7% (3.5–3.8). The values for natural gas in 2011 were 10.1% (7.6–10.3) and 4.4% (4.0–4.4) with and without China, respectively. A peak rate year for renewable energy has not occurred.

Figure 46.3 shows that the peak rate of earth surface conversion to cropland occurred in 1950 (1920–1960), and the expansion of cropland recently stabilized at the highest recorded levels, about 1.8×10^6 ha [11]. We find peak rate years recently passed for many agricultural products: soybeans in 2009 (1977–2011), milk in 2004 (1982–2009), eggs in 1993 (1992–2006), caught fish in 1988 (1984–1999), and maize in 1985 (1983–2007). Two major factors of agricultural productivity, N-fertilizers and the area of irrigated land, show peak rate years in 1983 (1978–2010) and 1978 (1976–2003), respectively. Water is a resource that many world policy bodies are concerned with and is largely understood as a renewable resource. But not all water is renewable. "Fossil water" stocks are isolated water resources that are consumed faster than are naturally renewed. There is currently a lack of time-series data at the global scale on the status of hydrological resources [12]. As an example of national trends, the greatest rate of groundwater extraction occurred in 1975 in the USA (1975–2005). Water conservation and rationing rules likely reduced the rate of groundwater extraction [6]. For maize, rice, wheat, and soybeans, the yield per area is stagnating or collapsing in 24–39% of the world's growing areas [13], which may explain why the peak rate years have passed at a global level. The peak rate years of renewable resources collectively suggest challenges to achieving global food security [14]. The pattern of peak rate years occurring in land and food, and not yet occurring for non-renewable resources, suggests that sustained acceleration of agricultural production is not limited by energy.

Following the observation of an apparently simultaneous pattern of peak rate years in Fig. 46.3, we tested the hypothesis of synchrony among peak rate years on 20 statistically independent time series of resources, of which 16 present a peak rate year. We find that peak rate years appear clustered around 2006 (1989–2008), given the uncertainty surrounding the peak rate year estimate of each resource (Fig. 46.4, [2]). It is unlikely that the synchrony is a statistical artefact because there is less than a 1 in 1000 chance that the distribution would have been obtained if it were sampled from a uniform distribution (i.e., null hypothesis of no synchrony is rejected). This pattern of synchronous peak rate years may have emerged from a mix of resource depletion, ongoing innovation or substitution, and changing tastes.

We find that the global population reached its highest growth rate in 1989 (1988–1989), a value within the boundaries of previous findings [15], and the peak rate year for global GDP growth was 2010 (2004–2012). In spite of a decelerating population growth, the rate of resource appropriation is not expected to decline. Indeed, the land area used for urban

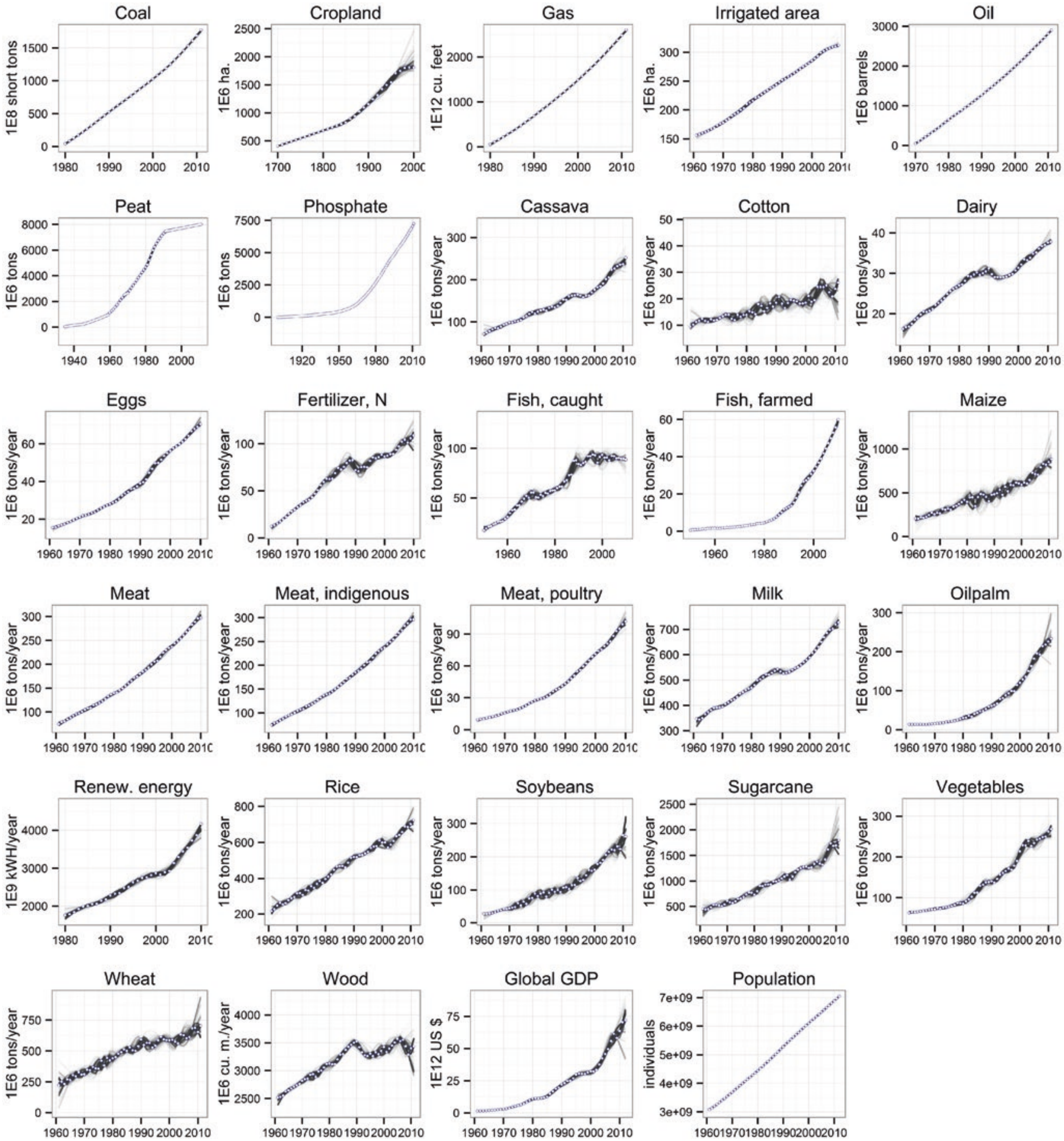

Fig. 46.2 For the global resources and drivers, the raw data is plotted as blue points (the non-renewable resources are accumulated over time for growth-rate analysis and for the renewable resources the yearly production is analysed). The grey to black lines each represent one smoothed time series of the bootstrap resamples (5000). A wider set of lines implies larger uncertainty when estimating peak rate year (e.g., compare peat vs. wheat). The uncertainty can be so small that the blue points hide the smoothed time series (e.g., population). There are local minima and maxima which increase the uncertainty (e.g., wood). (*Reprinted from* Seppelt et al. [7]; *with permission*)

settlements and household numbers continue to increase [16], which puts additional pressure on resources. However, we identified peak rate years in household intensity (number of households per 100 people) for a number of countries (mostly developed countries) suggesting a decreasing rate of consumption of space for living in those countries. In addition to the aggregated human demand for resources, the consumption pattern is important in influencing peak rate years. For example, the peak rate of meat consumption in the USA occurred in 1955 (1909–1999).

Table 46.1 Lists of results for all resources and drivers[a]

	Peak rate year				Rate at peak			
	2.5th	50th	97.5th	Peak?	2.5th	50th	97.5th	Independence status[b]
	Global non-renewable resources							
Coal	2008	2011	2011	No peak	4.2	4.7	4.9	Yes
Cropland	1920	1950	1960	Peak	0.5	0.6	0.9	Yes
Gas	2007	2011	2011	No peak	4.1	4.5	4.6	Yes
Irrigated area	1976	1978	2003	Peak	1.5	1.8	2.3	Yes
Oil	2006	2011	2011	No peak	2.9	3.1	3.1	Yes
Peat	1982	1983	1983	Peak	4.1	5.0	5.2	No
Phosphate	1988	2011	2011	No peak	2.3	2.8	3.1	No
	Global renewable resources							
Cassava[c]	2004	2006	2011	Peak	3.8	6.4	7.7	Yes
Cotton[c]	1983	2004	2011	Peak	9.6	14.8	19.2	No
Dairy	1964	1989	2004	Peak	2.5	3.3	4.0	No
Eggs	1992	1993	2006	Peak	3.2	4.4	5.5	Yes
Fertilizer, N[c]	1978	1983	2010	Peak	4.7	6.2	7.0	Yes
Fish, caught	1984	1988	1999	Peak	5.2	9.7	14.9	Yes
Fish, farmed	1994	2010	2010	No peak	5.1	6.1	7.7	No
Maize	1983	1985	2007	Peak	6.8	10.5	14.0	Yes
Meat	1996	1996	2009	Peak	2.5	3.1	3.4	Yes
Meat, indigenous	1996	1996	2009	Peak	2.6	3.1	3.3	No
Meat, poultry	2005	2006	2009	Peak	3.6	4.8	5.5	Yes
Milk	1982	2004	2009	Peak	2.4	2.7	2.8	Yes
Oilpalm	2003	2005	2008	Peak	6.2	8.0	10.1	Yes
Renew. energy	2004	2010	2010	No peak	3.8	5.7	8.3	Yes
Rice	1973	1988	2008	Peak	3.6	4.4	5.0	Yes
Soybeans[c]	1977	2009	2011	Peak	5.9	8.9	21.4	Yes
Sugarcane[c]	1981	2007	2011	Peak	5.9	9.8	10.7	Yes
Vegetables	1986	2000	2002	Peak	5.2	7.8	8.8	No
Wheat[c]	1975	2004	2011	Peak	6.9	9.5	11.8	Yes
Wood[c]	1976	2004	2011	Peak	2.2	3.1	5.1	Yes
	Global drivers							
GDP[c]	2004	2010	2012	Peak	6.8	10.2	13.9	NA
Population	1988	1989	1989	Peak	1.3	1.3	1.3	NA
	National drivers and resources, biodiversity							
Hous. intensity Australia	1971	1976	1991	Peak	1.1	1.4	1.6	NA
Hous. intensity Canada	1971	1976	1981	Peak	1.1	1.7	1.9	NA
Hous. intensity China	1960	1960	1988	Peak	2.0	3.2	3.7	NA
Hous. intensity England[c]	1931	1981	2001	Peak	0.6	0.7	0.7	NA
Hous. intensity Ireland[c]	1981	1996	2002	Peak	0.6	1.1	1.2	NA
Hous. intensity Japan	1960	1965	1995	Peak	1.3	2.1	2.4	NA
Hous. intensity Luxembourg[c]	1930	1970	2001	Peak	0.6	0.7	0.9	NA
Hous. intensity New Zealand	1976	1976	1986	Peak	0.8	1.0	1.2	NA
Hous. intensity USA[c]	1940	1970	2000	Peak	0.6	0.9	1.2	NA
Meat, USA	1909	1955	1999	Peak	2.4	3.6	8.8	NA
No. spp. dom.	−3500	−2600	−1500	Peak	0.0	0.1	0.1	NA
Patents, USA[c]	1997	2010	2012	Peak	9.5	14.8	24.8	NA
Water, USA[c]	1975	1975	2005	Peak	2.0	2.1	2.1	NA

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[a]Peak rate year (50th percentile from the bootstrap distribution of year of the maximum of the first derivative of the spline smoothed bootstrap time series resample) of extraction of non-renewable resources, harvest of renewable resources and growth of drivers. When the 50th percentile is equal to the last year in the time series it was concluded that no peak rate year was detected, suggesting a still accelerating rate. The standardized growth rate at the peak rate year with uncertainty is also provided

[b]Time series of drivers and of national resources are not tested for independence (NA: Not Applicable) because they are not used to test synchrony which is based only on global resources. These global resources have to be statistically independent so that peat, phosphate, cotton, dairy, fish farmed, indigenous meat, and vegetables are removed to leave a set of statistically independent global resources

[c]Resources show a peak rate year but the 97.5th percentile year equals the last year of the time series indicating that the upper uncertainty interval is truncated

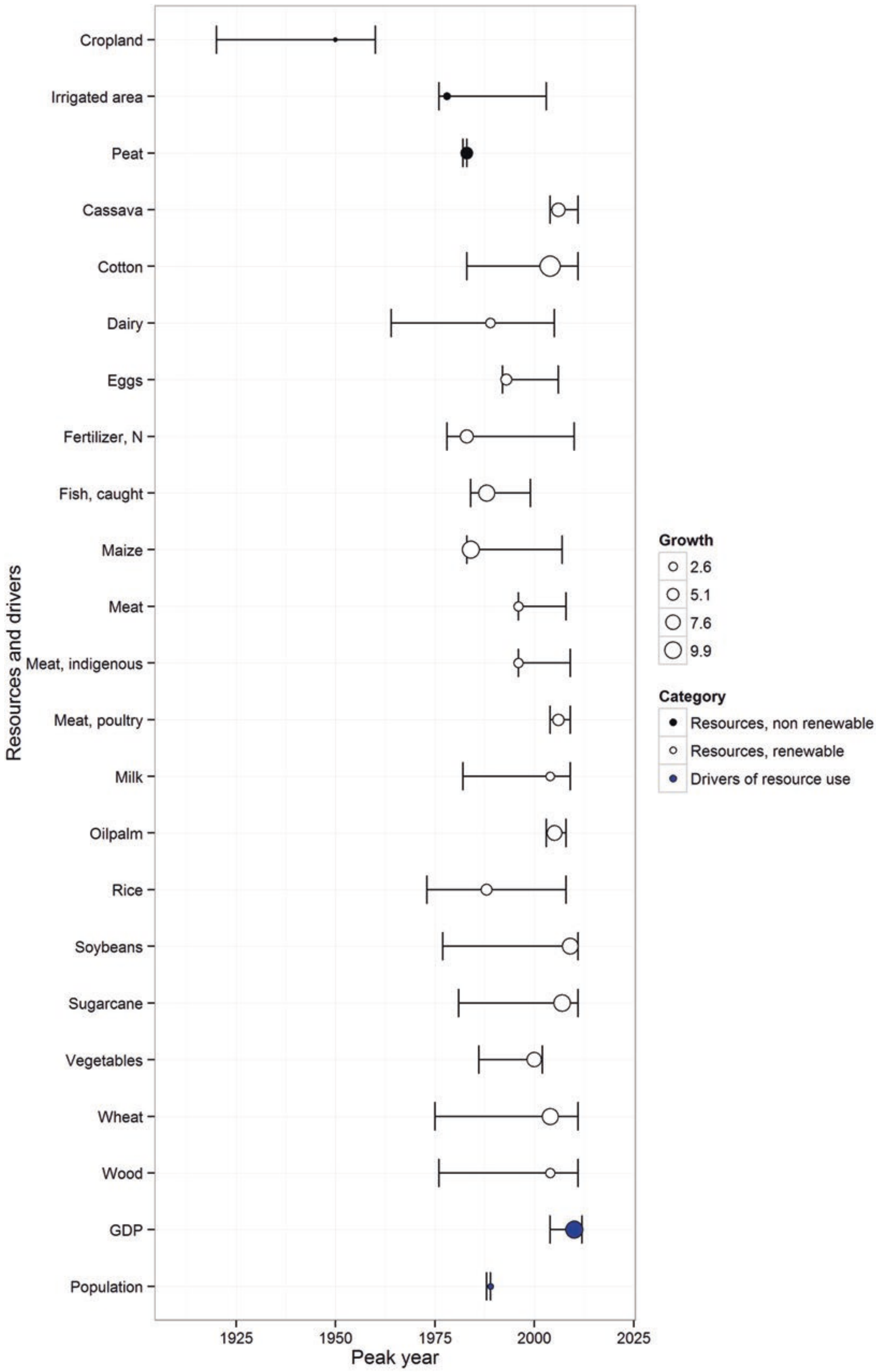

Fig. 46.3 Estimated peak rate year (median) of global resources and drivers (21 out of 27 resources, and 2 out of 2 drivers). The uncertainty bars represent 2.5th and 97.5th percentiles of 5000 bootstrap estimates. Point size denotes the relative growth rate in percentage in the year of maximum growth. The bootstrap distribution of peak rate year is skewed with the peak rate year often not centred in the confidence interval. (*Reprinted from* Seppelt et al. [7]; *with permission*)

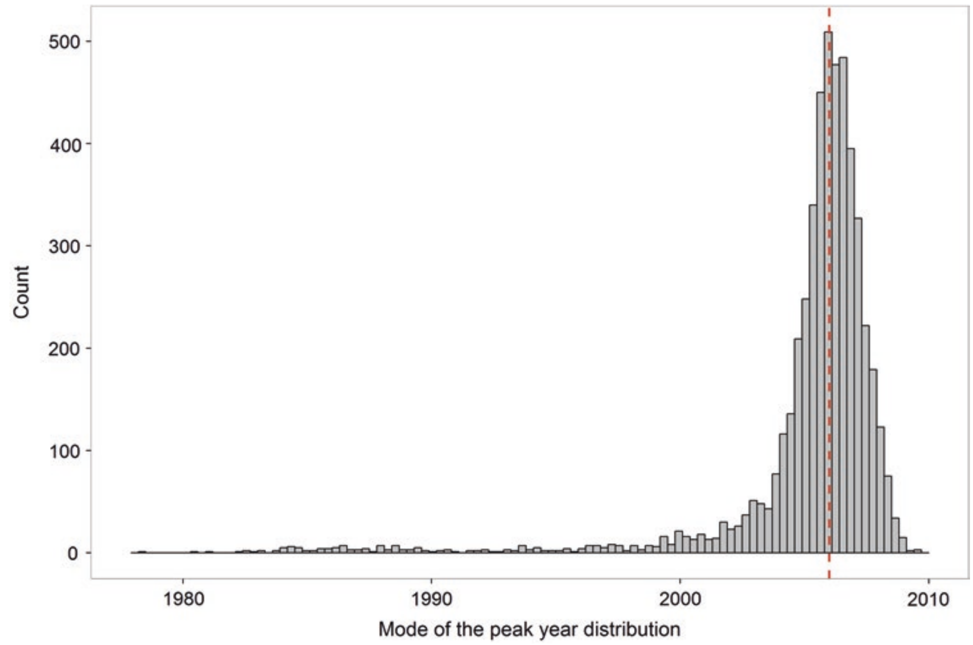

Fig. 46.4 Synchrony of the peak rate years is evident. For each of the 16 out of 20 statistically independent global resources showing a peak rate year, one peak rate year out of the 5000 from the bootstrap resample was randomly selected. The mode of the resulting smoothed distribution of 16 peak rate years was obtained, and the process repeated 5000 times resulting in the mode histogram, with a median of 2006 (1989–2008) in red. A non-parametric goodness of fit test rejects ($P < 0.001$) the two-sided null hypothesis that the histogram was sampled from a uniform distribution (i.e., no synchrony). (*Reprinted from* Seppelt et al. [7]; *with permission*)

46.3 Discussion

Passing an individual peak rate year means that a portion of the resource is still available and implies that society is entering a new phase with respect to its relation to the resource: substitutes may be available, or less of the resource is needed owing to more efficient use, or further expansion is either not possible or prohibitively costly because of inaccessibility. In general, the continued increase in extraction for an inaccessible resource results in an increased ecological and economic cost per unit extracted [17]. In all cases, adaptation is needed or underway, but may ultimately be constrained by physical limits. For instance, global food requirements need to be fulfilled by changing among crop species, distributing harvest more effectively, or shifting diets [14]. In this process of adaptation, agrobiodiversity is a fundamental constraint. The rate of domesticating species, the biological foundation of food provisioning, began to slow around 2600 BC (3600–1500 BC)—well before our era.

Current trends in technological development and institutional arrangements can induce a peak rate year for an individual resource. However, synchrony among the peak rate years signals that several planetary resources have to be managed simultaneously, accounting for resource distribution and utilization [18]. Synchrony does not necessarily imply a tipping point that leads to disastrous outcomes, because trade-offs are theoretically possible, and adaptation—such as the current increasing rate of renewable energy generation—has the potential to be accelerated. Synchrony also suggests that the debate about whether humans can devise substitutes for individual natural capital needs to be broadened to assess simultaneous substitutability [19]. Whether substitution and recycling will alleviate constraints to future economic growth [20] remains an open question, especially since maintaining the innovation rate requires increasing expenditures on human capital [21].

The synchronization of peak rate years of global resource appropriation can be far more disruptive than a peak rate year for one resource. Peak rate year synchrony suggests that the relationship among resource appropriation paths needs to be considered when assessing the likelihood of successful adaptation of the global society to physical scarcity.

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Part IV

Societal Responses

47 Introduction to Part IV: Societal Responses

Matthias Schröter

This part of the Atlas of Ecosystem Services focuses on the types of societal responses to addressing drivers and risks to ecosystems, as well as on societal demand and dependence on ecosystem services (Fig. 47.1). Society is affected by changes in ecosystem services provision that occur because of drivers affecting ecosystems and because of trade-offs between different ecosystem services. This has implications for human wellbeing and values [1, 2]. In response to these implications, different forms of land management can be triggered which either affect ecosystem properties [3] or target the management of different drivers [4]. In this context, the search for suitable governance options is considered one of the most pressing research challenges in the field of ecosystem services [5]. While work in this field has often aimed for providing additional arguments for conservation and hence decision-making [6], there have been few examples of how such information actually feeds in decision-making processes leading to societal responses to ecosystem service risks. Providing policy-relevant ecosystem service information tailored for decision-making processes has so far not been a focus in ecosystem service research [7, 8].

In the framework chapter of the Atlas (Schröter et al., Chap. 1), we have distinguished three types of societal responses: (1) avoidance; (2) adaptation; and (3) transformation. These are addressed in various ways in the chapters of this part.

47.1 Avoidance

Societal responses can target the avoidance of risks to ecosystems and the services they provide. In this context, Schröter-Schlaack et al. (Chap. 52) demonstrate the potential usefulness of monetary valuation for decision-making. An economic approach can help to recognise and demonstrate ecosystem service values, which in turn can lead to informed decisions. Economic approaches can also lead to a capture of ecosystem service values, i.e., the translation of these values into policy instruments that stimulate a reduction of drivers affecting ecosystems. When enforcing regulations to, e.g., reduce the negative effects of drivers on ecosystems and the services they provide, payment schemes could be used to compensate land users (Matzdorf et al., Chap. 51). Schröter-Schlaack and Hansjürgens (Chap. 48) explore important uncertainties arising around the choice of such policy instruments. These uncertainties relate to targets, i.e., the elements of biodiversity and ecosystems that policy should aim at. This requires an increased understanding of the relationship between biodiversity as well as ecosystem functions and ecosystem services (see, e.g., Harrison et al. [9]) and the complex interrelationships within ecosystems (see, e.g., Oliver et al. [10]). Also, the relevance of a mix of drivers that a policy aims to address, and the suitability of policy instruments, should be carefully studied, as policy outcomes are uncertain. Ring and Schröter-Schlaack (Chap. 49) therefore recommend a policy mix of different instruments that might partly overlap in addressing drivers as an insurance measure against the uncertainties involved in the application of these instruments.

47.2 Adaptation

Adaptation relates to living with the change of ecosystem services through finding substitutes or adjusting preferences. Schröter et al. (Chap. 50) exemplify this strategy by awareness-raising measures that have the potential to lead to adjusted preferences of locals and visitors in the context of conservation measures.

47.3 Transformation

Transformation focuses on ecosystem management. Applying a policy mix with partly overlapping instruments is recommended in this context by Ring and Schröter-Schlaack (Chap. 49). Schröter et al. (Chap. 50) exemplify the transformation strategy with a mangrove conservation and restora-

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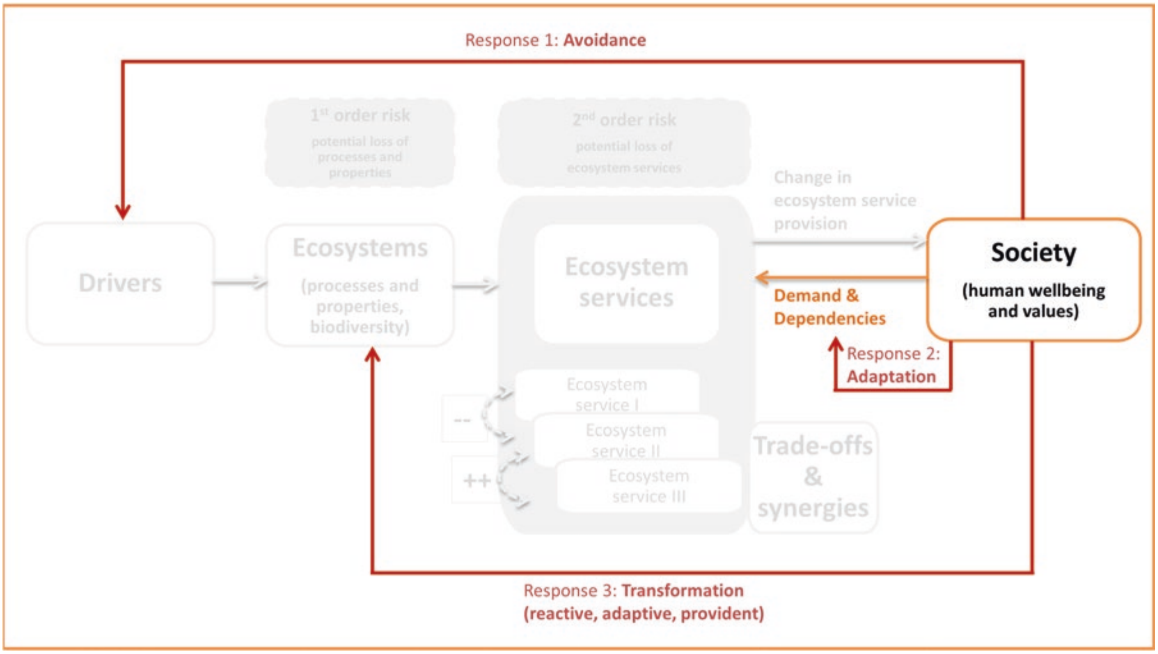

Fig. 47.1 Elements of the framework of The Atlas of Ecosystem Services addressed in this part

tion scheme. Furthermore, the authors describe a funding scheme related to water management based on ecosystem services in rangelands in Florida. Matzdorf et al. (Chap. 51) point out that payment schemes can be implemented to either address drivers (avoidance strategy), or to influence ecosystem management, taking ecosystem services into account. Urban city planners could use the results of spatially explicit analyses such as done by Kabisch (Chap. 53) and Banzhaf et al. (Chap. 26, Drivers), showing uneven distribution of urban green space, to active development of green space in deprived areas. The strong dependency of island-dwellers on ecosystem services that cannot simply be imported, highlights the need to work with ecosystems in order to adapt to the consequences of climate change (Förster et al., Chap. 54). Schulz-Zunkel et al. (Chap. 55) relate their findings on ecosystem services of floodplains to revitalisation measures helping to improve the provision of these services. de Knegt et al. (Chap. 56) suggest a couple of conservation measures, or nature-based solutions, that could be used to promote sustainable agriculture and thus to support biodiversity, the functioning of ecosystems, and the services they provide. Similarly, Settele et al. (Chap. 57) provide suggestions for ecological engineering to increase pest control on rice fields. Lakner et al. (Chap. 58) assess the effectiveness of environmental instruments in the context of the Common Agricultural Policy of the European Union.

47.4 Demand, Dependencies and the Distribution of Benefits

Economic and other, interdisciplinary approaches to ecosystem services can be used to elucidate societal demand, and the distribution of costs and benefits among different groups. This includes willingness-to-pay estimates for grassland ecosystem services (Schröter-Schlaack et al., Chap. 52) and the indication of preferences for services, as well as perceived risks by stakeholders (de Vreese, Chap. 59). Land management and planning could be informed through such analyses. A spatial overlay of the capacity of ecosystems to provide services with the societal demand or actual use can inform decision-makers about where actions need to be taken to promote sustainability (de Knegt et al., Chap. 56).

A crucial issue for societal responses is the distribution of benefits derived from ecosystem services. These benefits might be unevenly distributed among different groups [11, 12], and one additional set of policy instruments might specifically focus on this aspect. Ring and Schröter-Schlaack (Chap. 49) discuss the importance of public goods, often provided in the form of regulating or cultural services accessible to the public, and private goods, e.g., provisioning services traded on markets. Kabisch (Chap. 53) and Banzhaf et al. (Chap. 25, Drivers) present data on the uneven, and potentially unequitable, distribution of urban green space that provides public goods such as opportunities for recreation. Matzdorf et al. (Chap. 51) point to distributive consequences of payment schemes for ecosystem services and the normative choices involved in implementing these measures.

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48 Governance Risks in Designing Policy Responses to Manage Ecosystem Services

Christoph Schröter-Schlaack and Bernd Hansjürgens

48.1 Challenges in Designing Policy Responses to Manage Ecosystem Service Risks

Biodiversity and many ecosystem services possess the attributes of public goods; private markets, therefore, cannot fully recognize their value and will thus fail to ensure their sustainable management (market failure) [1]. Public policies therefore have an essential role to play in safeguarding biodiversity and managing ecosystem service risks. Ecosystem service risks can be defined as the interaction of a *hazard* that might harm the services an ecosystem provides to society (individual, groups, communities, sectors, regions, etc.) and the *vulnerability* of a social-ecological system, which is constituted by its exposure, susceptibility, and resilience (meaning its ability to cope with and adapt to the disruption/loss of service) (see Chap. 1). Public policies are needed to integrate ecosystem service values and (long-term) management consequences into private and public decision-making. This can be achieved by putting adequate institutions, regulations, financing, and price signals on markets in place.

The underlying assumption in many contributions on choice and design for effective biodiversity and ecosystem service governance policies is based on a simplistic cause-solution-chain (Fig. 48.1): An unwanted environmental harmful activity is to be regulated by a policy instrument; once the policy instrument is in place, the problem is solved. Early environmental policy, e.g., in air quality control, operated with direct regulation that prohibited environmentally damaging activities by setting management or emissions standards with which companies had to comply (see, e.g., Tietenberg [2]).

As the character of many environmental problems has changed (e.g., the decreasing role of point sources; the increasing relevance of diffuse sources, mixed pollutants, far-reaching temporal and regional scales), new policy instruments have emerged (e.g., market-based policy instruments) in response to the decreasing acceptability of command-and-control regulation, and such simplistic models of a policy-impact chain have begun to lose their persuasiveness. Tradable rights (emission/development) or environmental taxes or subsidies establish incentives for policy addressees that give them the freedom to decide whether, where, when, and how to adapt their behaviour, or to become obliged to buy a tradable right or to pay the tax. It is hence no longer clear whether a policy response will effectively reduce, e.g., environmental pollution by the desired degree (with respect to the certainty and speed a given target is reached, or the relevant regional scale). Moreover, policies never operate in isolation but are

Which ecosystem services are addressed? All types of ecosystem services, particularly those that have to be governed by public policies

What is the research question addressed? What are inherent governance challenges to the choice and design of effective policy responses to manage ecosystem service risks?

Which method has been applied? Conceptual consideration based on literature review

What is the main result? Setting targets and objectives of policies as well as selecting instruments based on a correct conceptualisation of human behaviour is full of potential pitfalls. These "disruptions" may be seen as types of governance failures or governance risks that pose risks to sustainably managing ecosystem services

What is concluded, recommended? To develop effective policy responses for managing ecosystem service provision, it is necessary to consider the specific characteristics of ecosystems, their services, and the impact of management decisions (e.g., uncertainty, spatial and temporal heterogeneity, irreversibility) in interrelation with the existing governance structures and institutional regulations influence both social dependence on, and demand for, ecosystem services

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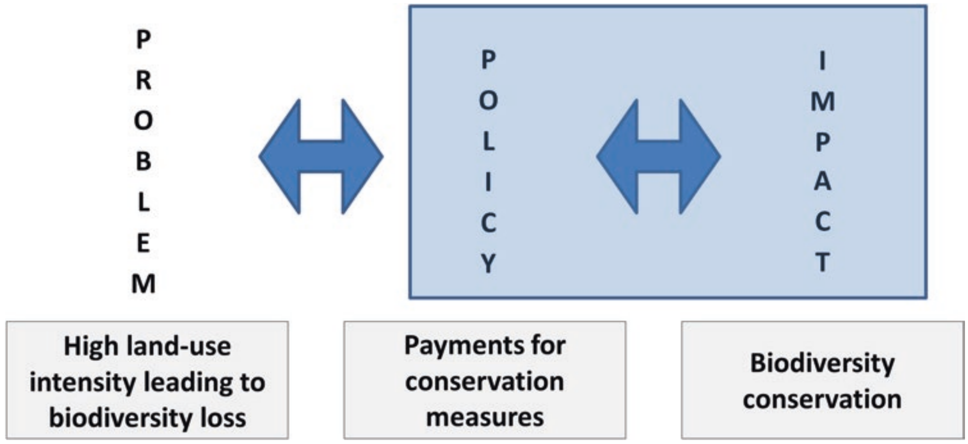

Fig. 48.1 Simplistic policy impact model

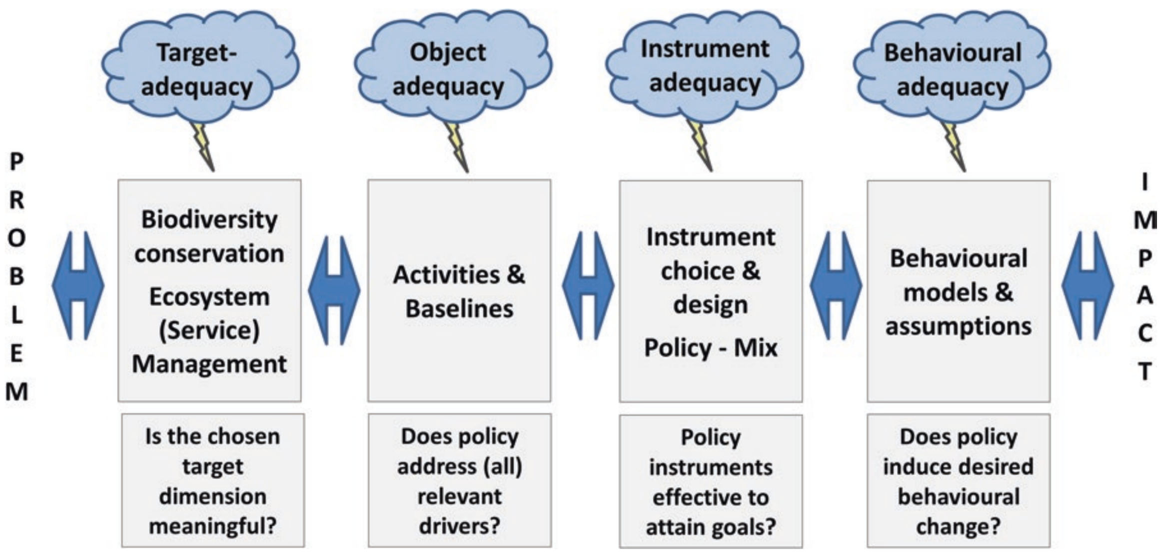

Fig. 48.2 Sophisticated policy impact model—disruptions in designing policy responses to address environmental problems

embedded into an existing institutional and cultural context that may render them inappropriate or at least ineffective. Thus, there is the risk of governance failure that poses, in turn, an ecosystem service risk.

Based on these observations, this chapter sheds some light on the impact chain from recognizing ecosystem service risks to implementing an effective policy solution for their management. The aim of our contribution is to highlight the various challenges inherent in policy choice and design for managing ecosystem service risks. Thus we focus on ecosystem service risks that are induced by the risk of governance failure. In this chapter, we refer to four disruptions (Fig. 48.2):

1. The identification of adequate target-dimension related to ecosystem service risks (**target-adequacy**). Understanding the relationships (synergies and trade-offs) between different target dimensions (e.g., biodiversity and/or ecosystem services; emissions and/or critical loads): Is the right target dimension addressed by the policy intervention?
2. The identification of the drivers of ecosystem service risks (**object-adequacy**). Does the policy response address the right drivers of ecosystem degradation and/or mismanagement of bundled ecosystem services?
3. The choice and design of policy instruments (**instrument-adequacy**): Given the existing institutional and cultural background, can the chosen policy instrument achieve the desired targets and thus effectively reduce ecosystem service risks?
4. Assumptions about human behaviour (**behavioural-adequacy**): Does humans' behaviour follow the predicted patterns, i.e., is the policy response setting an effective impetus to change behaviour and enhance ecosystem service management, or are there adverse behavioural patterns?

48.2 Governance Risks for Effective Biodiversity and Ecosystem Services Policy

48.2.1 Target-Adequacy

A policy response needs to be bound to a tangible target dimension. Examples of target dimensions include: an ecosystem, a landscape, a specific species to be protected, and an ecosystem service value to be better recognized in decision-making. It is uncertain whether such target dimensions are a suitable proxy for the underlying problem contributing to an ecosystem service risk. For example, protecting endangered species may increase the population of this species, but may not improve the overall biodiversity of the ecosystem and/or the provision of ecosystem services. Policies may thus become at best, ineffective or, at worst, elevate ecosystem service risks.

The nexus between biodiversity and ecosystem services as potential target dimension of policies may serve as an example. Although biodiversity conservation policy is increasingly justified based on the ecosystem services provided, there is still incomplete empirical evidence on the relationship between biodiversity conservation and supply of ecosystem services [3]. There is now sufficient evidence that biodiversity per se either directly influences or is strongly correlated with certain provisioning and regulating services, while for other services the evidence is mixed, the contribution of biodiversity to the service is less well defined, or there is insufficient data to evaluate the relationship between biodiversity and the service at all [4]. Nevertheless, in some assessments, and even more in political strategies to promote biodiversity conservation via ecosystem services (Nature-based solutions, Green Infrastructure), the two terms *biodiversity* and *ecosystem services* are used almost synonymously, implying that they are effectively the same thing and that if ecosystem services are managed well, biodiversity will be retained and vice versa. At the same time, biodiversity is itself sometimes regarded as an ecosystem service [5]. It is likely, however, that a focus on ecosystem service provision will overemphasise some components of biodiversity, e.g., habitats or species that are deemed to be important for certain ecosystem services. Other components, like those still unknown or those whose functions are not (fully) understood, may slip out of conservation priorities, thereby threatening the long-term functionality of ecosystems and their resilience. Similarly, Bartkowski et al. [6] have shown that most economic valuation studies that seek to value biodiversity economically focus, in fact, on targets other than biodiversity.

These uncertainties and knowledge gaps on the mutual influence of target dimensions (biodiversity and ecosystem services) or on valuation objects (in the case of economic valuation studies) may cause tremendous disruptions for designing effective policy responses to ecosystem service risks. This poses risks on the effectiveness of governance intervention. Several questions have to be addressed, e.g.: What type of biodiversity, what species, genes, sites or landscape pattern at what scale to focus on in order to address the (under-)provision of certain ecosystem services? What would be (unintended) consequences for biodiversity if conservation strategies were increasingly designed to deliver (certain) ecosystem services?

48.2.2 Object-Adequacy

Following the Millennium Ecosystem Assessment [7] and Pereira et al. [8], the main drivers of biodiversity loss and associated ecosystem degradation are land use change (and respective habitat loss), alien species invasion, environmental pollution (contaminants and nutrients), and climate change. In many cases, ecosystem service risks are the result of intensified land uses to promote the provision of a single service (e.g., intensive agriculture for food production) while reducing or diminishing the provision of other services (e.g., soil erosion control, groundwater quality control, landscape beauty) (see Foley et al. [9], p. 573). Identify and addressing *all* relevant drivers of ecosystem degradation is a decisive factor in designing effective policy responses to provide more sustainable solutions in managing such land use trade-offs. Policy interventions targeting only specific drivers or being contained in a specific sector (e.g., nature conservation) may be ineffective if other activities contributing to an ecosystem service risk (e.g., nutrient emissions through agricultural practises) aren't addressed as well.

This can be highlighted by the example of land use intensification in agriculture and the loss of grassland in Germany. For the last 25 years the proportion of agricultural land allocated to grassland has been in decline. In 1991, more than 5.3 million ha (31% of agricultural land) was managed as permanent grassland; by the end of 2013, this had decreased to 4.6 million ha (less than 28% of agricultural land) [10]. Extensively used grassland provides a range of valuable ecosystem services that get lost if grassland is converted into cropland [11].

Key driving forces behind the ploughing up of grassland in Germany are the intensification of dairy cattle farming and the growing profitability of field crops, including energy crops [12]. Furthermore—and often neglected—the relentless growth in land used for urban development has occurred primarily at the expense of agricultural land, particularly cropland. However, as land resources diminish, coupled with a high (and publicly subsidised) demand for food, feed crops, and energy crops, farmers compensate for the loss of crop-

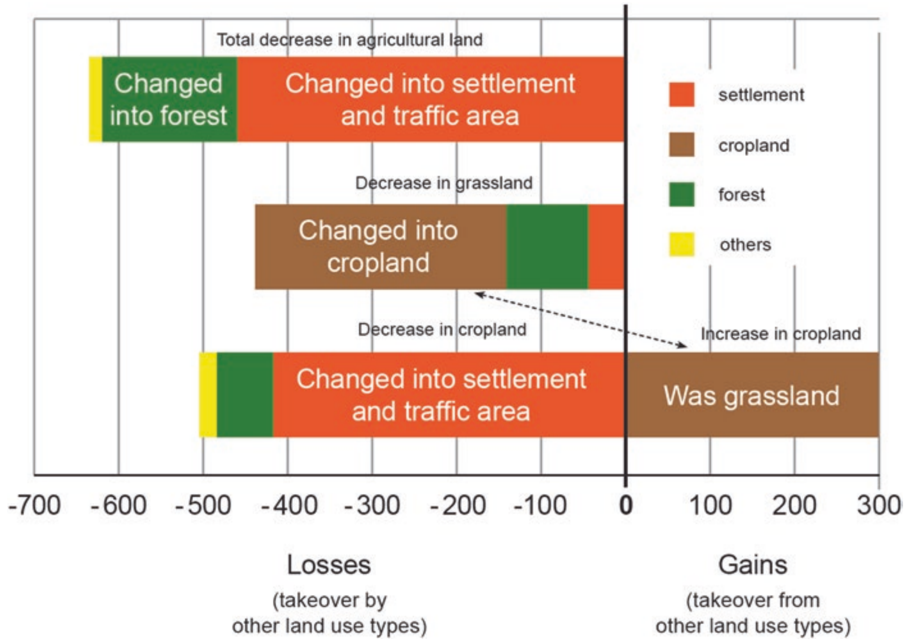

Fig. 48.3 Land use change in different types of agricultural land in Germany between 1990 and 2010 in 1000 ha. (*From* Natural Capital Germany—TEEB DE [11], slightly modified after Tietz et al. [21]; *with permission*)

land by ploughing up grassland. As a result, agricultural land decreased by more than 600,000 ha between 1990 and 2010, but over the same period, around 300,000 ha of grassland was converted into cropland (see Fig. 48.3).

Against this background, current mechanisms to protect grassland under the EU Common Agricultural Policy (CAP) appear inadequate. On the one hand, there is a lack of suitable instruments and regulations to capture the ecosystem services provided by grassland, above and beyond its provisioning services. On the other hand, notwithstanding long and extensive discussions [13], there is also a lack of effective policy mechanisms to combat urban sprawl, which renders existing policy mechanisms to protect grassland in Germany ineffective and hampers the management of associated ecosystem service risks. The policy target of Germany's sustainability strategy still allows a land consumption of 30 ha by 2030. This inadequate approach to the objects (drivers) of the policy problem—to protect grasslands in agricultural areas through strict EU CAP regulations and effective policies to combat urban sprawl—can be seen as a second governance failure that contributes to ecosystem services risks.

48.2.3 Instrument-Adequacy

A third challenge for implementing effective policies is the choice and design of adequate policy instruments. There is typically a divide between regulatory instruments (i.e., command-and-control instruments) and market-based interventions such as environmental taxes or tradable permits (see, e.g., Sterner [14]). Regulatory instruments define property rights and thus make use or access rights legally enforceable. They operate by either direct public provision of biodiversity conservation (e.g., protected area designation) or standard setting (management or pollution standards, spatial planning). These instruments create obligations for policy addressees independent of (perceived) opportunity costs. While this may reduce the cost-effectiveness of such instruments, they are nevertheless important in order to achieve spatially inclusive minimum standards, e.g., for ecosystem management. In contrast, market-based instruments provide financial incentives to stakeholders. Based on the assumption that decisions on ecosystem management are primarily taken on the basis of financial cost-benefit considerations, they strive to alter private costs and benefits so that any unaccounted social costs (and benefits) of environmental degradation can be "internalised" to ensure the desired environmental improvement [15].

Depending on the source, magnitude, and timing of ecosystem service risks, some policy instruments will be more effective than others [16]. Regulatory instruments will have to play a crucial role in safeguarding a minimum level of biodiversity to avoid crossing critical thresholds of ecosystem functioning and service provision. Besides comprehensible criticism on their fine-grain design (see Pe'er et al. [17]), the greening measures of the EU CAP, including the obligatory designation of ecological focus areas for every farm, are a prime example of such an instrument. Market-based instruments merit particular consideration for cost-effective policy interventions to manage ecosystem service provision. The discussions on the introduction of nitrogen taxes in different countries across Europe highlight the ability of market-based instruments to both control for diffuse emissions not easily caught by direct regulation and substantially decrease the overall use of fertilizers.

Two aspects are decisive for the effectiveness of policy instruments: First, policy instruments never operate in isolation, though at least many textbook studies pretend they do. Instead, they are embedded in an existing institutional and cultural context that may influence their effectiveness and efficiency (for policy mix analysis see Ring and Schröter-Schlaack, Chap. 49). Therefore, instrument selection is not in any way trivial and dependent on a proper understanding of the causes of ecosystem service risks. Second, designing policy instruments is an art: The fine-grain design of instruments (e.g., setting management standards, optimising the spatial allocation of protected areas, determining appropriate tax rates, setting caps for permit trading) constitute a huge range of challenges that may render policy responses ineffective (see, e.g., Wunder et al. [18] for an overview on payments for ecosystem services).

48.2.4 Behavioural Adequacy

A fourth disruption arises when policy addressees do not follow an otherwise well-designed intervention because the policymakers' underlying assumptions on their behaviour turn out to be incorrect. This is especially a challenge for biodiversity conservation and ecosystem service management, where a broad range of motivations (ethical, spiritual, economic) for action are involved. Changing (often traditional) behaviour via policy instruments may fail, if policy addressees feel that a certain stimulus is not appropriate. This phenomenon is well reported throughout conservation literature, where, e.g., the use of economic incentives can undermine ("crowd out") or reinforce ("crowd in") people's intrinsic motivations to engage in biodiversity and ecosystem conservation. With the rise of the ecosystem service-concept, economic incentives have been more and more widely applied in environmental policy, but there is evidence that raises concern about whether such instruments inspire desired behavioural changes. Some authors suggest that the changes these instruments can induce in motivations may, under certain conditions, undermine long-term conservation efforts (for a review on this topic, see Rode et al. [19]).

For example, a survey-study conducted by Lienhoop and Brouwer [20] indicates that German farmers have multiple motivations for enrolling in afforestation schemes, which are an important pillar of nature conservation activities in Western Saxony. The study concludes that one factor driving low implementation rates of the afforestation scheme is the neglect of the non-financial motivations that govern decision-making by farmers, in particular the lack of technical forest management advice and the possibility to return back to agriculture once the conservation contract ends (see Fig. 48.4). As different motivations correlate with a perception of barriers and preferences for certain policies, governments would be well advised to consider the diverse set of aspirations and motivations of policy addressees when designing conservation programs.

Fig. 48.4 Afforestation contract design attributes and changes in subsidies desired for enrolment [20]

48.3 Conclusion

The aim of this chapter was to highlight the complex challenges inherent in the choice and design of effective policy responses to manage ecosystem service risks. We have shown that setting targets and objectives of policies as well as selecting instruments based on a correct conceptualisation of human behaviour is full of potential pitfalls. These disruptions may be seen as types of governance failures or governance risks which themselves pose risks to ecosystem services. At the same time, these challenges call for interdisciplinary research activities to integrate natural and social sciences with respect to ecosystem management. It is necessary to consider the characteristics of ecosystems and the impact of their management (e.g., heterogeneity of ecosystems and potential service provision, irreversibility of change) in interrelation with the existing governance structure and institutional regulations influencing both the social dependence and demand for ecosystem services as well as ecosystem management and actual service provision. Decision-making to manage ecosystem service risk is taking place in a multi-level multi-actor system at the interface of private and public goods. Providing vital risk management solutions requires a thorough understanding of the governance challenges involved.

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49 Policy Mixes for Sustained Ecosystem Service Provision

Irene Ring and Christoph Schröter-Schlaack

49.1 Policy Instruments to Address Ecosystem Service Risks

Policies and, more specifically, policy instruments to achieve certain policy objectives, are important means to govern and manage ecosystem service risks and are thus a main class of societal responses to ecosystem service risks. When focusing on ecosystem services and associated risks, it is essential to keep in mind biodiversity *and* ecosystem service governance. Due to the manifold synergies and potential trade-offs between biodiversity and ecosystem services, as well as those among different ecosystem services [1], such governance arrangements usually involve several policies and policy instruments, so-called policy mixes. A *policy mix* is defined as a combination of policy instruments that has evolved to influence the quantity and quality of biodiversity conservation and ecosystem service provision in public and private sectors [2]. As can be seen from Fig. 49.1, there is a wide range of policy instruments to address biodiversity loss and associated ecosystem service risks [3]. Although all instruments may aim at conserving biodiversity and sustaining ecosystem service provision, they do so by very different mechanisms. On a general basis, one can distinguish between direct regulation, incentive-based approaches, and market facilitation. Whereas direct regulation operates by either direct public provision of biodiversity conservation (e.g., protected area designation) or standard setting (management or pollution standards, spatial planning), incentive-based instruments do so by providing financial incentives and disincentives to stakeholders. Within the group of incentive-based instruments, one can further distinguish between price-based and quantity-based mechanisms [4]. The former include, for example, pollution taxes to reduce environmentally harmful actions as well as various types of payments for environmental service provision or biodiversity conservation (such as PES, REDD, ecological fiscal transfers) to alter the costs of different land-use/land management options. Quantity-based approaches, such as tradable permits and habitat banking, directly impose quantitative restrictions on certain activities and create fungible easements. Finally, informative and motivational measures provide actors with knowledge about the consequences of their behaviour, thereby building on or facilitating intrinsic motivation for self-regulation in conserving biodiversity or avoiding ecosystem service risks.

Which ecosystem services are addressed? All types of ecosystem services and the trade-offs involved in their provision and with respect to biodiversity conservation

What is the research question addressed? How do policy instruments that aim to manage ecosystem service provision interact? How to combine instruments to design effective policy mixes for biodiversity conservation and sustained ecosystem service provision?

Which method has been applied? Conceptual consideration based on literature review

What is the main result? We present a policy mix framework that provides guidance for (1) identifying drivers and societal context of ecosystem service risks; (2) identifying gaps in existing policies; and (3) supporting instrument choice and design to develop effective policy responses

What is concluded, recommended? Given the manifold challenges involved in ecosystem service management, we emphasise the need to look at the interaction of single policy instruments and search for effective policy mixes to halt biodiversity loss and govern ecosystem service risks

A comprehensive literature review on policy instruments for biodiversity conservation and a sustainable provision of ecosystem services has shown that policy mixes are not only a matter of fact in real-world policies, but also that combining instruments can be theoretically justified for reasons of

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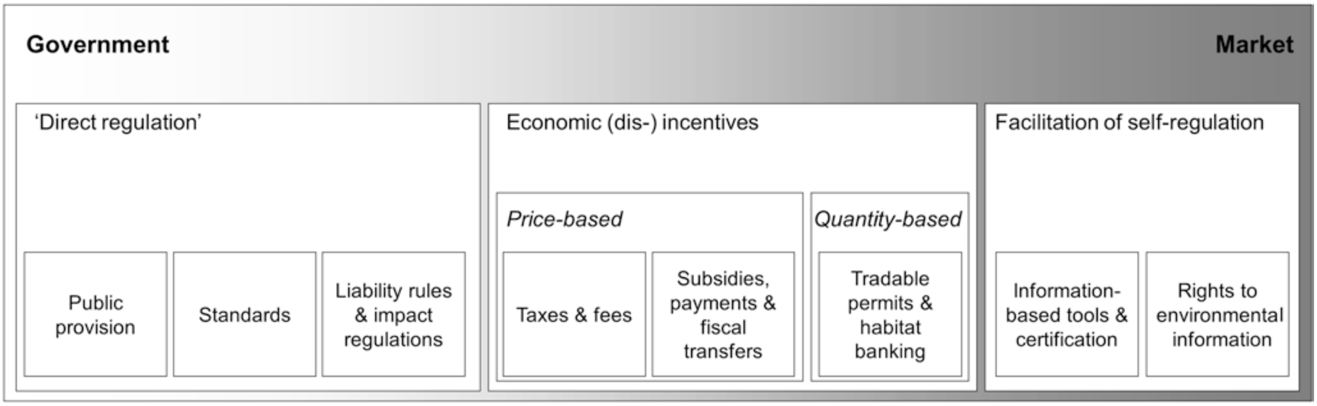

Fig. 49.1 Continuum of policy instruments to address ecosystem service risks. (*From* Ring and Barton [8]; *adapted from* Schröter-Schlaack and Ring [3])

ecological effectiveness and a range of other motives [5]. In the following we look at selected challenges for societal responses addressing ecosystem service risks. Each of these challenges may call for a different policy mix based on the specific roles of instruments, i.e., their strengths and weaknesses, in the mix. Challenges and suitable policy instruments can be related in varying degrees to the three types of societal responses, namely avoidance, adaptation, and transformation (Chap. 1).

49.2 Challenges Calling for the Combination of Instruments

49.2.1 Dealing with Uncertainty and Ignorance

There is still great uncertainty and ignorance about the resilience of ecosystems, thresholds in ecosystem change, and the biodiversity-ecosystem service nexus. When choosing and designing policy instruments, it seems to be wise to err on the side of caution, as unintended or unpredicted consequences of human activities may cause irreversible biodiversity loss and associated harm to human well-being [6:26]. Hence, the policy response to ecosystem service risks should include instruments whose strength is to protect a "safe minimum standard" of biodiversity conservation independent of dynamically evolving cost-benefit considerations of the addressed actors. In this context, the establishment of protected areas, no-take zones, or prohibitions of the use of certain products and substances heavily impacting biodiversity and causing ecosystem service risks, are important regulatory instruments. Direct regulation should thus be a key component of a sound policy mix [7] and is highly relevant in the context of societal responses of avoidance and anticipated transformation.

Furthermore, uncertainty, risk, and especially ignorance may call for multiple instruments to be applied simultaneously [8]. Although redundancy of instruments or their overlap in addressing actors, drivers, or pressures are often looked at with scepticism in environmental policies [9], the use of multiple instruments may indeed act as an insurance against knowledge gaps, policy, or implementation failures in the case of biodiversity conservation [2]. When drivers of biodiversity loss and ecosystem degradation are pervasive and cut across many sectoral policies, policies and avoidance strategies to mitigate these drivers should themselves be pervasive and capable of filtering through the entire economic system [10:271].

Lastly, as our knowledge of ecological functioning, resilience, and critical thresholds evolves over time, an important component of the policy response to manage ecosystem service risks will be motivational, educational, and informative tools. The strength of these tools is to facilitate preference change and the alteration of traditional management practices, which is particularly relevant to adaptation, but also, to a lesser degree to the other two societal responses.

49.2.2 Risks Associated with Ecosystem Service Bundles

Societal demand for ecosystem goods and services does not exist for a single service but for ecosystem service bundles [11]. Land-use and sectorial regulations likewise influence the provision of multiple services simultaneously. Within these bundles, some ecosystem services, like many provisioning services, possess private good characteristics and can be traded in markets, while other services, like many regulating services, mostly represent public goods that lack working institutions to capture their value in decision-making. For marketable provisioning services, incentive-based instruments to correct for market failures and to internalise external effects merit special consideration. For non-marketable regulating or cultural services, incentive-based instruments may be less appropriate or will require careful design, often in combination with other instruments.

For example, owing to ethical convictions, relevant stakeholders may not deem economic instruments as appropriate for governing such services [12]. Hence, while critical conservation thresholds to avert ecosystem service risks should be safeguarded by highly effective measures, such as direct regulation, more flexible incentive-based instruments merit consideration for governing utilisation rates of ecosystem services at safe distances from critical thresholds.

49.2.3 Addressing Spatial Heterogeneity of Ecosystem Service Risks

Ecosystem service provision and demand are often unevenly distributed across space. For example, water quantity control, flood regulation, recreational potential, and soil erosion control can differ substantially between upstream and downstream uses in watersheds (Fig. 49.2). This in turn has consequences for policy responses to govern ecosystem service risks of all three societal responses: Many incentive-based instruments are unable to control for spatial allocation of compliance or conservation activities to avoid ecosystem service risks [13]. Indeed, it will be necessary to either fine-tune the design of incentive measures, e.g., by spatial bonuses or by coupling them with direct regulation, such as zoning approaches to spatially target conservation efforts. Many incentive-based approaches to foster private conservation efforts require a baseline of compliance with minimum management standards and a certain level of self-contribution to conservation action in order to specify additional activities which are then eligible for remuneration. Very often, as is the case in Europe, such a baseline is provided by direct regulation, i.e. legally defined management standards such as good practices in agriculture or forestry [7].

Fig. 49.2 Watersheds are characterised by substantial spatial heterogeneity regarding upstream and downstream uses of ecosystem services and associated risks. Photo by Irene Ring

49.3 Evaluation of Policy Mixes to Cope with Ecosystem Service Risks

The design of biodiversity and ecosystem service governance as a societal response to ecosystem service risks must consider the interplay of different sectoral policies, and should employ policy mixes using smart combinations of instruments across space, time, and policy sectors. Against this background, it is vital to understand the role of different policy instruments in the mix, as well as interactions among them. When it comes to analysing policy mixes, the focus is on the interplay—the complementarity, synergies, and conflicts—among the instruments involved, and the ability of the policy mix to address the underlying problems [2, 8]. Building on earlier work, we suggest a three-step policy mix analysis framework (Fig. 49.3) [1, 3].

49.3.1 Step 1: Identifying Challenges and Context

The first step is to identify the context and the main challenges for a policy response, as well as the suitable societal response(s) to the relevant ecosystem service risks (avoidance, adaptation, and/or transformation). The appropriate mix of instruments will depend upon the nature of the ecosystem service risk, the target groups of policies, and wider contextual factors. Hence, this step consists of gaining a thorough understanding of the policy object. What are the drivers of biodiversity loss and ecosystem service risks, and how might these be adequately addressed? What are important characteristics that will influence appropriateness, applicability, and success of instruments and their combinations? Lastly, what are the policy objectives regarding risk governance and management?

Within this first step, it is necessary to identify private and public actors in the affected political and economic sectors at relevant governance levels. Moreover, constitutional and legal requirements, as well as cultural perceptions of nature, may open up options or constrain the choice, design, and implementation of policy instruments [12].

49.3.2 Step 2: Identifying Policy Gaps and Choosing Instruments

The second step includes criteria and recommendations regarding the choice of instruments, the functional role of different instruments in addressing the challenges and societal responses selected in step 1, and interactions between instruments in policy mixes.

Gaps in the existing policy mix must be identified in relation to the ecosystem service risk to be governed, and potential instrument alternatives or complements must be chosen. For this purpose, it is necessary to identify the policies already in place, as many aspects related to biodiversity and ecosystem governance or drivers of their degradation are already covered or at least influenced by existing policies.

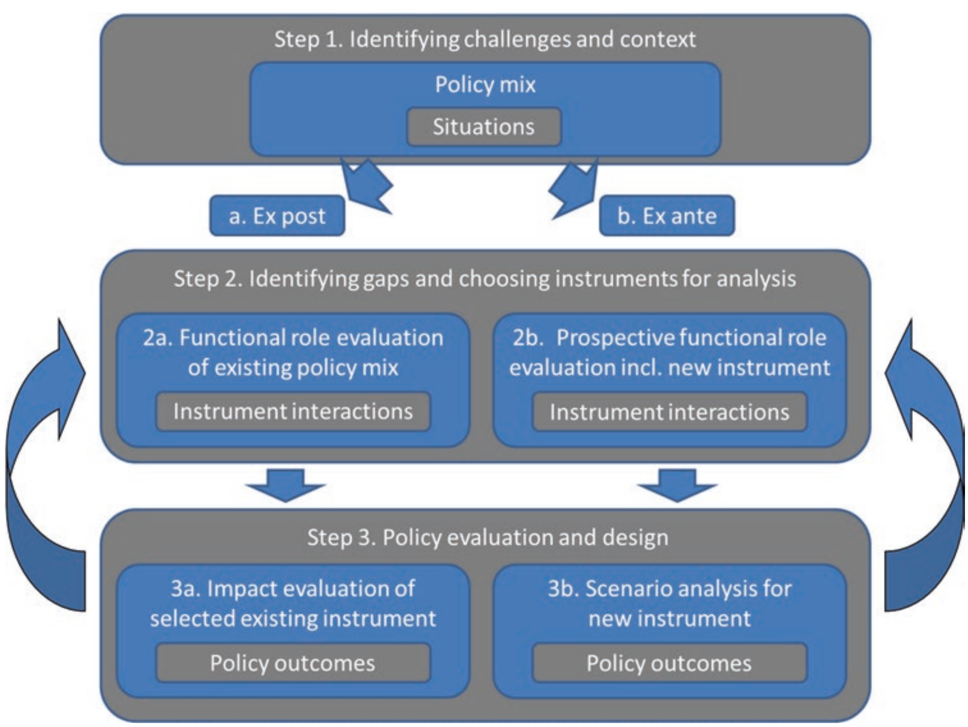

Fig. 49.3 Three-step-framework for ex post and ex ante analysis of policy mixes. (*From* Ring and Schröter-Schlaack [1], *with permission of* SpringerNature, and Schröter-Schlaack and Ring [3])

These policies will not always originate only from environmental policies, but might stem from different sectorial policies such as agri- and silvicultural, energy, transport, or trade policy. Taking stock of existing policies may point to shortcomings, unaccounted trade-offs, and blind spots of the currently applied instruments.

Based on such assessment, there are two pathways to enhance the overall performance of the policy mix. On the one hand, one could aim at improving the existing instrument mix by better considering the effects of instrument interaction in a fine-grain design of single components of the mix (ex post analysis). On the other hand, to account for yet unconsidered aspects of the problem (ex ante analysis), one may opt for introducing new instruments to the existing mix. This may include, for example, actors, activities, or sectors thus far not explicitly addressed or the consideration of recently evolved knowledge on ecosystem service risk—on either the supply or the demand side.

Lastly, if instruments are applied simultaneously they will not necessarily work towards the desired policy goal, e.g., ecosystem service risk governance; they may also interact and thereby influence the performance of the policy mix. Instrument interaction thus needs to be considered in terms of potential conflict, synergies, and complementarities between instruments, while also considering the strengths and weaknesses of individual instruments in the mix [3, 8].

49.3.3 Step 3: Policy Evaluation and Design

The third step focuses on evaluation and design of single instruments so that the value added by the relevant instrument to the existing policy mix for biodiversity and ecosystem service risk governance is maximised. To develop policy recommendations, we refer to the following policy instrument evaluation criteria: conservation effectiveness; cost-effectiveness; social impacts, fairness and policy legitimacy; and institutional aspects. When dealing with policy mixes, the ultimate goal for instrument design is no longer to develop first-best or second-best single policy solutions, but to optimise design regarding the functional role of each instrument in the overall policy mix [3].

49.4 Conclusion

Real-world policies are characterised by the existence of policy mixes. This holds true for policy responses to the ongoing biodiversity loss and the associated degradation of ecosystems' ability to provide ecosystem services. Such responses should not be limited to single instruments or mixes in "environmental" or "conservation" policies only, but should encompass policy mixes including other sectorial policies, like agriculture, energy, and transport. Despite this observation, most of the literature on instrument choice and design has focused on the analysis of individual instruments rather than on policy mixes.

This chapter has presented a step-wise framework for assessing instruments in policy mixes for biodiversity conservation and ecosystem service provision that—due to its generic character—can be easily transferred to governing and managing societal responses to ecosystem service risks. However, as in any other policy field, there will be no "blueprint" for optimally designing a policy mix for biodiversity and ecosystem service governance and associated risks, as all countries are different and rely to varying extents on biodiversity and ecosystem services [6]. Ecosystems may be in different stages of degradation and thus in different proximity to ecosystem service risks. Our policy mix analysis framework provides step-wise guidance and allows starting with the more easily available opportunities.

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Societal Response, Governance, and Managing Ecosystem Service Risks

50

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50.1 A Notion of Ecosystem Services Governance

In contrast to the traditional understanding of command-and-control government action, the governance of ecosystem services prominently refers to decision-making processes by which the use of common goods and services are decided upon by a broad range of stakeholders and societal actors. Different actors often operate not only alongside, but also in collaboration with, the state [1]. The concept involves an element of authority as well as processes and structures for shaping peoples' priorities and coordinating their actions [2]. To govern complex social-ecological interrelationships and manage ecosystems, cooperation between the public and private sector complements state legislation, law enforcement, and market distribution. Thus, in political sciences and institutional economic theory, three governance models are distinguished: hierarchies, markets, and community management based on cooperation. Natural resource management literature frequently refers to the latter as a promising approach. In reality, the different governance structures most often operate together in combinations and policy mixes that consist of hierarchical orders, market mechanisms, negotiated agreements, and consensus-finding processes, such as cooperatives, strategic alliances, joint ventures, consortia, and others (see Ménard [3] for an overview, and Vatn [4]; see also Fig. 50.1 and Table 50.1).

On a conceptual level, four main ideas summarize the governance concept: (1) Governance means governing and coordinating interdependences of different actors, including public and private ones; (2) governance is based on institutions in the sense of rules and rule-making systems that coordinate actors' actions; (3) governance models occur in form of hierarchical, market, community management approaches, and combinations of these; and (4) governance includes collective action that may be institutionalized, as for example in networks or contractual relations that often exceed organisational boundaries, especially in terms of state and society. The system boundaries of state and non-state action become more fluid [5].

Which ecosystem services are addressed? Research addresses especially the challenge to govern and manage risks of ecosystem services (ES) that show the characteristics of common or public goods, which includes primarily regulating, cultural services, and biodiversity (they differ for each presented case, however)

What is the research question addressed? How do different governance models (as described by involved actors and chosen institutional arrangements) help to address typical challenges of ES governance and manage ES risks?

Which method has been applied? Mixed method approach, combining results from literature studies with results from own case study field work, based on stakeholder interviews and focus group discussions

What is the main result? To exemplify different governance solutions for ES risk management, we present four case study examples, each referring to one of the following responses to ES risks: mitigation of drivers to prevent ES shortage, trade-off management to support multiple ES, adjusting social preferences for certain ES, and transformative adaptation strategies

What is concluded, recommended? We conclude that hybrid forms of governance, which involve actors from multiple societal spheres and combine features of hierarchical, market-based, and community-based approaches, support adaptive governance and help to provide the necessary flexibility for accommodating different strategies for ES risk management. Thereby context matters greatly, as each solution is an outcome uniquely shaped by the local frame conditions

The characteristics of most ecosystem services as common pool resources or public goods, the lack of detailed

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Fig. 50.1 Different governance models

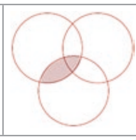
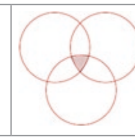
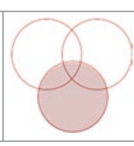
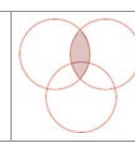
information on socio-ecological interactions, distinct management objectives, heterogeneous actors, and institutional diversity display a set of challenges specific to the governance of ecosystem services. Ecosystem services governance must deal with complexities, scientific uncertainties, and social-political ambiguities in order to prevent the potential loss of ecosystem services provision for society. Societal preferences of ecosystem services bear the risk of a second order, referring to the loss of ecosystem services due to hazards or vulnerability of socio-ecological systems (see Chap. 1 by Schröter et al.). Correspondingly, governance of ecosystem services comprises—ideally—multiple sectors and multiple levels. It concerns all social, political, and economic spheres of the society, and refers to and interconnects the global and local levels [2]. Governance structures determine processes for ecosystem uses, set standards, perform allocation, influence motivations, initiate or reduce conflicts, and resolve disputes among actors [6]. Examples include the development of international agreements for biodiversity conservation and climate protection at the global level, national biodiversity strategy decision-making through parliaments, and cooperative decisions of community-based natural resource management at the local level. Governance further includes integrated, border-crossing political processes, as national environmental policies are internationally linked and at the same time, coincidently interwoven with the creation of regional and local spaces.

We distinguish four major governance challenges for sustainable ecosystem services provisioning [7]. We refer to general aspects of governance—on the one hand, particular actors and institutions, and on the other hand, issues of social-ecological dynamics and limited and uncertain knowledge. Correspondingly, we acknowledge (1) heterogeneous actors; (2) institutional diversity; (3) dynamic processes; and (4) fragmented information and expertise. Handling these entails risks, depending on the vulnerability of socio-ecological systems, but it also offers opportunities to deal with these risks in terms of ecosystem services management, which provides solutions for finding trade-offs and managing inequity. We elaborate on these challenges for ecosystem services governance and ground them in respective scientific literature on governance, institutions, and ecosystem services research, and demonstrate implications for managing ecosystem service risks.

50.1.1 Heterogeneous Actors

Ecosystem governance is characterized by heterogeneous actors that can be broadly divided into three groups. First, there are economic actors, such as owners and users of natural resources, and these actors have certain rights. Second, there are political actors with the power to define property rights as well as interaction rules. Third, there are civil society actors who ensure the democratic legitimacy of political action and define a normative basis for the society [2]. Land-use systems consist of various land managers, overlapping agencies from different policy sectors and administrative levels; development and environmental NGOs, business and research organizations, and so on. The interests, motives, and roles of actors can be mixed: an individual may be a political and economic actor or be a member of an organization, as well as participating in civil society. Different interests arise as supply and benefits of ecosystem services are unequally distributed, while the provision of ecosystem services is usually a result of interplay between ecosystem management decisions at a certain level and the benefits of different management forms at another level. To give an example, on one hand, the protection of mangroves at the local level creates benefits at levels from local to global, ranging from fish as aliment to biodiversity for recreational effects to carbon storage and positive impacts on climate change mitigation [8]. On the other hand, land use planning and regulation on the regional or national level sets conditions of land management by local actors. Thus, governance of ecosystem services, when negotiating trade-offs in the provision of particular ecosystem functions and services, faces the challenge of considering diverse interests, but also the demands and value systems of different actors at different scales. However, many of these trade-offs are unknown or not recognized and therefore not well treated in some

Table 50.1 Four examples from practice representing different governance models

Region:	Cardoso Island, Brazil	Region:	Osa Peninsula, Costa Rica
Location:		**Location:**	
Governance Model:	Hybrid: Hierarchies and community management 	**Governance Model:**	Hybrid: Markets and community management and hierarchies 
Actors:	State actors and community actors	**Actors:**	State, market, and community actors
Institutions:	**Co-management** agreement between state park and community: The community was granted the right to stay in the park (typically communities get relocated by the state based on its sovereign rights), given that they now perform monitoring activities on **biodiversity** for the state park. To achieve this, the community self-organized and founded an association, through which it could also participate in the negotiation of the environmental management plan	**Institutions:**	Close cooperation between communities and several corporate enterprises (e.g., Volkswagen) for a **payment for ecosystem service (PES)** scheme for **carbon sequestration** projects for the conservation of mangroves. The PES is combined with an approach for environmental awareness rising in the communities. State actors are involved through providing the legislative framework for the PES arrangement
Processes:	Rules in the management plan can be renegotiated and adapted every 5 years.	**Processes:**	The Payment for Ecosystem Service agreement was short-term and currently is open for renegotiation
Shared Expertise:	The community actors monitor where intruders harm the resources of the park (e.g., illegal fishing, palm heart harvesting, taking out rare orchids, etc.), but only the park authorities can make arrests and enforce environmental laws	**Shared Expertise:**	The communities run a nursery for mangrove seedlings and reforest devastated areas and maintain them, the car retailers fund these activities and market generated credits to their customers for voluntary carbon mitigation
Response to Ecosystem Services Risk:	Avoidance through mediation of trade-offs between ecosystem services to satisfy the ecosystem services needs of different actor groups, e.g., local fishermen (fish species as provisioning services) vs. conservation authorities (fish species as part of local biodiversity)	**Response to Ecosystem Services Risk:**	Avoidance to mitigate global drivers that cause destruction of Mangrove ecosystems, e.g., through conservation of Mangroves into productive areas for shrimp farming
References:	[20, 21]	**References:**	[22]
Region:	**Spreewald, Germany**	**Region:**	**Northern Florida, USA**
Location:			
Governance Model:	Community management 	**Governance Model:**	Hybrid: Hierarchies and Markets 
Actors:	Community actors	**Actors:**	State and market actors (ranchers)
Institutions:	Community actors founded a **citizen foundation** with the mission to preserve the unique **cultural landscape** with its typical **biodiversity** in the region. A specific feature is that besides private individuals also the municipalities and towns of the region became benefactors of the foundation	**Institutions:**	Ranchers formulate bids to get contracted for an output-based **agri-environmental scheme** (entitled Northern Everglades Payment for Ecosystem Services Program, NE-PES) that aims for increased **water retention** on their land. The NE-PES resulted from the Florida Ranchlands Environmental Services Project (FRESP)

(continued)

Table 50.1 (continued)

Processes:	Projects are rather short term, for their financing private and public money is pooled	**Processes:**	Ranchers' contracts run from 5–10 years, before another program cycle starts
Shared Expertise:	For implementing conservation projects, the foundation relies to a large extent on the voluntary engagement of benefactors and their local knowledge. Only for specific tasks third parties are contracted	**Shared Expertise:**	Ranchers use the knowledge of their land to propose so-called water management alternatives (WMA) that result in higher water retention. Bids get selected based on cost-benefit considerations
Response to Ecosystem Services Risk:	Adaptation for adjusting social preferences of the local population and visitors to the region, i.e., to make them more aware of the uniqueness of the historic cultural landscape through direct voluntary engagement into the conservation efforts	**Response to Ecosystem Services Risk:**	Transformation through adaptation of employed strategies: the NGO-initiated small-scale FRESP project was upgraded into a full-blown state-financed agri-environmental scheme
References:	www.spreewaldstiftung.de	**References:**	[23], www.fresp.org/ne_pes.php

existing governance settings, which increases the risk of biodiversity and ecosystem services loss.

Solving issues of trade-offs largely depends on actors' power positions and access to decision-making processes, but presents a chance to handle ecosystem service risks. Fair stakeholder engagement and conflict management processes are needed for balancing interests or, even better, making productive use of conflicts about different ideas, values, and knowledge to develop a commonly shared solution. Participation structures and negotiation opportunities appear necessary for robust governance structures [2]. From a broader perspective, ecosystem services governance needs open and proactive network management that fosters exchange and cooperation between relevant actor groups [9].

50.1.2 Institutional Diversity

Governance refers to a dynamic interaction of actors and institutions in terms of rules [5]. Institutions include formal and informal rules defining policy processes and interaction in the civil society sphere, comprising constitutional and collective choice rules, the rights to resources and the rules of interaction, and the norms of civil society [10].

Socio-ecological systems count with institutional diversity in terms of multi-layered vertical interactions from the local to national to global, and polycentric horizontal interactions, such as different policy sectors like agriculture, nature conservation, and infrastructure. Policy interventions can differ in their objectives or even contradict or conflict with one another. Besides production or environmental objectives, a plethora of other objectives are targeted, such as rural development, bioenergy, and infrastructure. The ambivalence of policy and management objectives may lead to land-use conflicts, e.g., between economic development and nature conservation [11]. Finding ways to ensure that all players act coherently, effectively, and efficiently for the sustainable provisioning of ecosystem services underlines the crucial role of formal and informal institutions and their interplay. The interplay between institutions can be seen as a result of their functional interdependencies or as politically created for strategic purposes. A major challenge for governance is to ensure sound institutional interplay [12] and synchronize thematically related institutions in a socio-ecological system to increase chances of reaching intended objectives [13]. Problems of institutional interplay occur when (new) institutions hamper the performance of other institutions already in place, e.g., if rules for a protected area do not fit to traditional conservation practices of local people. Therefore, approaches of multi-level governance, which brings together actor and institutional levels, provide a chance to handle ecosystem service risks addressing scale mismatches.

50.1.3 Dynamic Processes

Governance settings are shaped by the unpredictable, dynamic, and evolved nature of linked social and ecological systems, from policy to diverse local practice and settings, from impacts and conflicts to uncertainty and limited information [14]. As a way forward, adaptive assessment and co-management frameworks have been developed [15] to better deal with system dynamics, uncertainty, and surprise by incorporating approaches combining different types of knowledge, feedback loops, and learning arrangements. These approaches focus on experimentation and learning, which brings together research on, and actors and organizations for, collaboration, collective action, and conflict resolution in relation to natural resource and ecosystem governance and management [16]. Key to such adaptive governance approaches is that they can only be understood within social-historical contexts, characterized by phases of institutional emergence, continuity, and change. As adaptive co-management approaches focus on collaboration of government and private civil society actors, they are intended to supplement policymaking with models of stakeholder participation, constructive social assessment of interests and values, evaluation, and feedback. They present the opportunity to adapt to the occurrence of ecosystem risks by focusing on learning and co-generation of knowledge.

50.1.4 Fragmented Information and Expertise

Another aspect that may lead to ecosystem services loss is that, even though knowledge about the linkages between biophysical processes, ecosystem functions, and the provision of ecosystem services by different management systems is growing, there are major gaps in scientific evidence regarding how exactly these links work, how they correspond to human wellbeing, what responses they trigger, and which feedback loops they may lead to (see Ekroos et al. [17] for an overview). Precise and measurable information on the status and quality of ecosystem service provisioning are often missing. Governance must accept that it is difficult for any actor, state or non-state, to generate all knowledge needed to decide about the appropriate regulation of natural resource uses and ecosystem service provisioning. This requires integrated knowledge and reflexive approaches that can deal with epistemic uncertainty and the various socio-political influences, interests, and conflicts that exist and change over time [18]. The development of approaches to sustainable ecosystem services governance therefore implies an inter- and transdisciplinary way of proceeding. Such attempts that link scientists and practitioners aim to strengthen the co-production of knowledge and integration to achieve more socially robust problem-solving and enable mutual learning [19]. However, anchoring such research endeavours to decision-making remains a difficult task. A major challenge is the establishment of science-policy interfaces that bundle, manage, and establish a comprehensive and applicable basis for knowledge integration and information sharing.

50.2 Governance in Practice

For exemplification, we present four governance approaches from practice in Table 50.1, which represent international examples of different governance models, including three hybrid solutions. The examples relayed to four different countries: (a) Brazil, (b) Costa Rica, (c) Germany and (d) USA (see Fig. 50.2).

For the description of the four cases, we make specific reference to the four types of challenges typically faced by ecosystem services governance as outlined above: (1) heterogeneous actors (in view of the involvement of state, market, and community actors); (2) institutional diversity (in view of the different governance models outlined above); (3) dynamic processes (in view of how often the arrangement can be reviewed and changed); and (4) fragmented information and expertise (in view of how actors share their expertise). Also, each case relates to one of the possible responses to ecosystem service risks elaborated on in Chap. 1: (a) and (b) avoidance to mitigate drivers and trade-offs to reduce the negative effects of maximizing service provision at the expense of other services; (c) adaptation for adjusting social preferences for ecosystem services; and (d) transformation through adaptation of employed strategies.

50.3 Conclusions

Hybrid governance models can help to make socio-ecological systems more stable, e.g., through adaptive co-management, monitoring, bringing people together, and institutions. Actors may create platforms to negotiate trade-offs and find early solutions to adapt and prevent risks from materialising and ensure adequate ecosystem services provision. The challenges of ecosystem services governance also bring opportunities to respond to ecosystem service risks through avoidance, expectation, trade-off management, adaptation, transformation, and by taking a larger range of concerned actors, sectors, and levels collaboratively into account for collective action.

Fig. 50.2 Pictures of the example cases: (**a**) Cardoso Island State Park in São Paulo State, Brazil (image courtesy of Claudia Sattler). (**b**) Mangrove Nursery in Osa, Costa Rica (image courtesy of Barbara Schröter). (**c**) Cultural landscape in Spreewald, Brandenburg, Germany (image courtesy of Claudia Sattler). (**d**) Ranchlands in Florida, USA (image courtesy of Claudia Sattler)

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51 Payments for Ecosystem Services: Private and Public Funding to Avoid Risks to Ecosystem Services

Bettina Matzdorf, Carolin Biedermann, and Lasse Loft

51.1 Characteristics of PES as an Instrument for Risk Management

In the introduction to this book, an ecosystem service risk is defined as "a hazard potentially causing harm to the services an ecosystem provides to society." If this "harm to the [ecosystem] services [ES]" is caused by humans, environmental economists consider it a negative "external effect," i.e., a cost that affects a party that did not choose to incur that cost [1]. A number of governance approaches and mixes of policy instruments for fixing the market failure of externalities have been discussed (see Chap. 50). Among these instruments, Payments for Ecosystem Services (PES) have been propagated in recent years as a promising tool for mitigating the loss of ecosystem services, including biodiversity.

A variety of PES definitions and typologies exist [2–4]. A core element of most PES definitions is the provision of positive economic incentives conditional upon the supply of well-defined ecosystem services (or activities thought to yield well-defined ecosystem services). In other words, land users are paid to reduce allowable negative external effects on ecosystem services or for taking action to preserve or restore ecosystem services [5]. Besides conditionality, many authors consider it essential that only activities that would not have been conducted anyway are paid for (additionality, see, e.g., Engel et al. [4]). While some scholars have argued that these criteria are important for efficiency reasons [6], others emphasize the challenge in enforcing these criteria in complex social-ecological systems [7].

Furthermore, for the success of PES, it is important to take the social-ecological context into consideration. This includes site-specific environmental conditions as well as place-specific social conditions, such as social norms [8]. For example, the question of whether it is the obligation of a land user to provide an ecosystem service, or whether the land user should get paid for the provision of an ecosystem service, is a highly normative one. The answer to this

Which ecosystem services are addressed? All ecosystem services with public good characteristics.

What is the research question addressed? What kind of financial incentives in terms of Payments for Ecosystem Services (PES) exist and what is their potential? What are challenges to PES implementation?

Which method has been applied? Literature review and case studies

What is the main result? To date, different types of PES approaches co-exist. There are various fruitful examples for innovative and successful design and implementation around the world. Governmental activities are highly important for PES. The social-ecological context must be considered during the design and implementation process. A PES design is not only a technical tool for effective and economically optimal ecosystem services provision. It also needs to be created with multiple aspects of social justice and equity in mind.

What is concluded, recommended?
- A policy mix that includes PES is important for ecosystem services risk management.
- Depending on the given social-ecological context conditions, the use of economic incentives to influence human behaviour and the use of trade mechanisms to allocate resources can be a cost-effective and socially accepted approach, if combined with other policy instruments.
- Progress in ecosystem services quantification could promote the development of more output-based payment schemes.
- Intermediaries that are active on a regional level are often key players for PES development and implementation. Their participation should thus be encouraged.

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question is a societal decision, usually manifested in tenure and property rights. A PES design is therefore not only a technical tool for effective and economically optimal ecosystem services provision. It also needs to be created with multiple aspects of social justice and equity in mind. These aspects include, among others, a just procedure during the decision-making process for the allocation of use rights and the fair and equitable sharing of benefits and burdens from the provision of ecosystem services [9]. Such equity considerations can become instrumental for the success of PES schemes, as perceptions of unfairness can undermine the effectiveness of economic instruments [10].

In the following section, we will provide a closer look at the variety of institutional settings in which PES are embedded. This will stimulate the discussion about the contexts and actor constellations that can converge to make PES an appropriate instrument for ecosystem services risk management.

51.2 Institutional View on PES

Although the PES approach was initially conceptualised as an alternative to government interventions, governments play a key role in most PES schemes implemented to date (e.g., Schomers [3] and Vatn [11]). In addition to the fact that many ecosystem services have public good characteristics, one reason for the strong involvement of governments is the challenge of developing and implementing conditional PES. It is likely that only the state is willing and able to take over high transaction costs and the financial risk of ecosystem services not being provided. Besides law enforcement, governments can play a key role in the following tasks:

1. First, governments can exert sovereign influence on both service providers and users by regulatory legislation and thereby determine whether the stakeholders participate voluntarily;
2. They can furthermore act as ecosystem services buyers or financiers, representing society's demand.

Figure 51.1 shows four different types of PES that can be observed in practice, although the boundaries between them are often blurred.

Type 1 describes a PES in which private financiers compensate suppliers who take appropriate action to provide ecosystem services. Both sides act without regulatory pressure and the government is not directly involved. A financier can be the immediate beneficiary of the provided ecosystem services, can have altruistic motives, or want to preserve nature for its own sake (see Box 51.1). In all these cases, private financiers have a major interest in the objectives. But there may be also cases of "greenwashing," in which companies only pay for ecosystem services provision to improve their image without having a true interest in ecosystem services provision. In such cases, the financiers' self-interest lies not primarily in the promised ecosystem services but rather in image enhancement or public relations. This could be important for the privately funded PES, as a critical monitoring actor may be missing.

Fig. 51.1 Classification into four types of Payments for Ecosystem Services (PES). (Reprinted from Matzdorf et al. [5] with permission; adapted from Matzdorf et al. [12].)

Box 51.1: MoorFutures® (Mecklenburg-Western Pomerania, Brandenburg and Schleswig-Holstein, Germany)

PES-Type: *1—Voluntary non-governmental funded payments for voluntary actions*

Implemented: *2011*

ES targeted: *Carbon sequestration (by peatlands)*

Driver(s) of risk: *Intensive drainage of peatlands for agricultural and forestry purposes and/or peat extraction leads to increased release of carbon dioxide and nitrous oxide (= direct anthropogenic driver)*

Short description: *Businesses or private individuals may voluntarily offset their carbon emissions by purchasing certificates. Certificates are generated by rewetting peatlands in the participating regions to reduce carbon loss. One certificate equates to a saving of one tonne of carbon dioxide, which is achieved over a period of 30 or 50 years. The price currently lies between € 35 and € 80 [5].*

Box 51.2: Florida Ranchlands Environmental Services Project, FRESP (Florida, USA)

PES-Type: *2—Voluntary governmental payments for voluntary actions*

Implemented: *2005 (closed in 2012); led to the state agency PES Northern Everglades Payment for Ecosystem Service Program*

ES targeted: *Water quality and water quantity*

Driver(s) of risk: *Transformation and drainage of the Everglades for agricultural purposes led to severe problems with water supply and the quality of surface and groundwater (= direct anthropogenic driver)*

Short description: *FRESP was launched by the World Wildlife Fund, Resources for the Futures, and local ranchers. Because it was supported by federal and state funds, a detailed analysis of effective water management on agricultural land was possible, as was testing the different action possibilities for a PES. Building on the successful implementation of the pilot project, a state agency PES program was introduced that includes a solicitation process and a model-and output-based payment scheme [5].*

Box 51.1 presents an example of PES-Type 1. Map source: Copyright GeoBasis-DE/BKG 2012; figure adapted from [5].

Governments play a key role in **Type 2**, as they finance the provision of ecosystem services through governmental programmes (see an example in Box 51.2, Fig. 51.2). Here, the financiers are not the direct beneficiaries but function as a proxy for a diffuse social demand. Publicly funded PES, as well as PES-like governmental agri-environmental programmes, are widely used in many countries [3] and of great importance due to their enormous financial volume [11]. To finance these programmes, governments rely on tax revenues or on specific fees not coming from the direct ecosystem service-users (cf. Costa Rica's national PES programme “Pagos por Servicios Ambientales” [13]). In principle, providers participate voluntarily in these programmes. While some scholars consider large scale agri-environmental programmes, such as the US Farm Bill or programmes within the EU common agri-cultural policy, to be PES programmes, others prefer to use the term “PES-like.” Regardless of the terminology used, these programmes often started as agricultural subsidy programmes long before the term “PES” was used, and often lack well-defined ecosystem services and conditionality.

Box 51.2 presents an example of PES-Type 2. Map source: ESRI Boundary Layers (World), Version 2016.0; adapted from Matzdorf et al. [5].

Fig. 51.2 Cattle ranching is the main method of land use in the Northern Everglades, the region where FRESP is implemented. Image courtesy of Claudia Sattler

Type 3 depicts cases of PES in which the demand for ecosystem services is initiated by legislation. Vatn [11] calls these approaches liability systems. If a government, for example, places limits on pollution while allowing flexibility regarding the achievement of those limits, a demand for ecosystem services may arise. Private financiers could now opt for a PES as an alternative to a technical solution in order to comply with the legally required limits (see an example in Box 51.3). Another form of regulation in this context would be laws like the Endangered Species Act in the U.S., or the impact mitigation regulation (Eingriffs-Ausgleichsregelung) in Germany: These laws require compensation for any impact on the ecosystem balance, such as damage made to certain habitats. Habitat banks, compensation agencies (Flächenagenturen), and other voluntary suppliers offer appropriate compensation and replacement measures in response to the demand generated. Even if we consider this model under the PES approach, we must note that the environmental protection benefits are achieved due to the command regulation (the "cap"), while a trading mechanism is added to the instrument design to allow for more flexibility in reaching the environmental goal and reducing costs [11]. In the trading mechanism, the final payments to the voluntary supplier of appropriate measures follow the PES model, using financial incentives to promote ecosystem services provision.

Box 51.3 presents an example of PES-Type 3. Map source: ESRI Boundary Layers (World), Version 2016.0; adapted from Matzdorf et al. [5].

In PES of **Type 4**, governments enforce the provision of ecosystem services by prohibiting certain activities with negative impacts on ecosystem services and biodiversity (Fig. 51.3). To compensate the economic impacts of those regulatory requirements on the land use rights holder, such as restrictions on agricultural use, governments can provide payments. These payments do not fit into a narrow conceptualisation of PES, but in some programmes broadly defined as PES, e.g., in China, the payments are at least in some parts made for services (or activities leading to the provision of the ecosystem services) that are already required by law [14]. In this case, public payments are used as an additional incentive to achieve the desired environmental goals. Among other things, these compensation payments aim to increase the acceptance and legitimacy of the policy intervention. As in Type 2, we have found examples where governments fund these programmes not only by ordinary taxes but also by earmarked user fees. Box 51.4 gives an example of such highly regulated PES.

Box 51.3: Medford Water Quality Trading Program (Oregon, USA)

PES-Type: 3—Mandatory polluter-funded payments for voluntary actions

Implemented: 2011

ES targeted: Water quality (esp. water temperature in connection with cold-water fish)

Driver(s) of risk: Daily release of thousands m^3 of warm (but clean) water by a Water Reclamation Facility (= direct anthropogenic driver)

Short description: To meet thermal limits for influent wastewater set by a governmental permit, a Water Reclamation Facility finances riparian restoration projects to shade the Rogue River and thereby reduce stream warming caused by solar loading. In comparison to alternatives considered, for instance installing large mechanical chillers, the restoration programme offers not only ancillary environmental and social benefits, such as habitat creation and bank stabilization. It also supports the local economy and labour market and is significantly more cost-effective [5].

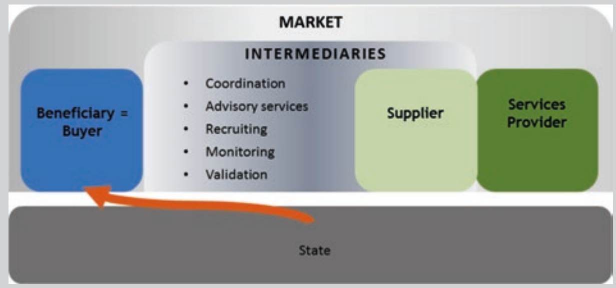


Fig. 51.3 Partially deforested slopes in Dien Bien Province, Vietnam. Image courtesy of Lasse Loft

Box 51.4 Vietnam's National Payments for Forest Ecosystem Services Programme, PFES (Vietnam)

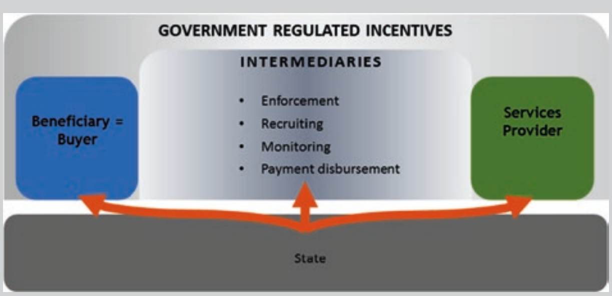
PES-Type: *4—Voluntary and mandatory governmental payments for involuntary actions*

Implemented: *2011 (after piloting phase 2008–2010)*

ES targeted: *Diverse ES from forests, such as watershed protection, natural landscape beauty and forest biodiversity (others, e.g., carbon sequestration, are planned to be included)*

Driver(s) of risk: *Logging of historic forests due to various reasons, such as the demand for high quality timber, clearance for aquaculture and agriculture as well as poor management (= direct anthropogenic driver)*

Short description: *Government agencies collect money from businesses that benefit from ES provided by healthy forests and distribute it to individual households who preserve the forests. The land users are not paid for the quantified provision of ES. Instead, they have to comply with use restrictions according to the Land Law and are financially compensated for conducting forest protection activities assumed to increase the provision of those ES with public goods characteristics. Due to the structure of commanded interactions between a) the entities and the government agencies as well as b) between the agencies and the land-owners, PFES is a policy instrument that combines tax-like elements on the user side with subsidy-like elements for forest stewardship on the provider side [9].*



Box 51.4 presents an example of PES-Type 4. Map source: ESRI Boundary Layers (World), Version 2016.0; adapted from Matzdorf et al. [5].

51.3 Outlook

Payment for Ecosystem Services, understood here as an umbrella concept for the provision of positive economic incentives, steer land-users' behaviour in complex social-ecological systems, and thereby play a key role in ecosystem services risk management. The use of economic incentives to influence human behaviour, and of trade mechanisms to allocate resources, can be a cost-effective and socially accepted approach when combined with other policy instruments. This policy mix is an alternative to classical governmental intervention alone [12].

- **Regionally active intermediaries are often key players for PES development and implementation, and should be encouraged:** Context-specific PES solutions are often characterized by complex actor constellations. Thus, intermediaries play a key role for PES [15]. Successful PES are often developed "bottom up" by intermediaries who want to preserve nature for society and for its own sake. These initiators have typically been rooted in the region for a long time and possess knowledge about the environmental and economic challenges of local service providers. To strengthen and actively involve the capacities of these kinds of actors, it is important to develop new and successful solutions on the ground [5].
- **Enhancements of large agri-environmental programmes are needed:** In Europe and North-America in particular, conventional agri-environmental (subsidy) programmes could be further developed into governmental PES programmes targeting site-specific and well-defined ecosystem services. These include high-quality drinking water, specific habitats, or carbon sequestration. Furthermore, these programmes should consider synergies and trade-offs between different ecosystem services. Additionally, output-based or results-oriented PES schemes, in which landholders are financially rewarded for their actual performance in terms of empirically verified provision of the ecosystem services, are often considered cost-effective [16]. However, in many current cases, information on the status and quality of biodiversity and ecosystem services provision is missing (see, e.g., Sommerville et al. [17]). The development of output-based PES schemes could thus profit from ongoing research activities to quantify and map ecosystem services. Targeted quantification efforts to measure biodiversity and ecosystem services for PES schemes would be helpful. However, output-based PES designs must address the complexity

and uncertainty of ecological systems and related risks for land users. Output-based schemes will only be accepted by land users if they are in control of the monitored outputs. As this cannot be ensured in many situations, while output-based PES may improve agri-environmental schemes, they are not a panacea. This holds true especially in the context of many developing countries in which the monitoring systems are often merely existent.

- **PES need active and innovative governments:** An additional approach to governmental politics for managing land use and related ecosystem services could be the active support in the development of voluntary markets for ecosystem services and the involvement of non-governmental financiers. A major task for governments would include the promotion and development of standards and certificates or credits. Further, governments could support pioneering projects financially. The main focus should be on innovative concepts for cooperation among relevant stakeholders, by means of which transaction costs can be reduced or shared [5, 18]. However, these privately financed instruments would need strong state regulation in terms of social and environmental safeguards.
- **Using market potential to supplement command and control systems:** Command and control systems are essential instruments for ecosystem services risk management. These systems can be supplemented by economic approaches, such as known biodiversity offsetting or cap-and-trade approaches, to increase efficiency. However, the interplay of regulations and economic tools must be evaluated critically by governments and civil society. If ecological compensation areas or specific ecosystem services are provided by a purely profit-oriented supplier in a mandatory market setting, there will be no interest in ecosystem services provision from either the supply or the demand side. In such cases, a precise formulation of the compensation requirements paired with rigorous governmental or civil society control is imperative (see, for critics, Spash [19]).

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The TEEB Approach for Demonstrating Societal Risks to Ecosystem Services: Taking Grassland Conservation as an Example

52

Christoph Schröter-Schlaack, Bernd Hansjürgens, and Miriam Brenck

52.1 An Economic Perspective on Ecosystem Service Risks

The provision of many ecosystems services is altered owing to present and future impacts on biodiversity that encompass both natural and anthropogenic drivers such as land use decisions. There is growing evidence that some ecosystems have been degraded to an extent that they are nearing critical thresholds or tipping points (conceptualized as first order ecosystem service risks in Chap. 1 of this Atlas), beyond which their capacity to provide ecosystem services may be drastically reduced (conceptualized as second order ecosystem service risks in Chap. 1), or beyond which the diversity of ecosystem services is reduced in order to maximize the provision of a few services.

An economic perspective is focusing on understanding the incentives behind land use decisions and illustrating the consequences of alternative land use options. Few ecosystem services have explicit prices or are traded in markets; more often than not, however, these marketable ecosystem services (typically, provision services such as crops, timber, or fibers) are preferred over non-marketable services (e.g., regulating and cultural services, such as climate regulation and landscape beauty, respectively) in land use decisions. While benefits of marketable services can be captured in the short term and by private land users (e.g., when crops or timber are sold on the market), costs of loss of non-marketable services tend to fall on society as a whole or on future generations, which exposes society to risks. Yet these ecosystem services are important to human well-being, and people may hold substantial values for these services, irrespective of whether they can be sold on markets. It is thus a huge challenge to identify and assess the benefits of non-marketable ecosystem services and associated risks and incorporate them into decision-making.

Against this background, the TEEB-study and Natural Capital Germany follow a tiered approach in analyzing and structuring valuation of ecosystem services, and thus the

Which ecosystem services are addressed? All types of ecosystem services, particularly non-marketed ecosystem services like regulating, cultural, and supporting services.

What is the research question addressed? What is an economic approach toward demonstrating societal risks of ecosystem degradation and biodiversity loss? How can societal costs of ploughing up grasslands be made visible?

Which method has been applied? Economic cost-benefit analysis of selected ecosystem services of grasslands.

What is the main result? The insufficient consideration of the full spectrum of ES and natural capital in land use and land management is leading to decisions that result in biodiversity loss and put the provision of ecosystem services at risk.

What is concluded, recommended? Understanding (the often economic) drivers of land use change and biodiversity loss as well, as assessing magnitude and societal distribution of costs and benefits of different land use options, can help mainstream the values of nature and ecosystem services into decision-making and ultimately helping to achieve a sustainable management of natural capital.

consideration of ecosystem service risks in decision-making [1, 2]:

1. Recognizing the multiple values of ecosystems (e.g., cultural, spiritual, social, economic) and thus the multiple sources of potential ecosystem services risks related to their loss;
2. Demonstrating ecosystem service values, i.e., making values and thus the magnitude and societal distribution of potential ecosystem service risks visible by applying

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economic valuation methods (including monetary, quantitative, and qualitative approaches);

3. Capturing ecosystem service values in decision-making by addressing drivers of biodiversity loss and ecosystem degradation to stimulate land users to provide well-balanced ecosystem service bundles that avoid or minimize the occurrence of ecosystem service risk (conceptualized in Chap. 1 of this atlas as ecosystem service risk response option 1: avoidance of drivers of biodiversity loss and ecosystem degradation).

In the following, we will use the example of ploughing up grasslands in Germany for demonstrating ecosystem service values of grasslands and for highlighting ecosystem service risks from its conversion. Details can be found in Natural Capital Germany [3], from which this example is taken.

52.2 Identifying Ecosystem Services of Grasslands in Germany

Grasslands are crucial for conserving biological diversity and for providing a wide range of ecosystem services above and beyond its role in agricultural production (see [3] for a recent synthesis of research results). Grasslands provide habitats for more than half of all the species occurring in Germany, and with its vegetation cover all year round, grasslands have high humus levels and a high capacity for water storage. Unlike arable land, grasslands offer better protection against soils drying up or being eroded by wind and water, in particular on sloping ground. On the periphery of waterbodies, grasslands act as a buffer by holding back nutrients and contaminants from the water system and reducing the risk of eutrophication. Grasslands are therefore pivotal to the protection of surface waters and drinking water.

For years, the proportion of total agricultural land allocated to grasslands has been in decline. While in 1991 in Germany more than 5.3 million ha (about 31% of all agricultural land) was managed as permanent grassland, by the end of 2013 this had decreased to just over 4.6 million ha (about 28% of agricultural land) [4]. Species-rich grassland with a particularly high nature value (HNV grassland) has been similarly affected: Between 2009 and 2013, the amount of HNV grassland in Germany decreased by 7.4%, or more than 82,000 ha, about half the size of the state of Hamburg [5].

The key driving forces behind the ploughing up of grassland are the intensification of dairy cattle farming in Germany, and the growing profitability of field crops, including energy crops [6]. Available agricultural land in Germany is also shrinking overall, mainly due to an increase of settlement structures and forest cover, which thereby augments existing trends of agricultural intensification [7].

The observable decrease in grassland has adverse consequences for numerous ecosystem services. For example, the greenhouse gas storage function of grassland is destroyed when it is ploughed; it also loses its function in groundwater purification and no longer provides habitat for a large number of species. Large sections of the population benefit from the supply of these ecosystem services—in the case of climate protection, mankind as a whole—yet the costs (or lost profits) associated with conserving and maintaining grassland rest with the local farmers. The problem is that the ploughing up of grassland is not exempt from valid grants and legislation, leading to adverse consequences for ecosystem services. The farmer's business decisions do not consider the costs of a reduced supply of these ecosystem services, yet they are ultimately borne by society.

52.3 Demonstrating Ecosystem Service Risks of Grassland Conversion

The decline of grasslands poses high risks for ecosystem services and thus for human well-being and economic prosperity. A comparison of the costs and benefits elucidates the economic advantages of preserving grassland versus ploughing it up (see Figs. 52.1 and 52.2).

For provisioning services, we based our calculations on the average additional yield of arable use versus grassland (data taken from Osterburg et al. [8]). For climate services we compared the average CO_2 emissions from soil under grassland with those from arable use, and extrapolated these with different compensation levels (data taken from Matzdorf et al. [9], Osterburg et al. [10], and Ring et al. [11]). For contributions to groundwater protection, we estimated the costs of measures needed to reduce elevated nutrient and contaminant levels with arable use to the equivalent level with grassland use (data taken from Osterburg et al. [8]). The value of nature conservation by keeping grasslands is based on the German public's willingness to pay for a program for permanent maintenance, creation, and upgrading of grasslands (data taken from Meyerhoff et al. [12]).

Figure 52.1 clearly shows that grassland conservation has major societal benefits, which by far outweigh revenues from ploughing up grassland to cultivate crops. Depending on the local conditions and the underlying assumptions made in the valuation, the net benefit to society of preserving grassland (difference between the lost business revenues and the social benefits) is estimated to be somewhere between 440 and 3000 € per hectare and year. Ecosystem services risks are particularly high for "high nature value"-grassland locations with sensitive soil conditions, such as low storage and buffer capacity for nutrients and contaminants, and locations at risk of soil erosion, which tend to be less profitable for arable farming.

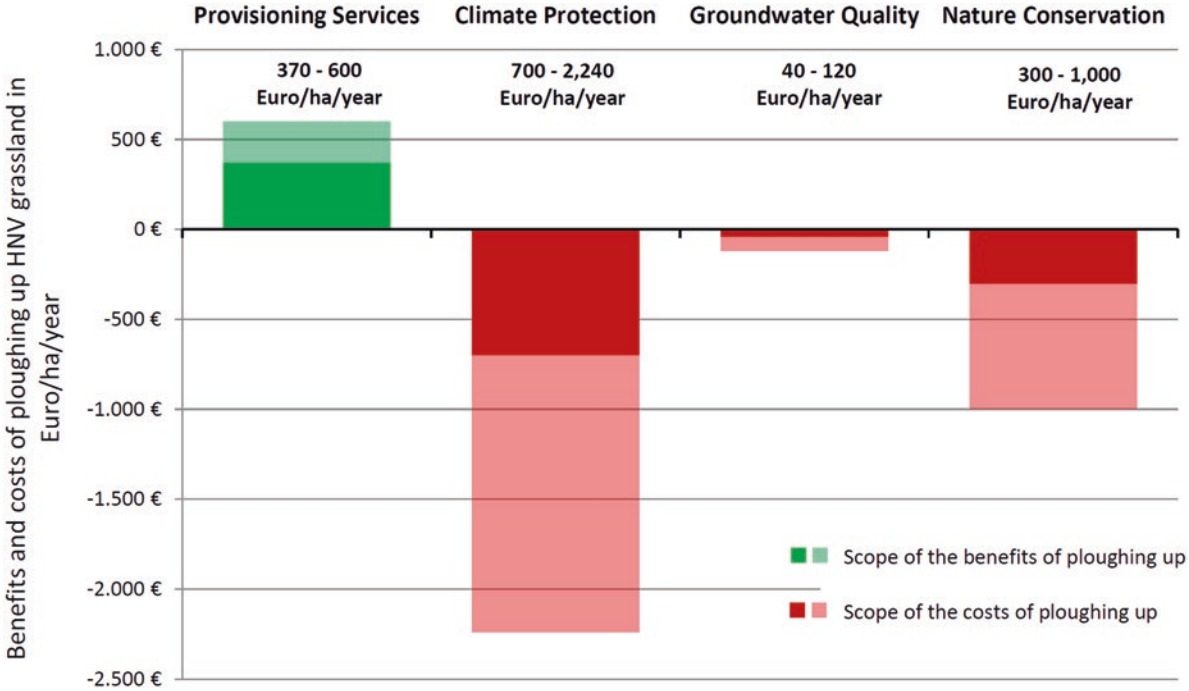

Fig. 52.1 Benefits and costs of ploughing up HNV (high nature value) grassland from a societal perspective. Sample representation of the costs and benefits associated with changing selected ecosystem services, and the willingness to pay for grassland-related nature conservation when ploughing up HNV grassland, per ha and annum [3]

Fig. 52.2 Provision of different ESS (provisioning services, carbon storage, groundwater quality regulation and nature conservation) before (right side) and after (left side) ploughing up HNV-grassland (stylized representation). (*Image courtesy of* Gerd Ostermann, Nature and Biodiversity Conservation Union (NABU), ESS-Icons by Jan Sasse for TEEB)

52.4 Concluding Remarks

Failures to incorporate the broad spectrum of values of nature and its services into decision-making has resulted in perpetuation of land use decisions that degrade natural capital and put the provision of specific ecosystem services at risk. The demonstration of economic value of ecosystem services can be an important support for decision-makers to address trade-offs in land management in a rational manner, correcting the typically biased decisions today, which tend to favor private wealth and financial and physical capital above public wealth and natural capital.

Assessing the costs and benefits of alternative land use options, however, is only a first step. Knowing that grassland conversion is putting valuable ecosystem services at risk will not by itself lead to changes in land use decisions. Translating knowledge into incentives that influence behavior is another. Recent legal reforms to permanently protect grasslands in Germany are a promising step forward, and our case example demonstrates that this is not only important for biodiversity protection, but it is also economically sensible. In other cases, changing behavior and avoiding ecosystem service risks might require further transition management and compensation of (private and/or local) opportunity costs. Including the value of ecosystem services in decision-making can be best achieved if avoiding ecosystem service risks is recognized as an economic opportunity rather than a constraint on development. Demonstrating the full range of ecosystem service values can help to increase awareness and to create compelling rationales for the conservation of our natural capital.

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53 Urban Ecosystem Service Provision and Social-Environmental Justice in the City of Leipzig, Germany

Nadja Kabisch

53.1 Urban Ecosystem Services Provided by Green and Blue Spaces

Urban green and blue spaces provide a number of ecosystem services [1], among them the potential to regulate climate and provide recreational benefits. The ability of a city to regulate local climate is crucial for the well-being of residents. Trees and plants in urban green spaces foster temperature reduction via evapotranspiration processes. Large unsealed and water areas lead to temperature reduction while trees additionally cool areas through shade [2]. Urban vegetation contributes to local and global carbon reduction because trees represent carbon sinks [3]. Indicators such as carbon storage, evapotranspiration, surface emissivity, and cooling potential by tree cover can highlight the performance of related ecosystem services such as air purification and climate regulation as regulating ecosystem services. The exposure to urban green and blue spaces also influences mental and physical health of residents [4]. Urban green spaces provide space outside the building environment to do sports, to relax, to experience nature, and to meet other people. These benefits are summarized under cultural ecosystem services. Recreation may be analyzed through the proxy of per capita green space to show a potential access to urban green spaces. Some cities provide threshold values for per-capita green spaces. The German city of Leipzig aims to provide 10 m^2 of urban green space per person [5]. The application of these indicators has been tested in previous studies [6–9]. However, urban green spaces and related ecosystem services are mostly not equally distributed over a city area and are therefore disproportionately available to the urban population [10]. This disproportionate provision of urban green spaces and the related ecosystem services raises concerns about socio-environmental justice, particularly in cities with a growing population [11] such as in Leipzig, Germany, which is now one of the fastest growing cities in Germany (Box 53.1 and Fig. 53.1). How urban growth is driving pressure on urban green spaces and the provision of ecosystem services is illustrated here based on statistics and GIS maps using the indicators of population density, impervious surface, f-evapotranspiration and per capita green space. F-evapotranspiration is used to show climate regulation as ecosystem service while per capita green space shows the recreation potential of urban green and blue spaces.

Which ecosystem services are addressed? Cultural ecosystem services (recreation).

Regulating ecosystem services (climate regulation).

What is the research question addressed? How is urban growth driving pressure on urban green spaces and the provision of recreational and regulating ecosystem services? Does urban growth lead to distributional inequality between ecosystem service provision and beneficiaries?

Which method has been applied? GIS-analysis based on urban land cover data.

What is the main result? Ecosystem service provisioning values are highest in Leipzig's urban forest and water areas near the floodplains of the city, especially in the southern and northwestern parts. Values are lowest in the dense inner-city areas that are characterized by very high degrees of imperviousness. They are supposed to further decrease because of continuous development pressure from ongoing population growth, raising concerns about an unequal distribution of green spaces among residents. However, there are inner-city areas, which are in high demand, that contain significant shares of urban green spaces. Here, rents are rising, contributing to potential gentrification processes.

What is concluded, recommended? New urban development strategies should focus on integrating residential development with green space development, while also taking into consideration potential raises in property values to avoid "green-gentrification".

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Box 53.1: Leipzig—Location in Germany, Population Development 1933–2014 and Prognosis Until 2030

The city of Leipzig is located in eastern Germany. The city reached its peak population of more than 700,000 inhabitants in the early 1930s. After WWII, which caused a loss of more than 200,000 people, the city entered its socialist period, which lasted until 1990. During the socialist time, the city continued to steadily lose population. Population decline further increased with the fall of the Berlin wall in 1989. Faced with tremendous societal transition after 1989, the city lost a high number of its population, remaining with 530,000 inhabitants in 1989 and 437,000 inhabitants in 1998. Nevertheless, with the end of the 1990s and the beginning of the 2000s, the city started to regain population and is now one of the fastest growing large cities in Germany, with 544,479 inhabitants in 2014 and a growth rate of 2.46% (2013–2014; for comparison: Frankfurt Main had a growth rate of 2.32, Berlin, 1.40, and Munich, 1.54). Population estimates even suggest a further growth of population of up to 722,000 inhabitants by 2030 [13]. Despite being compact, Leipzig contains a large quantity of green spaces with several large parks. The floodplains of the rivers Weiße Elster and Pleiße are covered by old riparian forests and meadows, separating the city into western and eastern parts.

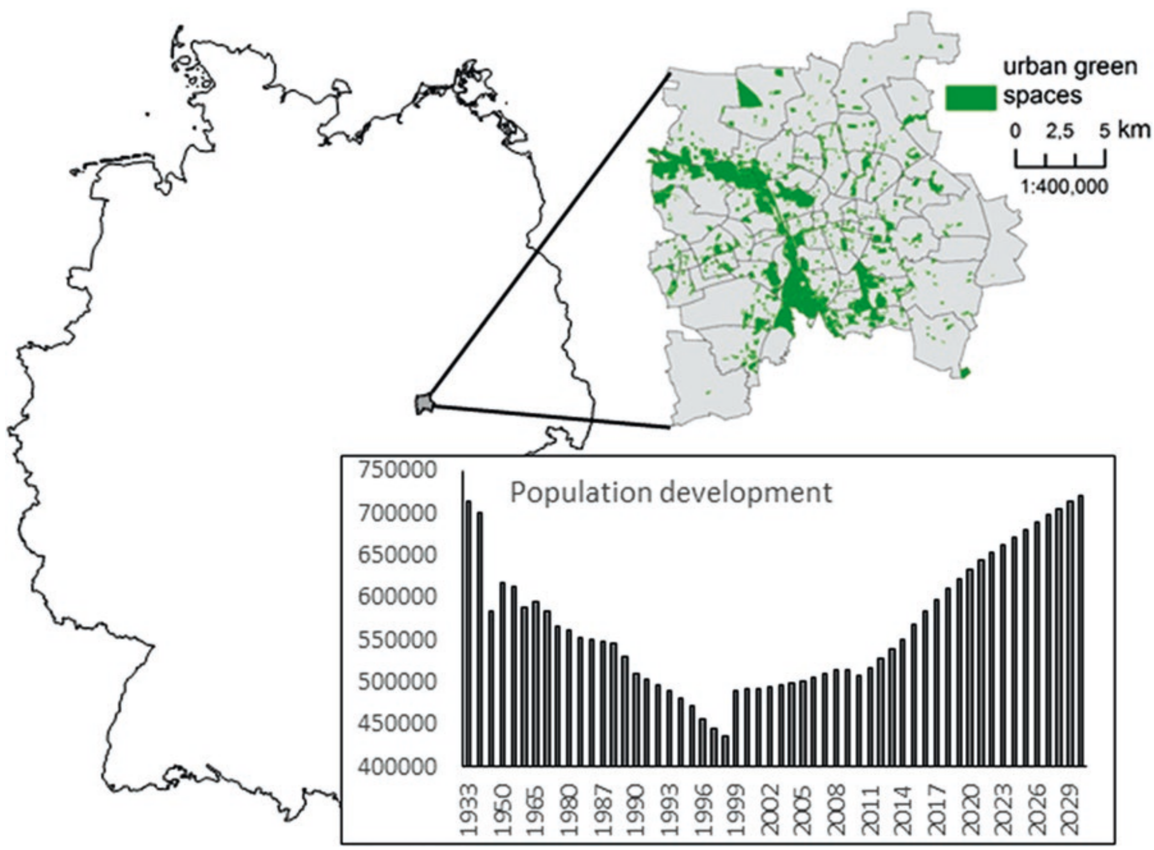

Fig. 53.1 Leipzig: Location in Germany, Population Development 1933–2014, and Prognosis until 2030. Data for urban green spaces were provided by Copernicus [12]

53.2 Urban Residential Development and Ecosystem Service Provision in Leipzig

In the nearly 10 years between 2006 and 2014, the total population of Leipzig increased by nearly 15% (Table 53.1 and Fig. 53.2). Population growth is accompanied by increases in population density (both 10.06%) and by a similar increase in residential area (10.49%). Although increasing in densities, Leipzig also showed a 3% increase in urban green space. The positive increase in urban green space can be related to urban redevelopment projects, greening brownfield sites, and urban park development projects. Despite having increased in total values, per capita green space significantly decreased

Table 53.1 Demographic and Environmental Characteristics for Leipzig, 2006 and 2012/2015[a]

Year	Population number	Population density (inh./km^2)	Residential area (km^2)	Number of flats	Urban green (km^2)	Per capita green space (m^2/inh.)
2006	494,709	1663.67	20.88	314,973	35.31	71.38
2014	544,479	1831.04	23.07	331,748	36.38 [b]	66.82
Change (%)	10.06	10.06	10.49	5.33	3.03	−6.39

[a]Population data were provided by the City of Leipzig [14] and reflects population in 2014
[b]Refers to Urban Atlas data [12]

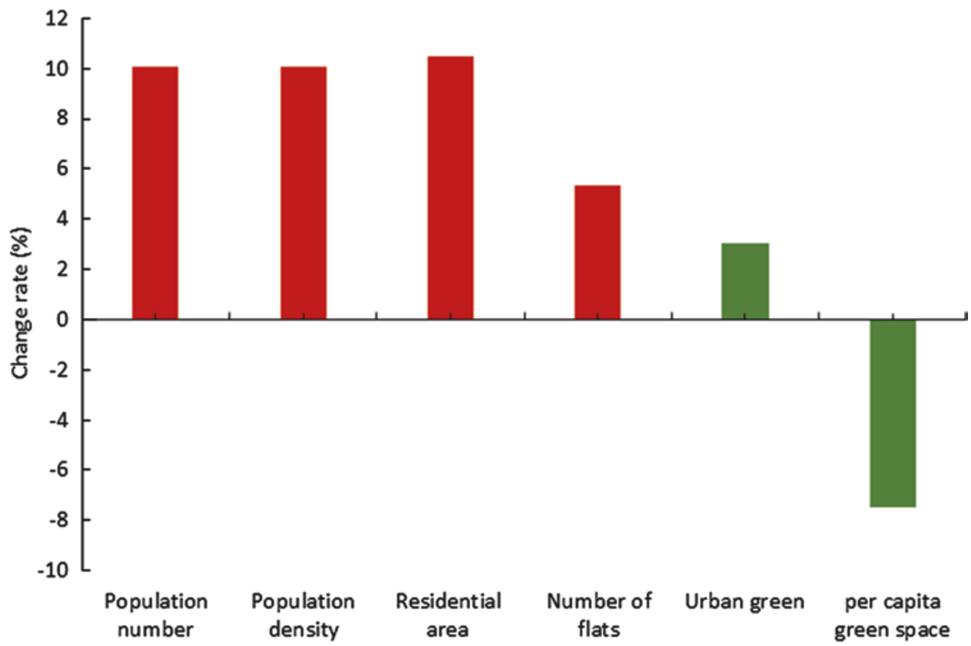

Fig. 53.2 Change rates of demographic and environmental development indicators for Leipzig

by −6.4%, and is projected to decrease even further (assuming the same growth rate of urban green space as between 2006 and 2014 with respect to the new population projection from 2015, which suggests a further increase of about 27% until 2030).

Urban green spaces in Leipzig contribute to the well-being of residents by providing important ecosystem services such as recreational space, but they also contribute to the regulation of the climate. Due to an increase in density, some residents may benefit less from urban green spaces than others. Figure 53.3 shows the distribution of ecosystem service provision over the city.

The influence of vegetated areas on ecosystem service indicators becomes apparent when looking at the map of f-evapotranspiration distribution. F-evapotranspiration is defined as the sum of transpiration from plants and evaporation from bare ground and water areas. The values are highest in Leipzig's urban forest and water areas. Particularly, the large amount of tree coverage in the urban forest areas leads to comparatively high values of f-evapotranspiration near the floodplains of the city, especially in the southern and northwestern parts. In some urban forest areas f-evapotranspiration is highest, e.g., in the triangle-shaped "Tannenwald"—a 251 ha large urban forest in the north of Leipzig. There are other urban parks and urban forests distributed over the whole city area, particularly in the inner-city, with small-scale differences. The indicator of f-evapotranspiration can be regarded as a proxy for the cooling potential of urban vegetation. This is lowest in the dense inner-city areas of the city that are characterized by high degrees of imperviousness. Large amounts of residential, transport, and industrial areas lead to values up to 95% imperviousness. Highest values do appear in the northeastern part, illustrating the location of a large automobile company, and in the northwestern part, illustrating the location of Leipzig's cargo transport center near the Leipzig-Halle airport.

The per capita urban green space map highlights a good provision of the local population by urban green spaces. In most of the city districts, per capita green space is higher than the city's target value of 10 m^2 per inhabitant. However, there are six central city districts with less

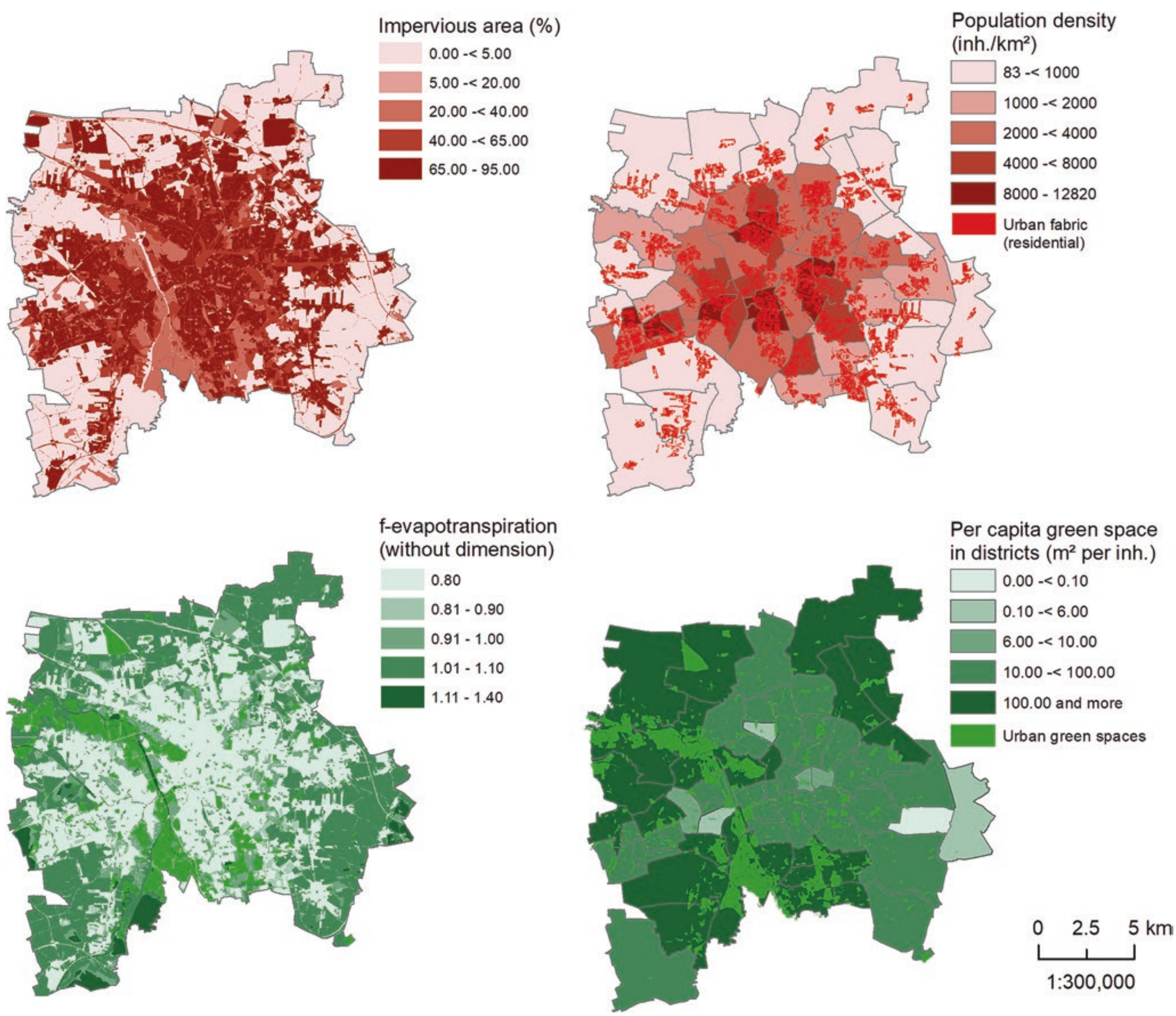

Fig. 53.3 Percentage impervious area (in 2012), population density (2014), f-evapotranspiration (2012), and per capita green space in the districts (2012, right) of Leipzig. Land cover and land use data were provided by Copernicus [12]. The values for f-evapotranspiration and imperviousness were derived from look-up tables based on empirical studies [8, 9, 15]. For the calculation of the recreation ecosystem service, the proxy of per capita green space in the 63 districts of Leipzig was used and is based on the Urban Atlas land cover classes "Green urban areas" and "Forest" [16]

than 10 m^2 per inhabitant. Per capita values are particularly high in the areas near the floodplains (high amount of urban green) and in some outer districts located near the city border. In those districts, population density is comparatively low, which leads to high provision values when green space is available. Accordingly, the central inner-city areas with density values of more than 8000 inhabitants per km^2 represent parts where per capita green space is less than 10 (e.g., the city center).

53.3 Future Directions of Urban Development in Leipzig: A Challenge for Ecosystem Service Provision and Socio-Environmental Justice

The study shows ecosystem service provision—climate regulation and recreation ecosystem services—for the districts of the city of Leipzig. The indicator f-evapotranspiration illustrated how urban vegetation can contribute to the ecosystem services of climate regulation—especially the potential for a city to be cooled by parks and forest areas. In Leipzig, it is the extensive green and forest areas of the floodplains and some larger (partly newly developed) urban parks, particularly in the inner-city area, that lead to the highest indicator values. Comparatively low indicator performance was found in areas near the city center and in some areas in the northern parts due to the highly sealed areas. In the inner-city areas, the renovated buildings from the Wilhelminian period are located near the floodplains. Housing and rent costs are higher in those areas. In other areas across the city, new green space developments, e.g., on former brownfield areas, are also accompanied by rising rents in the residential areas. Local residents may not be able to afford to live there anymore, and have to move to areas with lower percentages of green and water area and with this, perhaps, a lower quality of life. This has happened near the Lene-Voigt-Park in the eastern inner-city area of Leipzig on a former local railway brownfield. With the development of the park at the start of the 2000s, the quality of the housing area increased, new residents with higher incomes moved in, and housing vacancies started to decrease. Consequently, rents increased from 4.5 Euro per m^2 in 2000 to almost 7 Euro per m^2 today. In those areas, socio-environmental justice is an issue and may be linked to future urban development plans of the city when following an environmental justice strategy for the whole city area. Another study in Leipzig showed that residents with higher income had better access to high quality green areas [17].

The available recreational urban green and blue spaces in Leipzig are well developed and frequently used. With the increase in population, however, those spaces are more and more crowded, which may decrease the initial recreational value. When planning future land uses in Leipzig, the city may consider equal distribution of new urban green space development projects and taking measures to safeguard existing green spaces. The quality of new green development projects should especially be considered because quality contributes significantly to ecosystem services supply [18]. A heterogeneous structure with large trees significantly improves indicator values of regulating ecosystem services such as f-evapotranspiration, and leads to air cooling. Moreover, the value as potential recreational area depends not only on the size of the green space but also on the structural composition. For instance, studies of pocket parks in Scandinavian cities found that even the smallest parks, around 0.3 ha and even smaller, can have consequential effects on people's restoration and improve their physical and mental health. This is especially true when such small parks are designed to a certain degree of quality and also take infrastructural components, such as light, benches, and so on, into account [19].

Urban green space planning that takes into account diverse green structures, adapted to the needs and preferences of urban residents, may compensate for the possible low supply of per capita green space in dense inner-city areas. As the population number of Leipzig grows further, the city of Leipzig needs to increase its green space to meet the threshold value in all city districts. To avoid further broadening the gap between the availability of high and low environmental quality in areas with varying shares of impervious surface and urban green spaces, urban planners must evaluate which green areas should be further protected and which open spaces of a comparatively lower recreational value might be used for residential densification. This should be accompanied by a clever development of new urban green spaces on available open spaces such as brownfields. Here, residential development needs to go hand-in-hand with green space development, while taking potential raises in property values into consideration to avoid "eco- or green gentrification" [20, 21]. Small scale green-gentrification has already occured in Leipzig, e.g. the Lene-Voigt-Park, but there are other prominent international examples, such as New York's High Line or the Cheonggye Restoration Project in Seoul, South Korea. New governance models and large-scale public participation processes, as shown in an example in Chicago, US, where community efforts promoted socio-environmental justice [22], might be a step in the right direction to deal with this challenge [23].

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54 Climate Change Impacts on Small Island States: Ecosystem Services Risks and Opportunities

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Mae M. Bruton-Adams, and Heidi Wittmer

54.1 Introduction

Worldwide there are 63.2 million people living in 39 Small Island States [1]. Small island nations are politically, economically, socially, and culturally diverse, yet they also share a number of common characteristics [2]. Due to the small land mass, the often remote location of islands, and limited access to markets, the livelihoods of island communities are highly dependent on the ecosystem services that the terrestrial and marine environment provide. Small Island States are recognized as highly vulnerable to climate change and other non-climate related drivers (Box 54.1), and indeed, many are already experiencing climate impacts [2].

This chapter highlights how ecosystem-based adaptation—informed conservation and management practices that help people adapt to climate change—can enhance the resilience of island communities and reduce ecosystem service risks. Payments for Ecosystem Services are described as a policy option for securing the ecosystem services of island watersheds.

On small islands, coastal and marine ecosystems provide important ecosystem services to local communities, yet are also severely impacted by climate change, natural climate patterns, and human action. Intact coral reefs and mangroves provide coastal protection, timber, filtering pollutants, tourism income, fisheries, and nursery habitat that support coastal and pelagic fish stocks [3] (Fig. 54.1). If coral reefs and mangroves are degraded and lost, the benefits they provide to local communities are also lost. Many of these ecosystem services cannot be replaced. Compensating for the loss of ecosystem services by importing goods (e.g., food, water, and building material) is often not an option on remote islands due to high transportation costs.

Multiple drivers put ecosystem services at risk. Climate change impacts include sea-level rise, ocean acidification, sea-temperature rise, and changes in storm patterns and precipitation, which can increase coastal erosion, flooding, saltwater intrusion, and droughts. In addition, population growth is increasing the demand for food, water, and income, lead-

Which ecosystem services are addressed? Provision: food, freshwater for drinking, fisheries, material, timber;

Regulation: coastal protection, defence against sea-level rise and extreme weather events, erosion control, mitigating landslides and floods, filtering pollutants from freshwater, nursery habitat supporting coastal and pelagic fish stocks;

Cultural: cultural identity, sense of place, tourism

What is the research question addressed? How can a focus on ecosystem services, especially the use of Payments for Ecosystem Services (PES) , help to develop strategies for ecosystem-based adaptation that increase the resilience of ecosystems and communities to climate change in Small Island States?

Which method has been applied? Stakeholder consultations using workshops, interviews, and field visits. Expert consultations and literature review.

What is the main result? Payments for Ecosystem Services are not a quick-fix option to address environmental problems. A detailed scoping assessment considering local conditions and cultural sensitivities is essential to determine whether Payments for Ecosystem Services programs are a viable option, and if so, what safeguards would be required to ensure environmental improvements and social equity.

What is concluded, recommended? While an important tool to support ecosystem-based adaptation, Payments for Ecosystem Services should be considered along with the full suite of tools and policies, including traditional management practices, to support improved ecological and social well-being outcomes. Prior to the implementation of strategies and policies for adaptation it is important to ensure that their effectiveness, efficiency, and equity trade-offs are considered.

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ing to a more intensified use of natural ecosystems. Cultural changes due to globalization and urban migration lead to loss of traditional management practices and can cause overexploitation of ecosystems (e.g., overfishing of fish stocks due to more invasive fishing practices; pollution of freshwater sources and coastal waters with sewage and pesticides; increased soil erosion and sediment deposition on coral reefs caused by degradation of vegetation cover) [4].

Box 54.1: Factors Contributing to the Vulnerability of Small Island States

- Small land mass, remote location; distance from markets; often poorly developed infrastructure; limited funds and human resources;
- Exposure to extreme events;
- Exposure to climate change impacts, in particular, sea-level rise, ocean acidification, sea-temperature rise, changes in storm patterns and precipitation;
- High sensitivity of island ecosystems to climate change, in particular, coral reefs;
- High dependency of local communities on ecosystems and their services

Coral reefs have been particularly affected by the combination of climate change, natural climate patterns, and local stressors (e.g., overfishing, pollution, coastal development). During 2015–2016, a massive coral bleaching event occurred in the Pacific, primarily due to abnormally high sea-surface temperatures, driven by climate change and an El Niño-Southern Oscillation (ENSO) event [5]. Mass coral bleaching has occurred during the past three decades and scientists suggest that nearly 20% of global coral reefs have died since 1950, with another 35% under threat [6]. Ocean acidification is expected to threaten the prospects for reef survival, slowing their growth and weakening the reef framework [7]. While some coral reefs recover, others experience regime shifts to a permanently degraded state. It is predicted that the majority of coral reefs will be lost if climate change continues and local management efforts are not in place to control additional reef stressors [8]. Mangroves are also highly vulnerable to the combination of local human stressors and climate change. Globally, 20–35% of mangroves have been lost since 1980 [8] primarily due to deforestation for coastal development, aquaculture, and resource use. Sea-level rise is expected to exacerbate these impacts. Strategies to protect and restore coral reefs and mangroves, combined with global policies to reduce emissions (e.g., Paris Agreement), are therefore

Fig. 54.1 Coastal fishery in Palau. (*Image courtesy of* Johannes Förster, 2016)

critical if the valuable services that these ecosystems provide are to be maintained. Such strategies will become increasingly important components of adaptation efforts.

54.2 Ecosystem-Based Adaptation

Ecosystem-based adaptation integrates biodiversity and ecosystem services into an overall strategy for adaptation and development that increases the resilience of ecosystems and communities to climate change through the conservation, restoration, and sustainable management of ecosystems [9]. Specifically, it includes strategies for reducing the multiple stressors on ecosystems, thus strengthening their ability to cope with climate change and maintain the delivery of ecosystem services (Box 54.2). Ecosystem-based adaptation can deliver benefits for livelihoods of communities in the near term, while building resilience of ecosystems and communities in the long term. Due to the great dependency of island communities on ecosystem services, ecosystem-based adaptation is particularly relevant for Small Island States.

Box 54.2: Ecosystem-Based Adaptation (EbA)

- EbA includes informed conservation and management practices that maintain or enhance ecosystems and their ecosystem services critical for climate change adaptation.
- On islands, forest conservation in watersheds can be important for water security, as forests provide erosion control and reduce risks of floods and droughts.

54.3 A Ridge-to-Reef Approach to Support Adaptation

On islands, terrestrial, coastal, and marine ecosystems are highly interconnected. Changes in one ecosystem often have knock-on effects on other ecosystems and the services they provide. Effective conservation strategies in Small Island States must therefore address the interconnections within island ecosystems and acknowledge that strategies implemented in one location affect the supply of ecosystem services in another.

In many Small Island States, a high level of connectivity between ecosystems is associated with high topographic relief, allowing a *ridge-to-reef* approach to be used (Box 54.3). This perspective allows conservation managers and local stakeholders to address the links between terrestrial, coastal, and marine ecosystems in adaptation planning and policies (Figs. 54.2 and 54.3). It allows for the identification of potential stressors and ensures that management actions

Box 54.3: Ridge-to-Reef Approach

- On islands, terrestrial, coastal, and marine ecosystems are interconnected, with changes in one ecosystem often having knock-on effects on other ecosystems and their services.
- Managing watersheds from land to sea with a focus on critical ecosystems and their services is an important adaptation option for small islands facing water scarcity as result of climate change.

Fig. 54.2 Participatory modeling and mapping as a tool for improving watershed management and informing strategies for ecosystem-based adaptation from ridge to reef in Palau. (*Image courtesy of* Johannes Förster, 2016)

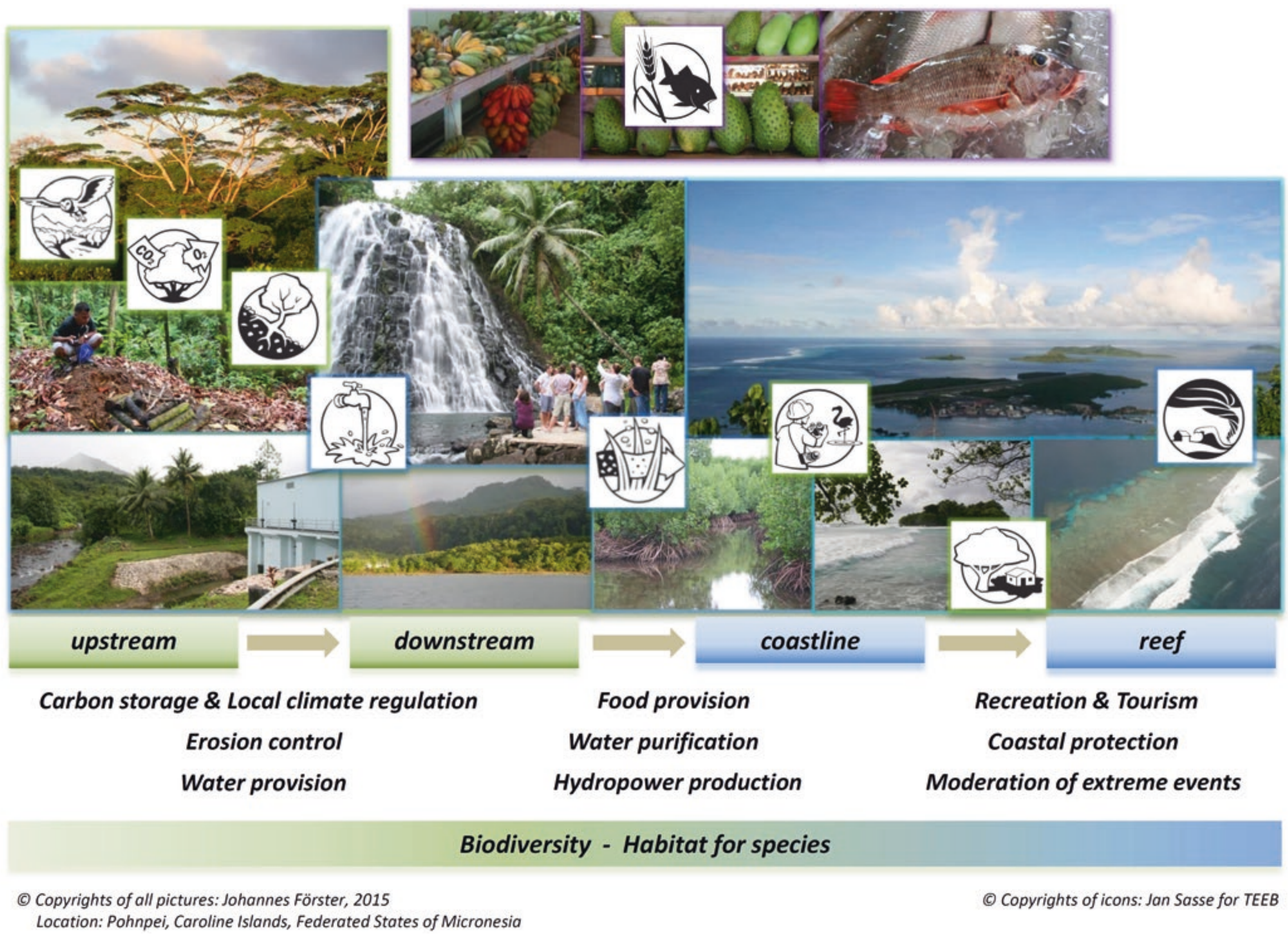

Fig. 54.3 Potential ecosystem services (icons) occurring within a ridge-to-reef gradient from upstream, terrestrial ecosystems (left) to downstream, marine ecosystems (right). Biodiversity enables the provision of ecosystem services along the entire gradient. Ecosystem services related to water provision tend to be more important in upstream areas. (*Images and graphics courtesy of* Johannes Förster, 2016)

applied in one location support the supply of ecosystem services in other locations. As high as the ridgelines, forests protect soils from erosion, improving the quality of freshwater supplies, the provision of timber as building material, and food production in agroforestry (Fig. 54.3). Forests and mangroves also play an important role in reducing the load of nutrients, sediments, and pollutants reaching the sea, which is important for protecting coral reefs. Intact coral reefs and mangroves are critical nursery habitats for maintaining healthy fish stocks [10] and serve as an important source of food and income for island communities. Coastal ecosystems are also the first line of defense against storms, floods, and coastal erosion, providing a natural way of reducing the impacts of sea-level rise and extreme weather events.

The ridge-to-reef approach is relevant for watershed management. Within island watersheds, healthy upstream terrestrial ecosystems deliver a wide range of ecosystem services—notably freshwater services—to downstream communities. Changes in the management of upstream ecosystems have an impact on the delivery of downstream ecosystem services, with further impacts on coastal and marine ecosystems (Fig. 54.3). Sustainable land management that protects and restores upstream terrestrial ecosystems contributes to erosion control, maintains freshwater quality, and mitigates landslides and floods. Hence, a *ridge-to-reef* perspective can support planning for ecosystem-based adaptation and the design of instruments such as Payments for Ecosystem Services [11].

54.4 Using Payments for Ecosystem Services for Ecosystem-Based Adaptation

Policy instruments, such as Payments for Ecosystem Services, can support ecosystem-based adaptation because they provide incentives for the sustainable management of land and marine resources and the provision of ecosystem services (Box 54.4). In principle, those who benefit from the ecosystem services directly or indirectly pay the provider(s)—those who protect or enhance the provision of the services. For example, downstream users (e.g., municipalities) of watershed services may pay upstream farmers to implement soil conservation practices that improve water quality in a downstream reservoir.

However, there is no one-size-fits-all approach. Upstream land management needs to be adapted to site-specific conditions to effectively ensure the delivery of downstream benefits. Monetary and/or in-kind compensations need to be negotiated among downstream users and upstream providers so that sufficient incentives motivate all participating stakeholders to comply with the agreed payment scheme. Often the policy instrument of Payments for Ecosystem Services requires contributions from both sides, which is why Payments for Ecosystem Services schemes are also referred to as a co-investment [12, 13].

Payments for Ecosystem Services programs are implemented primarily to improve the provision of ecosystem services, but they may also be designed to support poverty alleviation, development, job creation, and biodiversity. An analysis of Payments for Ecosystem Services case studies demonstrated that while none resulted in substantial welfare improvements, most supported small gains over opportunity costs [11]. Such gains are particularly important in areas with few alternative sources of income. Payments for Ecosystem Services schemes have also improved human health through access to cleaner, more regularly supplied water, and have strengthened property rights in areas where they were weakly defined. Providing sustained sources of income and water security will become increasingly important for adaptation as climate impacts intensify.

Box 54.4: Payments for Ecosystem Services (PES) in Watersheds

- Ecosystem service (ES) providers (e.g., upstream land user) maintain or improve a defined ES (e.g., erosion control for water quality through conservation and restoration).
- ES beneficiaries (e.g., downstream municipalities) benefit from improved water quality.
- If ES providers are not obliged by laws and regulations to protect ecosystems and their services, payments from ES beneficiaries to ES providers can be an option. This can include monetary compensations or non-monetary incentives, e.g., securing land rights, capacity building for improved land use, access to infrastructure, etc.

54.5 Payments for Ecosystem Services: Case Studies from the Pacific

A partnership between the Helmholtz-Centre for Environmental Research—UFZ, The Nature Conservancy (TNC), and local partners in the Pacific is exploring options for using Payments for Ecosystem Services to support adaptation strategies in Small Island States. This research demonstrated the critical importance of addressing local formal and informal governance for implementing ecosystem-based adaptation.

On the island of Manus in Papua New Guinea, Payments for Ecosystem Services schemes were proposed as a policy option for supporting ecosystem-based adaptation in the Lorengau watershed [14]. However, conflicts over land rights currently impede the participatory process required for ridge-to-reef watershed planning to improve the delivery of downstream ecosystem services. Hence, conflict resolution can play a critical role in implementing long-term adaptation strategies, building resilience of island communities, and reducing ecosystem service risks. Furthermore, there is a risk that emerging threats to ecosystems (e.g., mining activities) will negatively impact watersheds, coastal habitats, and the resources they provide. Therefore, in addition to instruments like Payments for Ecosystem Services, spatial planning for ecosystem-based adaptation needs to address current and emerging threats and involve all sectors, including mining. Otherwise, there is a risk that extractive activities will degrade important ecosystem services and counteract strategies for ecosystem-based adaptation.

On the island of Pohnpei, Federated States of Micronesia, a participatory process that involves traditional leaders and municipalities is under way to improve the management of the Nett watershed. Payments for Ecosystem Services are being considered as a policy instrument to support ecosystem-based adaptation. At the current stage of implementation, it is critical to monitor upstream land management and downstream water quality to ensure that the applied measures maintain and improve the delivery of downstream ecosystem services—specifically erosion control and the maintenance of water quality—to local communities and the town of Kolonia. Such efforts provide important case studies for demonstrating the importance of improving watershed management for ecosystem-based adaptation.

Payments for Ecosystem Services can provide positive conservation and development outcomes in terms of livelihoods, land-use change, incomes, and governance [15], yet there are a number of challenges to consider [16], including permanence, leakage, and perverse incentives [17]. Permanence refers to whether a Payments for Ecosystem Services program generating ecosystem services continues over the long term, and is often dependent on compliance and sustainable financing of the program. Leakage is when environmentally destructive activities are displaced, rather than reduced (e.g., clearing one plot of land to substitute for another that is under protection). The introduction of payments can create perverse incentives with negative outcomes, for example, if traditional conservation practices are only continued under the condition of payment.

Practical obstacles to implementing Payments for Ecosystem Services can also include limitations in design and payment structure; challenges managing trade-offs arising from the need to balance efficiency, effectiveness, and equity; and challenges addressing property rights [15, 16]. Finally, the focus of ecosystem management on maximizing a selection of "desired" ecosystem services could undermine the role of biodiversity conservation, more broadly, for ensuring the stability of the entire ecosystem.

Adverse social impacts of Payments for Ecosystem Services may occur if monetary incentives for conservation replace intrinsic and cultural motivations for conservation and customary practices of sustainable ecosystem management.

Furthermore, there is the risk that the process of designing Payments for Ecosystem Services can be dominated by powerful stakeholder groups, leading to the marginalization of others and creating winners and losers [18]. To address these limitations, researchers suggest important efforts, including: encourage property rights and tenure reform; strengthen the linkages between ecosystem services production and land-use practices; boost private and voluntary sector involvement; improve financial viability; and adequately account for the distribution of costs and benefits among all participants [15].

Hence, Payments for Ecosystem Services are not a quick-fix option to address environmental problems. A detailed scoping assessment considering local conditions and cultural sensitivities is essential to determine whether Payments for Ecosystem Services programs are a viable option, and if so, what safeguards would be required to ensure environmental improvements and social equity. While an important tool to support ecosystem-based adaptation, Payments for Ecosystem Services should be considered along with the full suite of tools and policies, including traditional management practices, to support outcomes of improved ecological and social well-being.

54.6 Conclusion

Dependencies on ecosystem services and related risks need to be assessed and accounted for when developing strategies for climate change adaptation. Policy instruments like spatial planning or Payments for Ecosystem Services are helpful tools to support the integration of ecosystem services into decision-making. Prior to their implementation, it will be important to ensure that their effectiveness, efficiency, and equity trade-offs are considered. Strategies of ecosystem-based adaptation are not only an option for Small Island States, but also for many situations in developed or developing countries.

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55 The Loss of Ecosystem Functions in Riverine Floodplains in Germany

Christiane Schulz-Zunkel, Mathias Scholz, Hans D. Kasperidus, Dietmar Mehl, and Klaus Henle

Which ecosystem services are addressed? Water quality.

What is the research question addressed? What are appropriate approaches to quantify ecosystem functions in floodplains, and how does this support the implementation of suitable management options for those ecosystems?

Which method has been applied? Proxy-based approach.

What is the main result? The actual loss of nutrient (N and P) retention potential in German floodplains is severe.

What is concluded, recommended? In many river-floodplain ecosystems in Germany, special adapted management measures are required to halt this loss. Already, rather simple measures could distinctly enhance nutrient retention capacity.

Riverine floodplains are usually defined as eco-tones or transition zones [1] between terrestrial and aquatic areas influenced by fluctuating groundwater tables and flooding. Hydrologic connectivity is a key process in riverine areas, since it refers to the water-mediated transfer of energy, matter, and organisms within and among the features of river-floodplain ecosystems [2]. Natural floodplains are dynamic and heterogeneous ecosystems showing complex patterns of variation over temporal and spatial scales [3], and they provide an extraordinary amount of unique and important ecosystem functions and thus services like biodiversity support, water quality improvement, flood abatement, and carbon storage [4].

Based on a first nationwide inventory of the loss and status of floodplains along 79 large rivers in Germany [5], we developed methods to quantify and assess four floodplain functions in active floodplains along large rivers: flood retention, nutrient retention, carbon stocks, and biodiversity [6]. These quantifications were mainly based on a nexus of various landscape data like floodplain area, soil types, land use classes, and protected areas. Concerning nutrients, floodplains are important sinks for nitrogen (N) and phosphorous (P), mainly through the processes of denitrification (N) and sedimentation (P). Overall, the retention capacity for nitrogen reaches a maximum of about 41,800 tonnes a^{-1} in all 79 active floodplains (which is up to 9% of the annually transported load in the studied rivers), whereas phosphorous retention is maximally about 1,200 tonnes a^{-1}, which comprises approximately 11% of the annually transported load in the studied rivers [7].

However, the actual decrease in N and P retention potential in German floodplains is severe (Fig. 55.1). The maximum calculated values of N and P retention potential in prior defined units (1 km wide floodplain segments) were taken as a baseline for this evaluation. In most of the investigated floodplain areas the loss of N and P-retention potential ranges between the classes ‘very high’ and ‘high’; this means that the maximum possible retention capacity for N (about 3,450–106,200 kg a^{-1} $unit^{-1}$) and P (about 102–3,300 kg a^{-1} $unit^{-1}$) is reached only in few floodplain areas in Germany [7]. Thus, it can be assumed that in the remaining active floodplains in Germany several drivers have already influenced the provision of ecosystem functions. We conclude that those drivers are mainly the disconnection between rivers and their adjacent floodplains and intensive land use activities, resulting in a noticeable risk regarding the provision of nutrient retention and most likely for the provision of further floodplain functions, too. In conclusion, river-floodplain ecosystems are under dramatic risk in Germany; they underlie continuous anthropogenic changes, with up to 90% of active floodplain areas being already lost and remaining active floodplains being degraded [5].

Thus, in many river-floodplain ecosystems special adapted management measures are required to prevent additional impacts caused by further disconnection between rivers and their adjacent floodplains (including proceeding river bed erosion), intensive land use practices, drainage of wetlands within floodplains, and habitat fragmentation.

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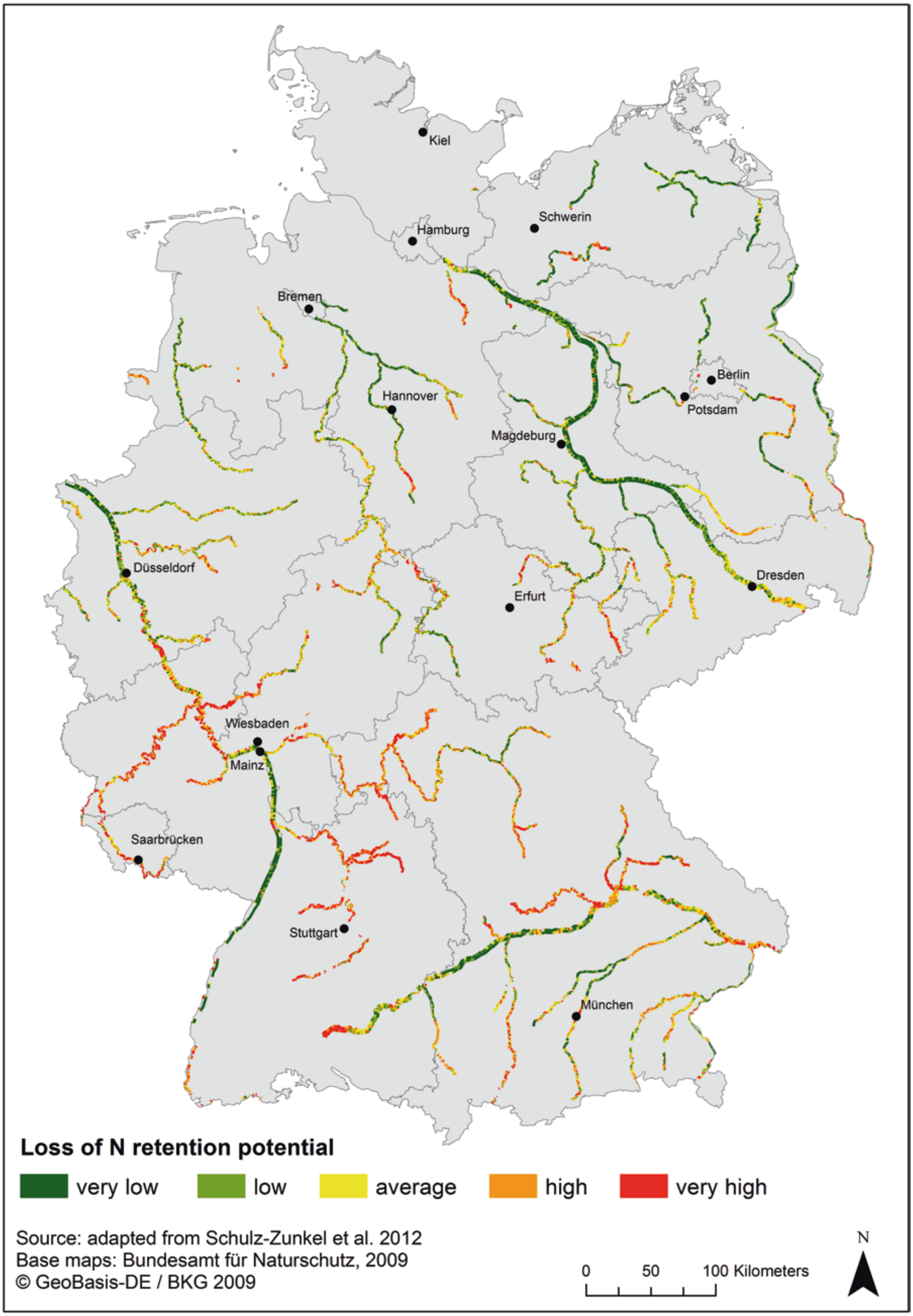

Fig. 55.1 Loss of nitrogen (N)- and phosphorous (P)- retention potential in German floodplains (maximum N-retention potential (baseline) = 106,200 kg a^{-1} unit^{-1}; maximum P-retention potential (baseline) = 3,300 kg a^{-1} unit^{-1}). (*Adapted from* Schulz-Zunkel et al. [7])

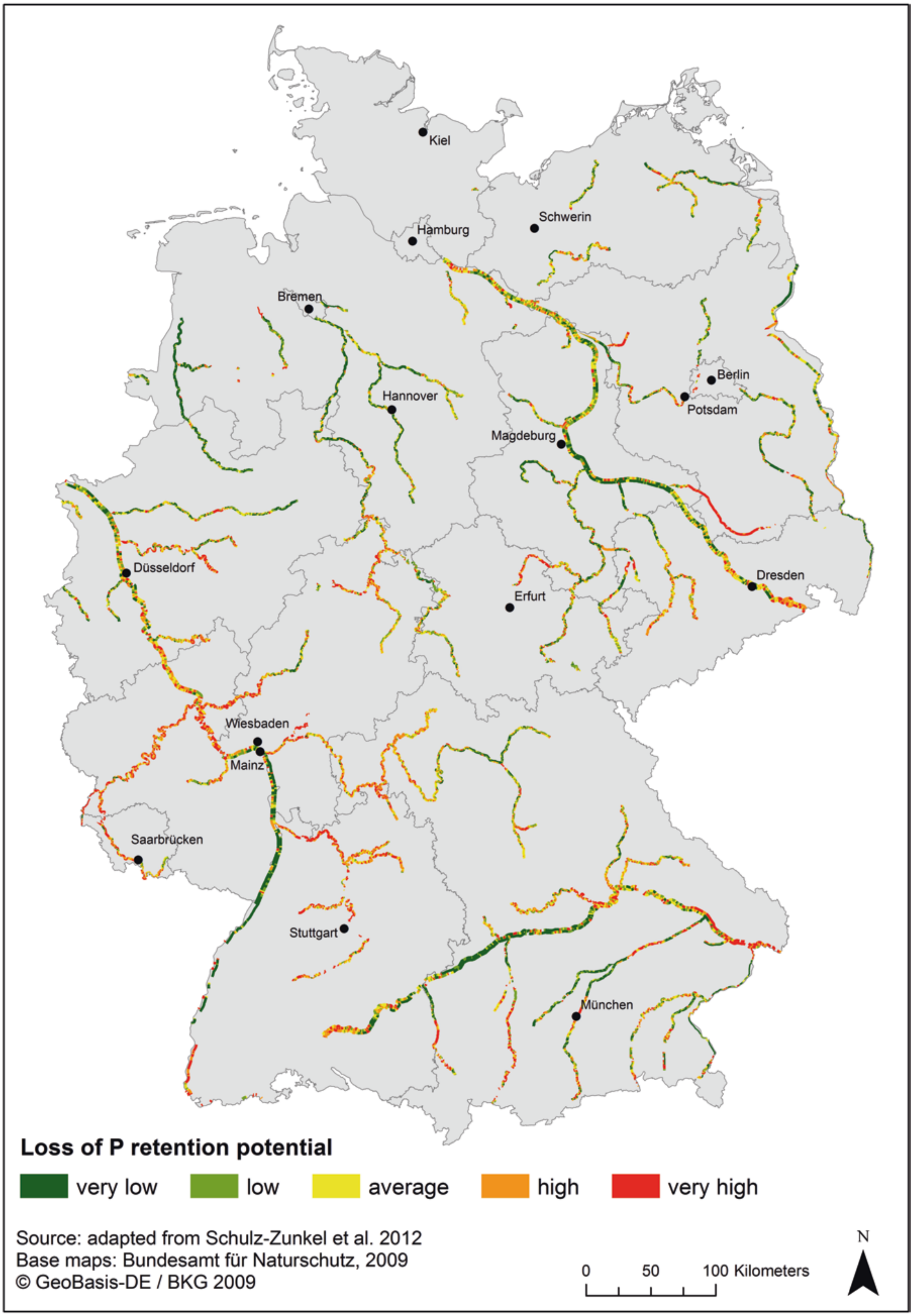

Fig. 55.1 (continued)

Additionally, revitalisation measures are absolutely needed to stop the loss of active floodplain areas. Measures that aim to enlarge active floodplain areas, by e.g. dyke relocations, are particularly important. For example, if the active floodplain areas in Germany were increased by 10% by 2020, which is demanded by the German National Biodiversity Strategy [8], nutrient (N and P) retention capacity could be enhanced by 20%. Moreover, by the same scenario, the availability of typical floodplain habitats would increase by 27% [9]. Additionally, land use changes, e.g. from arable land use to grassland or by initiating the establishment of hardwood forests within such revitalisation areas that underlie wet-dry dynamics, could further optimize the provision of ecosystem functions like nutrient retention or the provision of natural habitats. Such measures are especially important because most natural and semi-natural floodplain habitats make up only a very small part of today's floodplains and are often relicts of a once highly diverse natural landscape. Protecting and developing floodplains thus provides several synergies regarding the goals of the Water Framework Directive, the EU Flood Directive, as well the EU Birds and Habitats Directives [10].

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56 Opportunity Maps for Sustainable Use of Natural Capital

Bart de Knegt, Dirk C. J. van der Hoek, and Clara J. Veerkamp

56.1 Introduction

The Dutch government has the ambition to make its policies more "nature-inclusive" [1, 2]. Nature-inclusive policy recognises the wide range of services provided by ecosystems and biodiversity, aiming for sustainable[1] use of these services. Hence, an important objective of the Dutch government is to more explicitly address these benefits and the effects of interventions on natural capital in decision-making processes. Our study contributes to this objective by identifying areas with opportunities for sustainable use of natural capital. It helps policymakers and other stakeholders to focus their policies and to set priorities.

We developed a method for making *opportunity maps* that identify potential areas to use natural capital in a sustainable way. This method was applied to three cases: sustainable food production, flood safety improvement, and sustainable drinking water production. These cases were conducted within a two-year research programme in the Netherlands that looked at a wide array of local projects that brought the natural capital approach into practice [3]. In these cases, opportunities and barriers were identified for combining sustainable use of nature with economic profitability.

Which ecosystem services are addressed? Provisioning services: drinking water.

Regulating services: soil fertility, erosion prevention, water retention, coastal protection, water purification, pest and disease control, pollination, carbon sequestration.

Cultural services: outdoor recreation, natural heritage.

What is the research question addressed? What are opportunities for a more sustainable use of natural capital?

Which method has been applied? Assessment of opportunities by mapping in combination with case studies.

What is the main result? National maps with opportunities for a more sustainable use of natural capital.

What is concluded, recommended? The opportunity maps presented here offer policymakers and other stakeholders tangible options for sustainable use of natural capital; the maps are thus a step toward incorporation of these options into the decision-making processes.

56.1.1 Local Project 1: Sustainable Agricultural Production

Current agricultural practices are often not sustainable because they are aimed at maximising production at the expense of other ecosystem services such as soil fertility, pest and disease control, and natural heritage. Local Project 1 explored whether an ecosystem services perspective is useful for generating ideas for post-2020 CAP reform aiming to make agricultural production more sustainable while preserving and developing natural capital [4]. For example, farmers may promote soil quality, essential for food production, by adding compost or leaving crop residues on harvested fields to increase soil organic matter content and hence improve soil fertility. These measures also have a positive effect on water storage capacity and carbon sequestration.

56.1.2 Local Project 2: Sustainable Drinking Water Production

Local Project 2 focused on sustainable drinking water production [5]. In several water infiltration zones, groundwater quality does not meet the standards for drinking water, due to the presence of pollutants such as pesticides and nitrate from

[1] By *sustainable* we mean avoiding depletion of (natural) resources so that present needs are met without compromising the ability of future generations to meet their own needs.

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agricultural land use. The local projects showed that a transition can be made from a "curative" farming system to a "preventative" farming system. In preventative farming systems, use of chemical fertilisers and pesticides is limited; as a result, groundwater quality increases, simplifying the production of clean drinking water without loss of crop production.

56.1.3 Local Project 3: Flood Safety Improvement

Local Project 3 explored the ecological and economic added value of sustainable use of natural capital for long-term flood safety improvement [6]. Owing to climate change, flood risk along Dutch rivers has increased in recent years as peak flows have increased, and many river dykes no longer meet current safety standards. Nature-based solutions such as coastal dunes, intertidal zones, and riparian wetlands not only promote flood safety but also offer opportunities for nature conservation and outdoor recreation.

Having identified the opportunities and barriers within the different local case studies, we would now like to know whether these are also relevant elsewhere in the Netherlands. We therefore designed a method to translate these local project outcomes to national level, using *opportunity maps*.

56.2 Method

56.2.1 Primary and Secondary Ecosystem Services

The local projects [4–6] showed that sustainable use of ecosystem services offers opportunities for mutual improvement of natural capital and the economy through nature-based solutions. Ecosystem services that directly contribute to the sustainable delivery of "final" ecosystem services such as food, drinking water, and flood safety, are called "primary" ecosystem services. Conservation and improvement of natural capital will increase the supply of these primary ecosystem services, providing the basis for sustainable food and drinking water production and long-term flood safety. In many cases, this offers parallel opportunities for improvement of other, "secondary" or co-benefitting, ecosystem services. Table 56.1 shows the primary and secondary ecosystem services identified in the three local projects.

56.2.2 Shortages in Ecosystem Services

To identify which areas in the Netherlands offer opportunities for sustainable use of natural capital, we mapped the potential demand and supply of primary and secondary ecosystem services relevant to the three local projects (Table 56.1). In this context, demand is determined by the need for these ecosystem services by users, while supply is determined by the size, quality, and spatial configuration of the ecosystems delivering these services. Shortages occur in locations where supply is lower than demand [7, 8]. For example, there is a shortage in outdoor recreation opportunities if the supply of nature areas and attractive landscapes near cities does not meet the demand [7]. Solving these shortages—for example, by creating riparian zones along water courses to filter agricultural runoff—provides opportunities to increase sustainability.

Table 56.1 Primary and secondary ecosystem services relevant to the three local projects

Study	1. Sustainable agricultural production	2. Sustainable drinking water production	3. Flood safety improvement
Primary Ecosystem Services:	Pest and disease control Soil fertility Pollination Soil erosion control	Pest and disease control Soil fertility Water purification	Coastal protection flood safety along rivers
Secondary Ecosystem Services:	Natural heritage Outdoor recreation Carbon sequestration Water storage Water purification Drinking water production	Natural heritage Outdoor recreation Carbon sequestration Water storage Pollination	Natural heritage Outdoor recreation

56.2.3 Parallel Opportunities for Improvement of Ecosystem Services

Shortages in primary ecosystem services (Table 56.1) relevant to the three projects were mapped and combined into a single map for the Netherlands. The same was done for secondary ecosystem services. When the two resulting maps are combined, four outcomes are possible:

1. Areas with shortages in both primary and secondary ecosystem services: Solving shortages in primary services provides parallel opportunities for improving the supply of secondary ecosystem services (purple colour in Figs. 56.2 and 56.4).
2. Areas with shortages only in primary ecosystem services: Solving these shortages does not provide parallel opportunities for secondary ecosystem services (blue colour in Figs. 56.2 and 56.4).
3. Areas with shortages only in secondary ecosystem services: Solving these shortages benefits only secondary ecosystem services (green colour in Figs. 56.2 and 56.4).
4. Areas without shortages, where demand for both primary and secondary ecosystem services is met by local supply. "Surpluses" are possible (green colour in Figs. 56.2 and 56.4).

56.2.4 Priority

As a final step, we identified areas in the Netherlands where investment in natural capital is most urgently needed, focusing on sustainable drinking water production and long-term flood safety. For example, the priority of investing in sustainable drinking water production is highest in drinking water infiltration zones where groundwater concentrations of pesticides, nitrate, and related parameters exceed water quality standards [9]. Similarly, improvement of flood safety has the highest priority in places where primary flood defences fail the current safety standards [10]. In the latter locations, spatial interventions to improve flood safety offer unique opportunities for nature-inclusive solutions. There are many other ways, of course, to assess this priority.

56.3 Results

We present the results for the opportunity map for sustainable agricultural production (Local Project 1) step-wise, while for the other two cases we present only the final opportunity maps.

56.3.1 Opportunity Map for Sustainable Agricultural Production

As a first step, we created maps of potential supply and demand for each of the four primary ecosystem services relevant to sustainable food production (Table 56.1). This is illustrated in Fig. 56.1 using the example pest and disease control. The demand for this ecosystem service applies to crops that are susceptible to agricultural pests and diseases, such as potato, cereal, and fruit crops. Spatial data for where these crops are grown are available at field scale. Potential supply was assessed by mapping the ecosystems in the vicinity of these crops and estimating their capacity and spatial reach for delivering the service of pest and disease control. Next, maps of supply and demand were combined to identify areas of shortages (or surpluses). Explanation of supply, demand, and their combination of other ecosystem services are described as well [7, 8].

The resulting maps of supply and demand for each of the four services were stacked into a single image (Fig. 56.2, top left map), which shows where shortages (and surpluses) exist in primary ecosystem services required for sustainable food production. In this map, areas where the supply of ecosystem

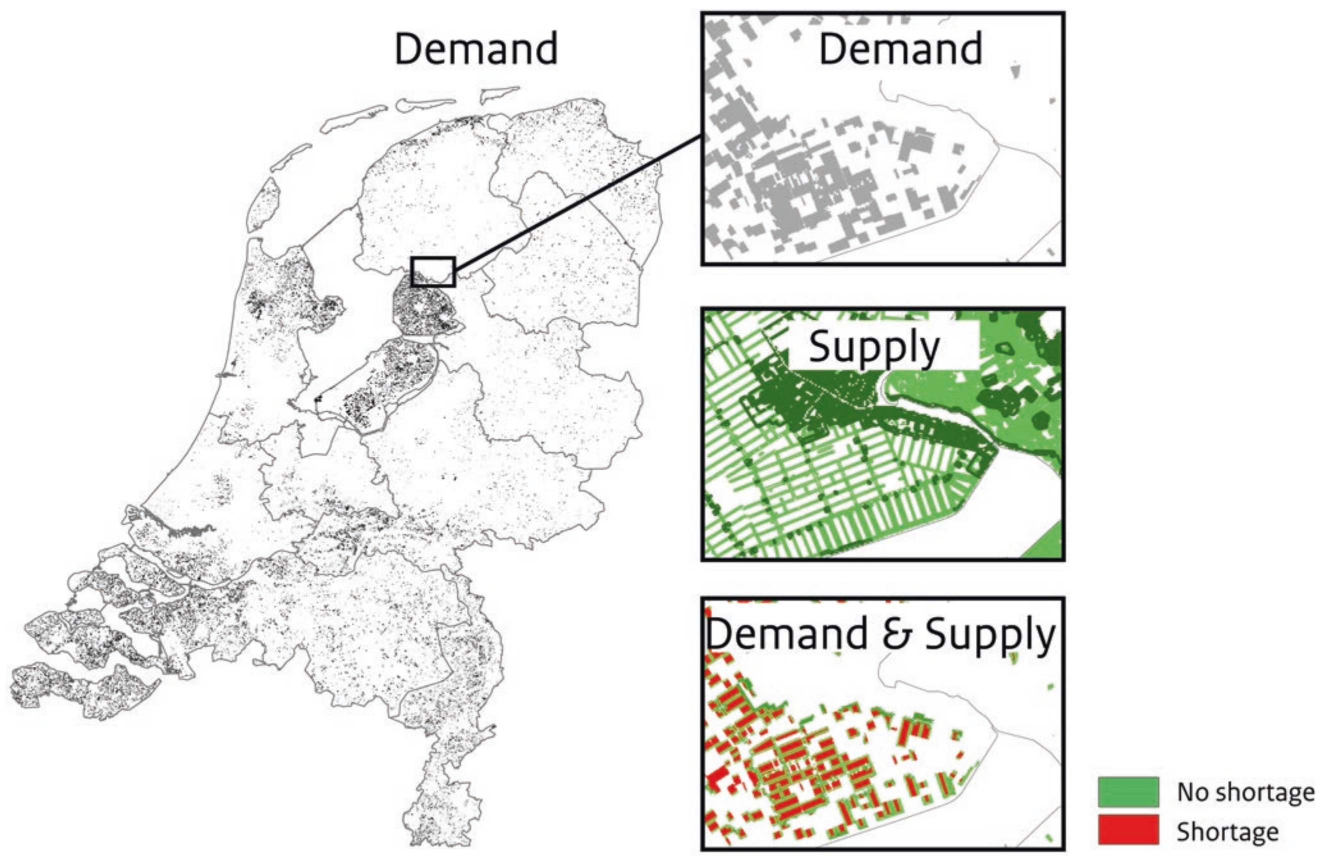

Fig. 56.1 Demand and supply of the ecosystem service “pest and disease control”

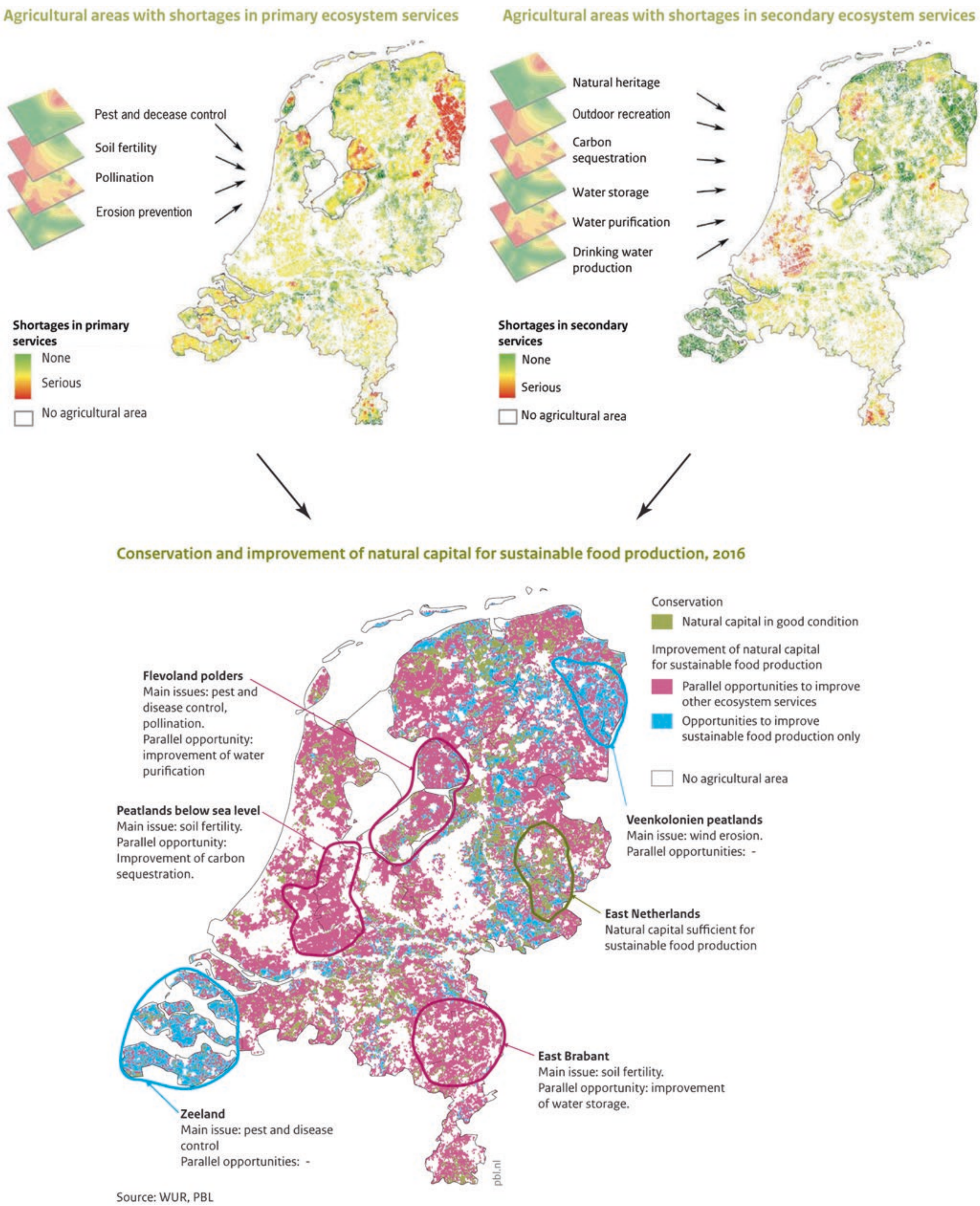

Fig. 56.2 Opportunity map for conservation and improvement of natural capital for sustainable food production

services is currently sufficient for sustainable food production are indicated in green. In yellow and red areas, one or more ecosystem services are in short supply, with most shortages in red areas. Solving these shortages offers opportunities to improve the agricultural production systems while promoting sustainable use of natural capital.

The top right map shows where shortages exist in secondary ecosystem services. The lower map in Fig. 56.2 shows that, on about 20% of total agricultural land, such as in the eastern part of the Netherlands, the current condition of local natural capital is sufficient to allow sustainable food production. The key priority in these areas is conservation of the ecosystems to secure the present and future supply of ecosystem services. Here, the agricultural sector and knowledge institutions could investigate together how to prevent degradation, and how to make effective use of the ecosystem services available (which, in many locations, is not yet being done). The purple areas in Fig. 56.2 (about 65% of total agricultural land) indicate areas where one or more primary ecosystem services are in short supply, and where solving these shortages offers parallel opportunities for improving secondary ecosystem services; in East Brabant, for example, improving soil fertility will also improve the water storage capacity of the agricultural land. Finally, the blue areas (about 15% of total agricultural land) indicate areas where opportunities are limited to improvement of primary ecosystem services only. In these areas, such as Zeeland, it is primarily up to the agricultural sector to take measures to make food production more sustainable.

56.3.2 Opportunities for Sustainable Drinking Water Production

There are many opportunities to scale up the results of this local project to other parts of the country. In about 55% of the total area of water infiltration zones (total demand) in the Netherlands, drinking water production is already based on sustainable use of natural capital; the water infiltration zones are located in nature areas (e.g., coastal dunes) or areas with organic farming. However, in the remaining water infiltration zones (45% of total area), groundwater quality has been impaired by agricultural fertiliser and pesticide. In many of these zones (30% of total area), improvement of groundwater quality offers parallel opportunities for other ecosystem services, such as carbon sequestration, water storage, and outdoor recreation. Here, solving ecosystem service shortages might be a shared interest of farmers, drinking water companies, and other stakeholders, such as water management authorities and nature conservation organisations. In other zones (about 15%), improvement of ecosystem services for drinking water production offers no parallel opportunities for other ecosystem services. In the latter areas, it is up to the drinking water companies and farmers to find sustainable solutions.

To prioritize intervention to improve the sustainable drinking water production, a map of priority was created (Fig. 56.3). This map shows which drinking water infiltration zones in the Netherlands are at risk because they fully or partly (75% level) exceed water quality standards due to high concentrations of pesticides, nitrate, sulphate, and nickel in the groundwater [9]. In these areas, investment in the conservation and improvement of natural capital has the highest priority.

56.3.3 Opportunity Map for Flood Safety Improvement

Natural capital in the form of coastal dunes and elevations along rivers plays a key role in long-term flood safety in various parts of the Netherlands (green sections in Fig. 56.4, left-hand map). However, there are many areas along the coast and rivers where primary flood defences (i.e., dykes) could be improved by making better use of natural capital. For about 60% of the total length of primary flood defences (total demand; purple sections in Fig. 56.4, right-hand map), nature-based solutions for flood safety improvement will offer parallel opportunities for other ecosystem services, such as natural heritage and outdoor recreation. However, in other areas—particularly along the Maas and IJssel rivers and the inner coasts of Zeeland—flood safety improvement will not offer such parallel opportunities (blue sections in Fig. 56.4; about 30% of the total length of primary flood defences). In the latter areas, secondary ecosystem services such as room for outdoor recreation are important, but are not in short supply at those locations.

Figure 56.4 (right-hand map) shows that many primary flood defences in the Netherlands fail the current safety standards [10]. Improvement of water defence infrastructure is most urgent, but also offers opportunities for long-term flood safety solutions while using natural capital sustainably. Conservation and restoration of, for instance, shallow areas, tidal marshes, riparian zones, reed beds, and nature areas could make a valuable contribution to long-term flood safety in these priority areas.

56.4 Use and Application of Opportunity Maps

The opportunity maps presented here provide tangible options of sustainable use of natural capital for policymakers and other stakeholders and are thus a step to enable their incorporation into the decision-making processes.

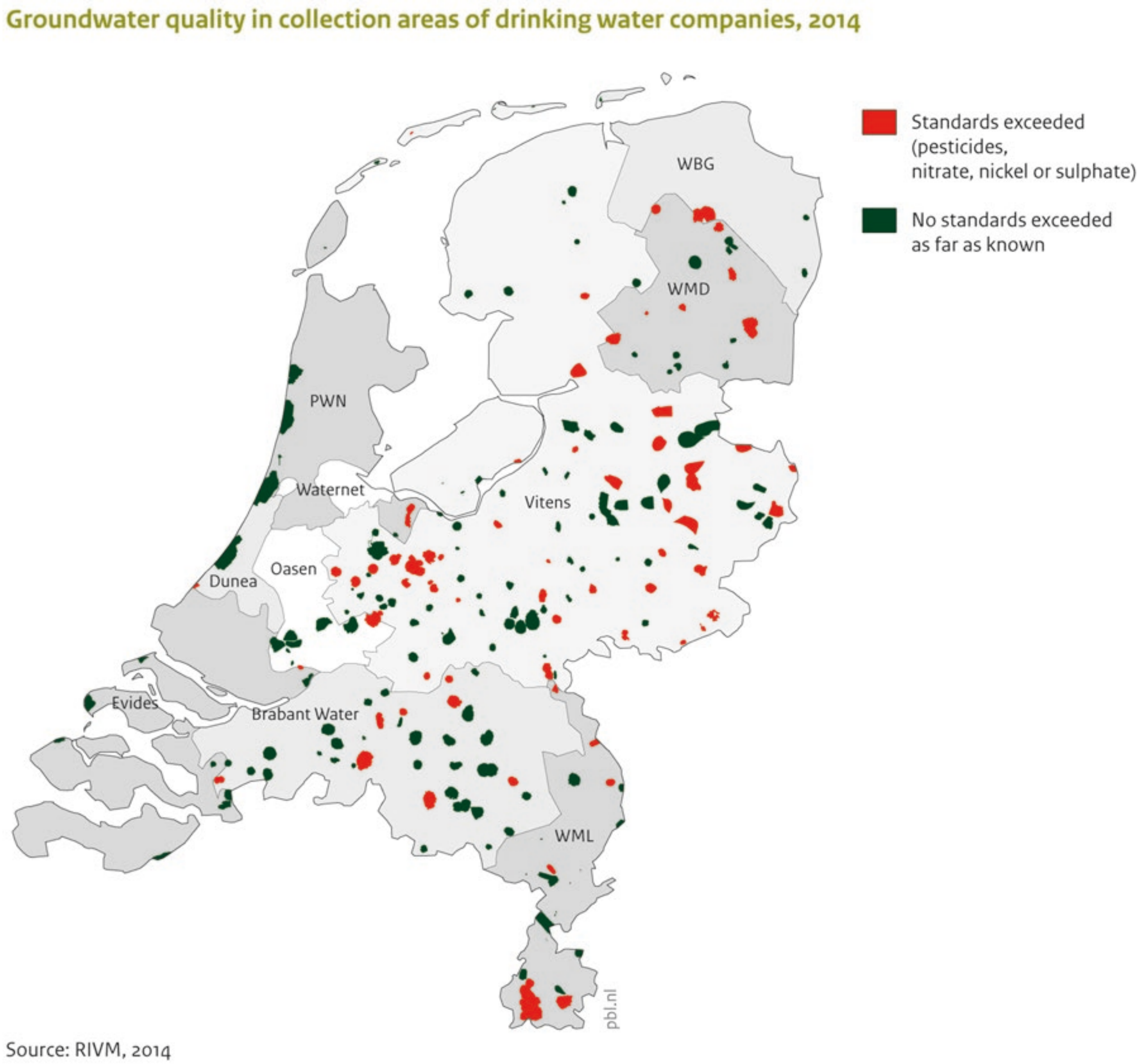

Fig. 56.3 Map of priority to invest in natural capital because concentrations of substances caused by unsustainable food production (pesticides, nitrate, nickel, or sulphate). Red areas indicate places where concentrations already exceed standards [9]

These opportunity maps could be applied to a whole range of other issues such as, for example, securing the delivery of ecosystem services that are essential to sustainable economic growth or climate change adaptation and mitigation. In the context of nature conservation management, the maps might also be used to identify locations where natural capital might contribute to the protection of biodiversity to meet (inter) national biodiversity targets.

The opportunity maps presented in this paper are the first result of a national assessment using various data sources and expert estimates [7]. The challenge will be to integrate the use of these maps in, for example, policy development and spatial planning strategies. Actual realisation of the opportunities identified will be a process in and of itself, requiring the commitment and willingness of relevant stakeholders to make the necessary transitions.

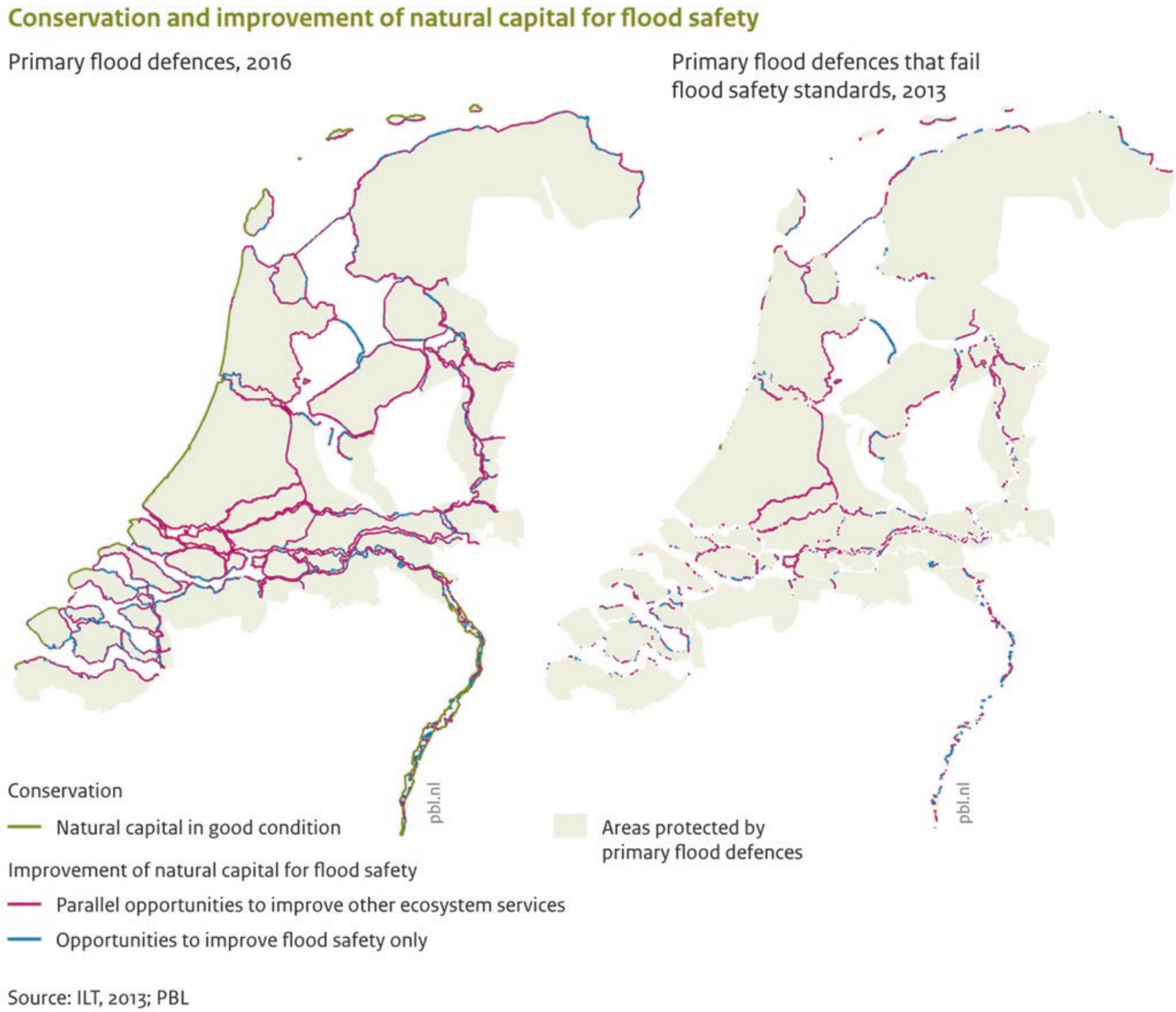

Fig. 56.4 Opportunity map (left-hand map) for conservation and improvement of natural capital for flood safety. The green sections indicate primary flood defences based on natural capital in good condition. The purple sections indicate areas where flood safety improvement based on natural capital would offer parallel opportunities for improvement of other ecosystem services (natural heritage, outdoor recreation). The blue sections indicate areas where flood safety improvement offers no parallel opportunities for other ecosystem services

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57 Rice Ecosystem Services in South-East Asia: The LEGATO Project, Its Approaches and Main Results with a Focus on Biocontrol Services

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57.1 Introduction

LEGATO stands for "Land-use intensity and Ecological EnGineering – Assessment Tools for risks and Opportunities in irrigated rice based production systems."

To advance long-term sustainable development of intensive land-use systems against risks arising from multiple aspects of global change, LEGATO quantified ecosystem functions (ESF) and the services (ESS) generated from them in irrigated rice landscapes in South-East Asia. The focus was on local as well as regional land-use intensity (including the socio-cultural and economic background) and biodiversity, and the potential impacts of future climate and land-use change.

In particular, LEGATO investigated the interactions between irrigated rice and the surrounding landscapes in the

Which ecosystem services are addressed? Provisioning services (soil nutrients).

Regulating services (biocontrol, pollination).

Cultural services (recreation, cultural identity, tourism).

What is the research question addressed? How can provisioning, regulating, and cultural ecosystem services in irrigated rice production systems in South-East Asia be improved through modifications particularly of land use-related drivers (intensity, pesticide application, etc.)?

Which methods have been applied? Field studies on nutrients, decomposition, and insect dynamics; Designed experiments; Species inventories; GIS-analyses based on urban land cover data; Questionnaires.

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What is the main result? (a) Ecological engineering is a promising approach for more sustainability in intensive rice production landscapes;

(b) Planting of flower strips around rice fields is an example of ecological engineering that increases biodiversity and provides habitats for natural antagonists of rice pest species, thereby reducing the need for insecticide use; and

(c) Participatory approaches are needed to convince farmers to switch to more sustainable management practices.

What is concluded, recommended? (a) Ecological engineering; (b) Planting of flower strips; (c) Participatory approaches.

light of ecological engineering (as an emerging discipline, concerned with design, monitoring, and construction of ecosystems). The overall objective was the elaboration and testing of generally applicable principles for the improvement of provisioning, regulating, and cultural ecosystem services through modifications particularly of land use-related drivers (intensity, pesticide application, etc.). For a general project overview, see Settele et al. [1].

57.2 Ecosystem Services Under Study

Following the framework of the Millennium Ecosystem Assessment [2], LEGATO defined supporting services as ESF and dealt with selected characteristic elements of the three service strands defined by the MA: (a) Provisioning (PS): nutrient cycling (e.g., Schmidt et al. [3] and crop production (e.g., Klotzbücher et al. [4]); (b) Regulating (RS): biocontrol and pollination (e.g., Westphal et al. [5]); (c) Cultural Services (CS): cultural identity and aesthetics (e.g., Tekken et al. [6]). Studies were conducted mainly in two countries: Vietnam and the Philippines, in landscapes along a gradient reflecting changing geo-climatic and land-use intensity, and, where possible, also cultural conditions (for more details see Settele et al. [1]).

57.3 Study Regions and Sites for Field-Based Research

As solutions elaborated within the funding scheme were expected to have a model character (i.e., these should be transferable to other regions [7], LEGATO opted for a trans-regional and international approach. The Philippines and Vietnam are particularly suitable, as they represent both important similarities and differences in a region of critical importance for global development. The topological similarities allowed the selection of comparable transects in both countries along gradients that reflect different land-system archetypes [7] with changing geo-climatic conditions and land-use intensities, and also different levels of socioeconomic and cultural diversities (see Table 57.1). They range from mountain areas to fertile hilly lowlands to low-lying, flood-prone high production areas. In both countries, the mountain areas are characterised by the terrace agriculture of indigenous peoples (see, e.g., Fig. 57.1). The final selection of study regions was also based on results of focus group discussions and interviews with stakeholders, resulting in the selection of seven regions (each 15 × 15 km^2), three in Luzon/Philippines, three in northern Vietnam, and one in the Mekong delta in southern Vietnam. For their locations see Figs. 57.2a, b. For further details on climates, land uses, and soils, see Klotzbücher et al. [4].

In each of these regions, 10 core sites (all of them are rice fields; i.e., 70 rice fields in total) were selected to ensure collection of sufficient data for scientifically-profound, comparative analyses (see also Klotzbücher et al. [4]). The 10 sites made up 5 site-pairs, with: (a) one site of each pair being located in an agriculturally more intensively used setting (structurally poor, more homogenous surroundings with more than 50% of rice fields in an area within a radius of 100 m around the centroid of the patch/site), henceforth called "monoculture rice field"; and (b) a second site, at a distance of 300–1000 m from the previously described site, with more heterogeneous surroundings (structurally rich, less than 30% rice fields in an area within a radius of 100 m and a higher proportion of non-intensively used areas such as house gardens, fallows, forests, etc.), henceforth called "structurally diverse rice field." The selection was based on the hypothesis that higher structural diversity leads to higher biodiversity, enabling us to test biodiversity effects on irrigated rice agro-ecosystems. During the project, the above two categories were complemented by "agroforest" fields without rice in the vicinity of most of the 10 sites (resulting in 5 triple-sites; see Fig. 57.3).

Site selections were made in close consultation with local administrators and LEGATO collaborators. This also made it possible to include in the project local communities that differed in some socio-cultural and economic characteristics. Inclusion of local communities allowed comparative socio-cultural analyses, of community responses to generally similar environmental conditions, thus providing a good baseline for social research (e.g., comparison of topographic pictures to identity terraced landscapes).

Table 57.1 LEGATO regions (selected along geologic-climatic gradients) and their categorisations along a spectrum of land-use intensity (e.g., workload, agro-chemical input), landscape structural diversity (large monocultures vs. small fields with other habitat elements in between), and cultural diversity (traditional knowledge and practices applied; diversity of ethnic groups). (Source: qualitative assessment based on authors' knowledge of the regions before the start of the project; if two levels are marked for one region this indicates a range of levels with the region)

LEGATO region (code and name of province)	Land use intensity			Landscape structural diversity			Cultural diversity	
	low	medium	high	low	medium	High	Low	high
Philippines (Luzon island)								
PH_1: Laguna		■	■		■		■	
PH_2: Nueva Ecija			■	■	■		■	
PH_3: Ifugao (Fig. 57.1)	■					■		■
Vietnam								
VN_1: Hai Duong		■	■	■	■		■	
VN_2: Vinh Phuc	■	■			■		■	
VN_3: Sapa	■					■		■
VN_4: Tien Giang (Fig. 57.5)			■	■	■		■	

Fig. 57.1 LEGATO landscape at Batad within region "PH_3 Ifugao" (compare Table 57.1). These Amphitheatre-like terraces are part of the UNESCO world heritage sites of Ifugao province, North Luzon, Philippines. (*Image courtesy of* J. Settele, 2012)

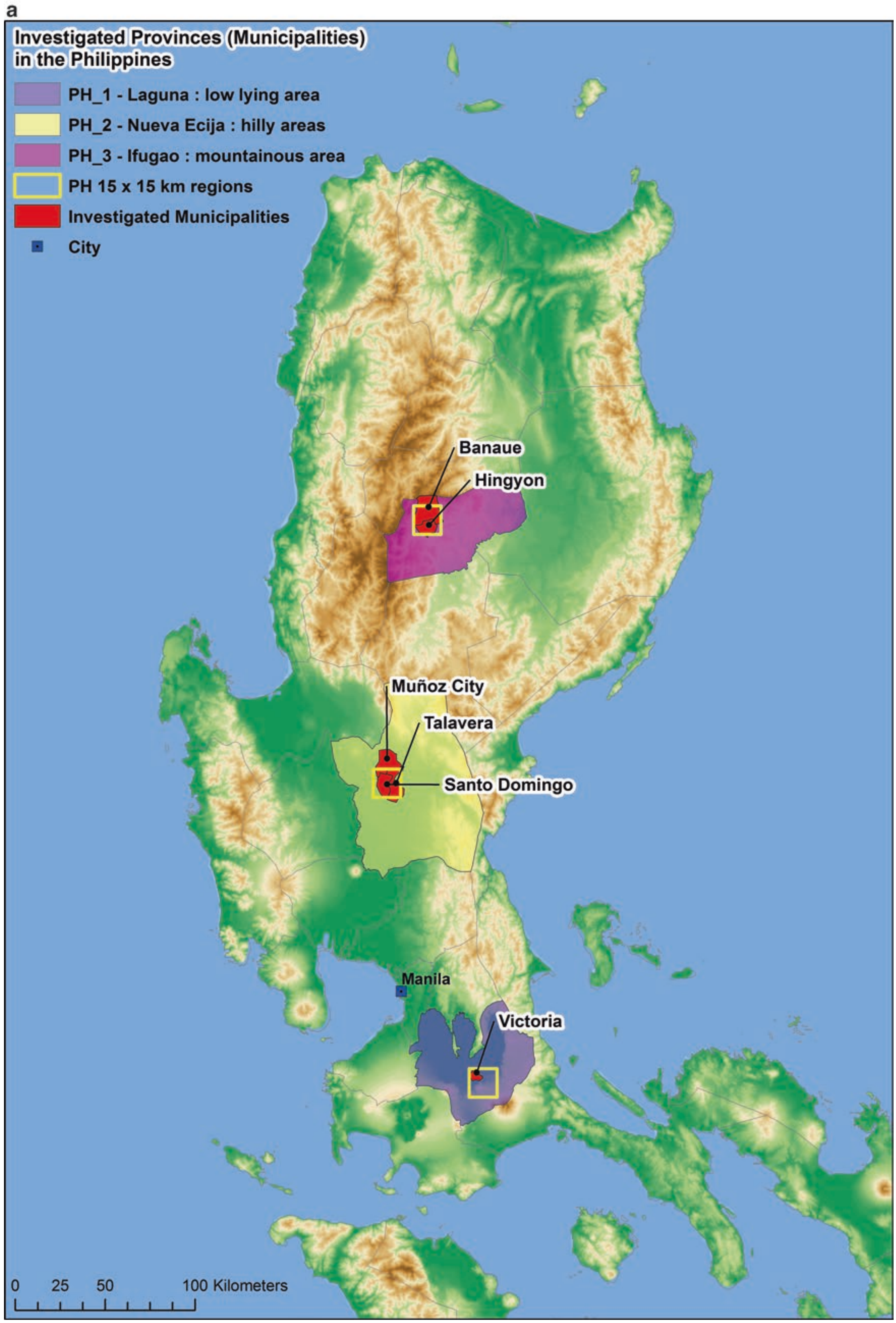

Fig. 57.2 (**a**) Geographical distribution of the 3 LEGATO research regions in the Philippines (© Harpke/Grescho, UFZ). (**b**) Geographical distribution of the 4 LEGATO research regions in Vietnam. (*Image courtesy of* Harpke/Grescho, UFZ)

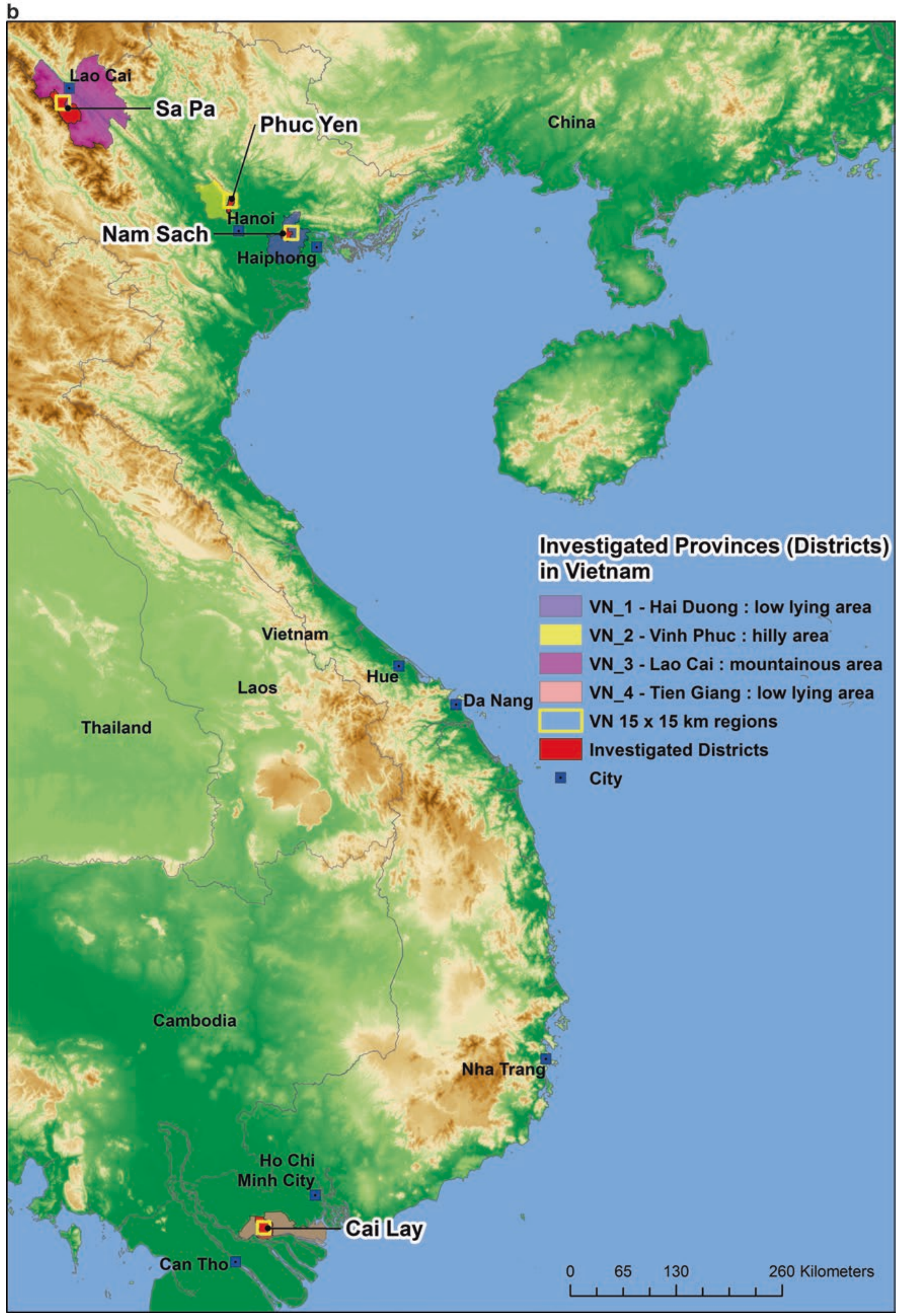

Fig. 57.2 (continued)

Fig. 57.3 The three habitat types investigated in LEGATO – exemplified for PH_1, Laguna, Philippines; (**a**) "monoculture rice field" (image *courtesy of* C. Sann); (**b**) "structurally diverse rice field;" (**c**) "agroforest." (*Images* **B** and **C** *courtesy of* A. L. Hass)

57.4 Co–Design of Research and Co–Production of Knowledge

As with most projects, LEGATO was organised into Work Packages (WPs) and followed the work flow shown in Fig. 57.4. Core elements of the project structure were the feedback loops, particularly those in relation to the co-design of research with stakeholders that directly influence recommendations and implementations – often via several feedback loops (co-design: feedback WPs 1 with WP 2/3; and co-production for practical outputs, like e.g., Ecological Engineering: feedback WP 5 with WPs 2/3 and 4).

Our experience with this approach was very positive in terms of openness of farmers to our research activities and our approach has been analysed in more detail by Görg et al. [8] and Spangenberg et al. [9]. The main conclusions of these studies were that large integrated research projects are necessary to address the complexity of nature-society interactions within biodiversity research and beyond. Such large-scale research projects create challenges in terms of management and knowledge integration, but also offer promising opportunities for transdisciplinary research if managed properly. However, for an appropriate integration of knowledge across different disciplines and with stakeholders, two-way communication between researchers and practice partners is critical to the development of solutions to complex societal problems [10]. Such two-way communication goes beyond the more linear outreach and dissemination activities that are often involved in conventional project management.

Therefore, a particular characteristic and highlight of LEGATO was the close collaboration with farmers and other local stakeholders (partnerships which are necessary to achieve real progress in the field of sustainable land use and biodiversity conservation [11]). The research sites were all located in farmers' fields, selected during close interaction with these farmers and managed throughout the project with their enormous support. The project interacted with a range of different groups of stakeholders: The number of people involved in each group is roughly estimated to include the following (see also Förster et al. [12]):

- Government institutions (Agriculture: 10; Environmental protection: 5; Municipal administration: 20; Tourism and culture: 20; general/top level: 20).
- Private sector (Business catering for the local market: 40; Business catering for the national and international market: 5).
- NGOs: 5.
- International organisations: 20.
- Individual farmers and land owners: 500.

57.5 Outputs: The Example of Biological Control Services

As core output, LEGATO has developed guidelines for optimising ESF/ESS and their stabilisation under future climate and land-use change, which will affect South-East Asia in particular. LEGATO examined the potential for ecological engineering to achieve this, and tested its implementation and transferability across regions. The latter was achieved through inclusion of, for example, local agricultural agencies and extension services as partners. Implementation included assessments of ESS risks and opportunities in the light of changes in land-use intensity, biodiversity, and climate.

One of the key problems in intensively managed irrigated rice production systems is the high level of pesticide use [13], which can lead to health problems and declining biodiversity. Lower biodiversity can aggravate problems with pest outbreaks, because insecticides often have a greater impact on the more sensitive natural antagonists of pest species, such as predatory spiders or parasitoid wasps, than they do on major rice pest species such as planthoppers and leafhoppers, particularly when they have developed insecticide-resistance.

Ecological engineering aims to address this problem by providing habitats for the natural antagonists of rice pests and thereby reducing the need for insecticide applications. One technology of ecological engineering used by LEGATO was the planting of flower strips along rice field margins [5]. These flower strips, according to local farmers, also have the added benefit of improving the aesthetic appearance of rice landscapes (Fig. 57.5). However, this technology can work only if the whole farmer community of a region either stops using pesticides or uses them in a very restricted way as part of an integrated pest management (IPM) approach. As shown in a few first exploratory case studies, another way to increase biodiversity and improve sustainability is to leave a few square meters in a paddy unplanted. This allows dragonfly populations to establish themselves; dragonflies are natural antagonists of the pests. A permanent pond with water vegetation could be installed, which functions as a stable source of dragonflies to (re-)populate the nearby paddies.

Fig. 57.4 LEGATO overview structure and work flow – the basis for co-design and co-production

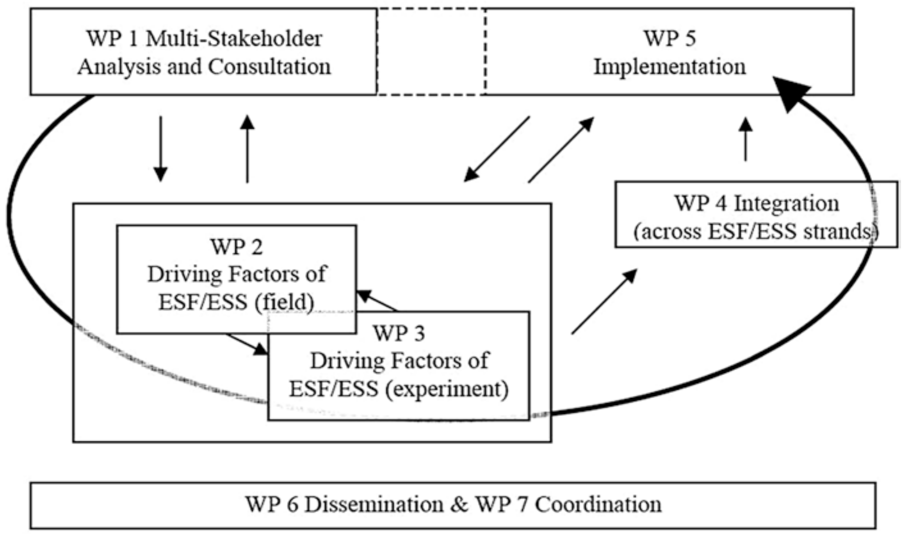

Fig. 57.5 Flower strips along rice field margins during a LEGATO school PR event in southern Vietnam. (*Image courtesy of* Le Huu Hai)

To convince farmer communities to revert to more sustainable management practices requires effective communication and education, often against powerful lobbying from the pesticide industry. One LEGATO approach was entertainment education, which uses mass media campaigns to spread the information among farming communities [5].

The first results from this approach are encouraging, particularly among farmer communities in southern Vietnam, where ecological engineering techniques have been adopted especially by female members of the communities. Analogous to advertisements of agro-chemicals, mass media campaigns can be highly efficient and cost-effective, and have the potential to reach and motivate thousands of farmers to implement ecological engineering practices. In southern Vietnam, insecticide spraying was reduced by over 50% in response to the dissemination of information in posters and leaflets [14]. A follow-up mass media campaign (locally named "Three Reductions, Three Gains") has been developed and reached more than three million farmers in South and Central Vietnam [15]. Within LEGATO, a TV series to promote ecological engineering was launched in Vietnam (Fig. 57.6). The TV series sought to modify farmers' attitudes and practices, and is estimated to have reduced insecticide use among farmer-viewers by 19% [16].

To identify management deficits and achieve improvements, it is important to understand that rice farmers often base their decisions on simple rules of thumb [17]. Scientific information must therefore be distilled into simple and easy to communicate rules. LEGATO suggested a heuristic communication Scheme [13] to structure complex information and convey it in a simplified but meaningful way. These insights can be presented in several linked rules to explain complex biotic interactions, the importance of different groups of service-providing animals, and the synergistic management of their services [18].

Implementing ecological engineering as a dominant practice in irrigated rice production systems therefore requires continuous support of farmer communities using participatory approaches More research is also needed, e.g., to identify the most suitable plant composition for flower strips in different regions.

57.6 Some Key Messages

Based on the experience from and investigations performed within LEGATO, we present some important key messages:

(a) Ecological engineering is a promising approach for more sustainability in intensive rice production landscapes.
(b) Planting of flower strips around rice fields is an example of ecological engineering that increases biodiversity and provides habitats for natural antagonists of rice pest species, thereby reducing the need for insecticide use.
(c) Participatory approaches are needed to convince farmers to switch to more sustainable management practices.

Fig. 57.6 Launching of TV series on education entertainment in southern Vietnam. (*Image courtesy of* M. Escalada)

Initial successes have been achieved, but continuous support of farmers and additional research is required for long-term adoption of sustainable management practices.

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58 Impacts of the EU's Common Agricultural Policy on Biodiversity and Ecosystem Services

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Which ecosystem services are addressed? Biodiversity and Landscape structures.

What is the research question addressed? Which Common Agricultural Policy (CAP) measures are effective and efficient in improving support and maintenance of biodiversity and landscape structures?

We investigated the Ecological Focus Area (EFA) within Greening (Pillar I) and the Agri-Environment and Climate Measures (AECM) (Pillar II) in Lower Saxony.

Which method has been applied? A quantitative assessment of biodiversity impacts by calculating "ecology scores" by using weighting factors from a survey among ecologists published in Pe'er et al. [1].

What is the main result? We found relatively low effectiveness of both Ecological Focus Area (EFA) and Agri-Environmental and Climate Measures (AECM) in the West, and potential for complementarity in the East of Lower Saxony. Overall, the figures indicate that AECM are more efficient.

What is concluded, recommended? 1. Substantially reform or even abolish Greening and EFA in Pillar I.

2. Strengthen and reform the AECM in Pillar II.

3. If Greening is maintained, improve policy integration between EFA and AECM.

58.1 Pressure and State

58.1.1 Pressure

Over the past decades, the EU's Common Agricultural Policy (CAP) has been supporting farmers. At the same time, one could observe a sharp decline in farmland-biodiversity. Especially the application of the price-support Scheme (1957–1992) had a direct environmental impact by incentivising intensification. The MacSharry-reform (1992) and the Fischler-reform (2005) gradually reduced this negative impact by transforming the financial support from highly distortive price support into decoupled direct payments.

58.1.2 First Response

For many years, deficits in farmland biodiversity and Ecosystem Services were addressed by Agri-Environment and Climate Measures (AECM) in Pillar II ("Rural Development Programme"), and received much attention by ecologists trying to evaluate their effectiveness (see, e.g., Batáry et al. [2]). The AECM are partly co-financed by the EU and the member states.

58.1.3 State

Despite the different reforms of the Common Agricultural Policy and despite increased spending on AECM, the decline of farmland biodiversity and of ecosystem services continues, especially in Central and Eastern Europe, where the Common Agricultural Policy was introduced in 1999 and, more broadly, with the accession of 12 new member states in 2004 and 2007. Multiple studies documented these impacts on, e.g., farmland birds [3].

A reliable indicator to describe the status of biodiversity and ecosystem services on arable land in Germany is the "Biotope Index" developed by the Julius Kühn Institute (JKI), which utilizes aerial photographs to measure the repository of small structural elements such as landscape elements, buffer strips, and small grassland plots [4]. Focusing on the Federal State of Lower Saxony in Germany (Fig. 58.1), we present the average values of the JKI-Biotope Index on the county level (Fig. 58.2). Values range from 7 to 47, depending on the share of landscape features and buffer strips and, accordingly, low-to-high provision of ecosystem services.

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Figure 58.2 shows low scores in the Börde and Leinetal regions in southern Lower Saxony, regions with favourable soil conditions and high farming intensities. The JKI-Index has a weak representation for grassland area, as landscape elements attached to grassland are not included, species-rich grasslands are not differentiated. However, for an analysis of structural deficits on arable land, this index is appropriate.

58.1.4 Societal Pressure

Societal pressure on the Common Agricultural Policy has been growing, with a demand for public goods in return for public money [8], namely, that Common Agricultural Policy secures not only food but also environmental services (i.e., ecosystem services) provided by farmland, such as clean water, pollination, pest-control services, aesthetic values and cultural services of traditional agriculture, and many others.

58.1.5 Second Response

The recent reform of the Common Agricultural Policy (CAP) of 2013 responded to these public demands by "Greening" about 30% of the CAP's Direct Payments (Pillar I). A key greening-measure to address the biodiversity decline is the Ecological Focus Area (EFA), aiming to protect a range of features that are considered beneficial for biodiversity and ecosystem services. However, various authors doubted the potential success of the greening [9], and in a recent evaluation of EFA design and implementation in Europe, key EFA options that were scored highly by agro-ecologists received low uptake by farmers (about 25% of the total EFA area [1]). This raises concern, although, to a certain extent, lower effectiveness might be somewhat compensated by larger area cover.

We analyse the impact of Ecological Focus Areas versus Agri-Environment and Climate Measures on biodiversity and ecosystem services, by calculating an "ecology score" that considers the potential impact of both measures on biodiversity. We investigate whether the greening measures (EFA in Pillar I) and AECM (Pillar II) are comparable or

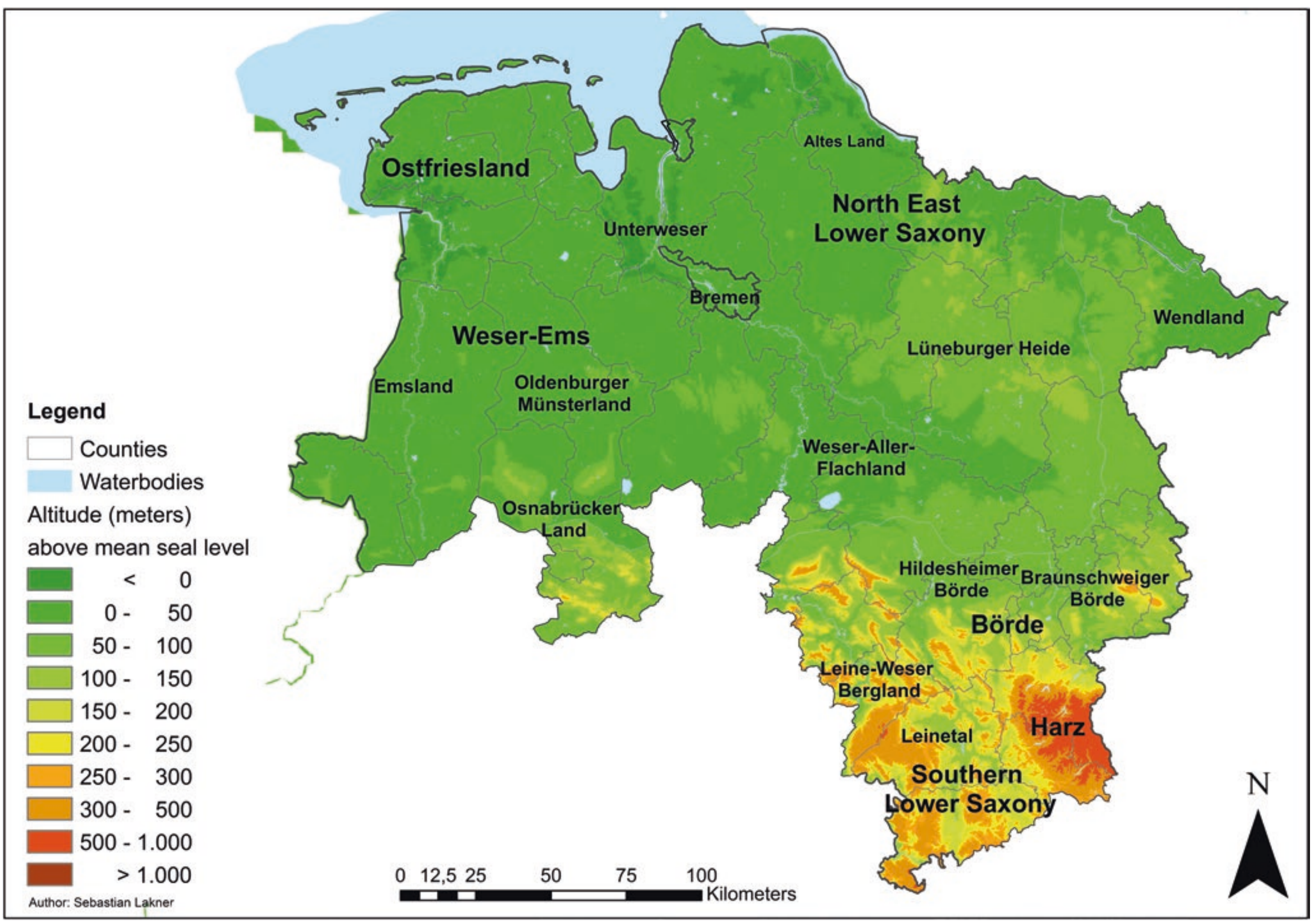

Fig. 58.1 Topography of Lower Saxony. Shapefiles for waterbodies, altitude, and administrative borders by FACG [5–7]

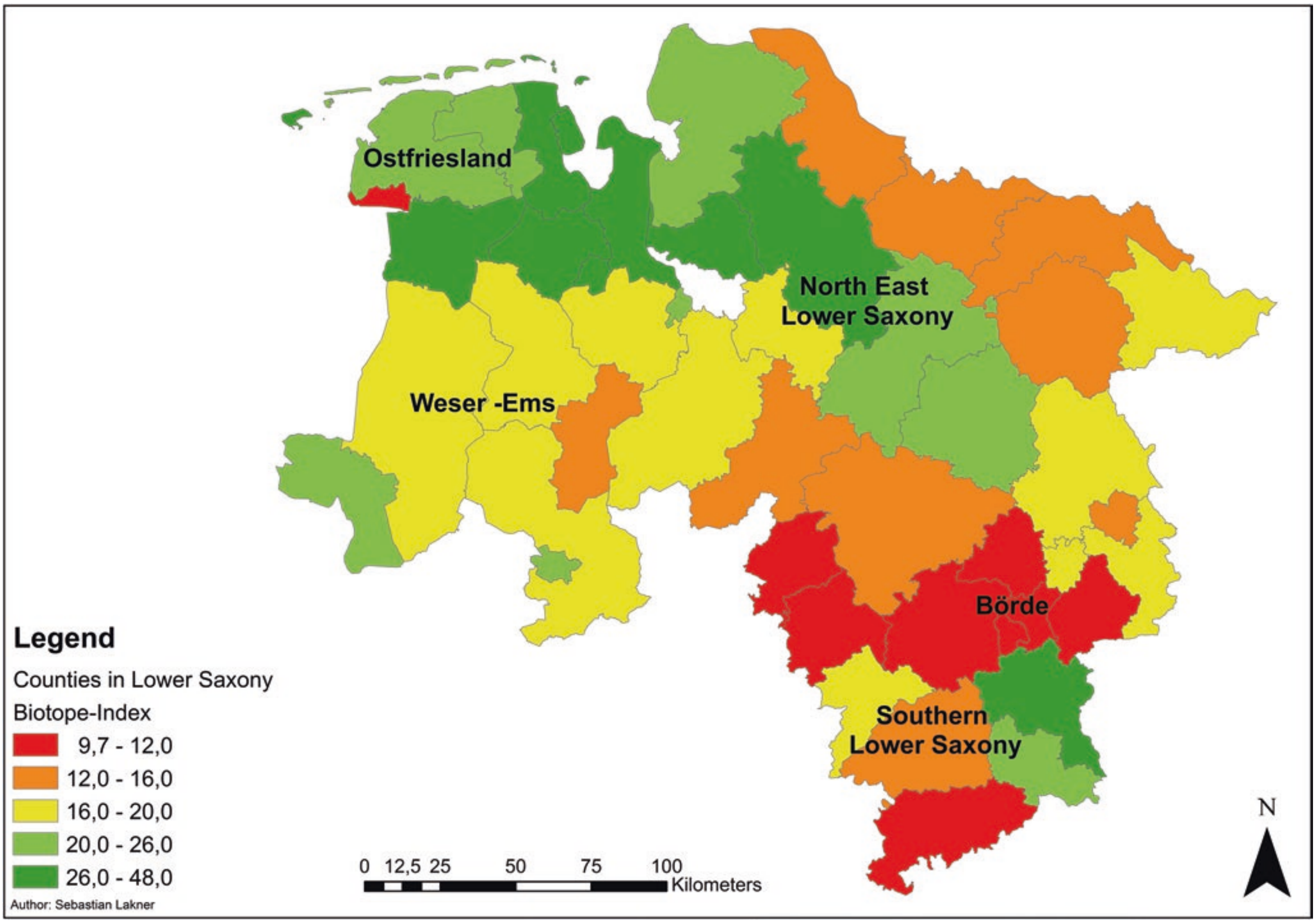

Fig. 58.2 Ecosystem Services measured by the Biotope Index in Lower Saxony, based on the data from JKI-Biotope Index [4]. Colours represent the proportion of landscape structures and buffer strips. Shapefiles for administrative borders from FACG [6]

complementary, and especially what can be learned from their spatial distribution. Lower Saxony is interesting to study because it has the largest gross value added of agriculture in Germany and the second largest agricultural area [10].

58.2 Methods

The ecology score is based on an online survey conducted among 88 ecologists in 17 European countries. Experts were asked, first, to evaluate each Ecological Focus Area (EFA) option based on its potential contribution to biodiversity protection (using scores ranging from +5 to −5 [1]). Based on these evaluations, we derive an aggregated "standardized ecology score" for each of the counties in Lower Saxony. This was done for both EFA and Agri-Environment and Climate Measures (AECM): Some AECM options are comparable with EFA, such as catch crops, flowering strips, buffer strips, and landscape elements, albeit AECM set stricter requirements.

We calculated a sum of ecology scores for EFA or AECM per county (Table 58.1, column [3]), using the area of arable land allocated to these measures (in hectares, column (1)) multiplied by the EFA score (column (2)), in order to obtain an ecological score (column (3)). We divided the county's ecological score (line (4)) by the respective arable area (line (5)). This resulted in the final "standardized ecology score" for each county (line (6)). This calculus was done for both EFA and AECM, to reflect the total anticipated effect of both per 100 ha arable land.

58.3 Results

Figure 58.3 shows the standardized ecology score of the Ecological Focus Areas (EFAs) in the different counties of Lower Saxony. One can see a lower ecological effectiveness in the western counties (light green), and higher coverage of EFA in the northern and eastern regions (dark green) with highly specialized arable production. These include arable regions with very fertile soils (Börde, especially the counties of Wolfenbüttel, Helmstedt), near the Harz-mountains (Goslar, Osterode), but also in arable regions with less fertile soils in North Eastern Lower Saxony (Uelzen, Gifhorn, Harburg, Rotenburg). The gradual spatial pattern of improved-effectiveness toward the East, with a North-South "corridor" of higher effectiveness, is directly linked to the share of fallow land in the counties—with a total uptake of 8.7% of the total EFA in Lower Saxony [11].

Table 58.1 Example of the Method of Calculating the "Standardized Ecology Score" Value for one of the Counties in Lower Saxony*

EFA Option	(1) EFA Area in a County (in ha)	(2) Biodiversity Scoring by Ecologists	(3) Ecology Scores
Catch crops	3141	0.4	1256
Nitrogen fixing crops	635	0.7	445
Fallow land	1195	2.4	2868
Landscape element	86	1.6	138
Buffer strips	71	2.5	178
(4) Sum of Ecology Scores			**4884**
(5) Arable Land in the County			**49,443**
(6) "Standardized Ecology Score"			**9.88**

Biodiversity score from Pe'er et al. [1]; implementation area data from Dahl [11]

Figure 58.4 shows the standardized ecology score of Agri-Environment and Climate Measures (AECM) on arable land (catch crops, flower and buffer strips, and landscape elements). Overall, the scores are significantly lower, primarily because the total area is much smaller (but noting that we did not include AECM on grassland). The spatial patterns of Fig. 58.4 are somewhat different from Fig. 58.3: We find low standardized ecology scores in the west (Weser-Ems, Ostfriesland in red). These regions are dominated by intensive meat and dairy production, arable land is expensive, and therefore farmers are rarely willing to participate in AECM. In the arable regions with favourable soil conditions (Börde) we also find rather low participation rates in AECM. We find higher participation rates only in the arable regions with

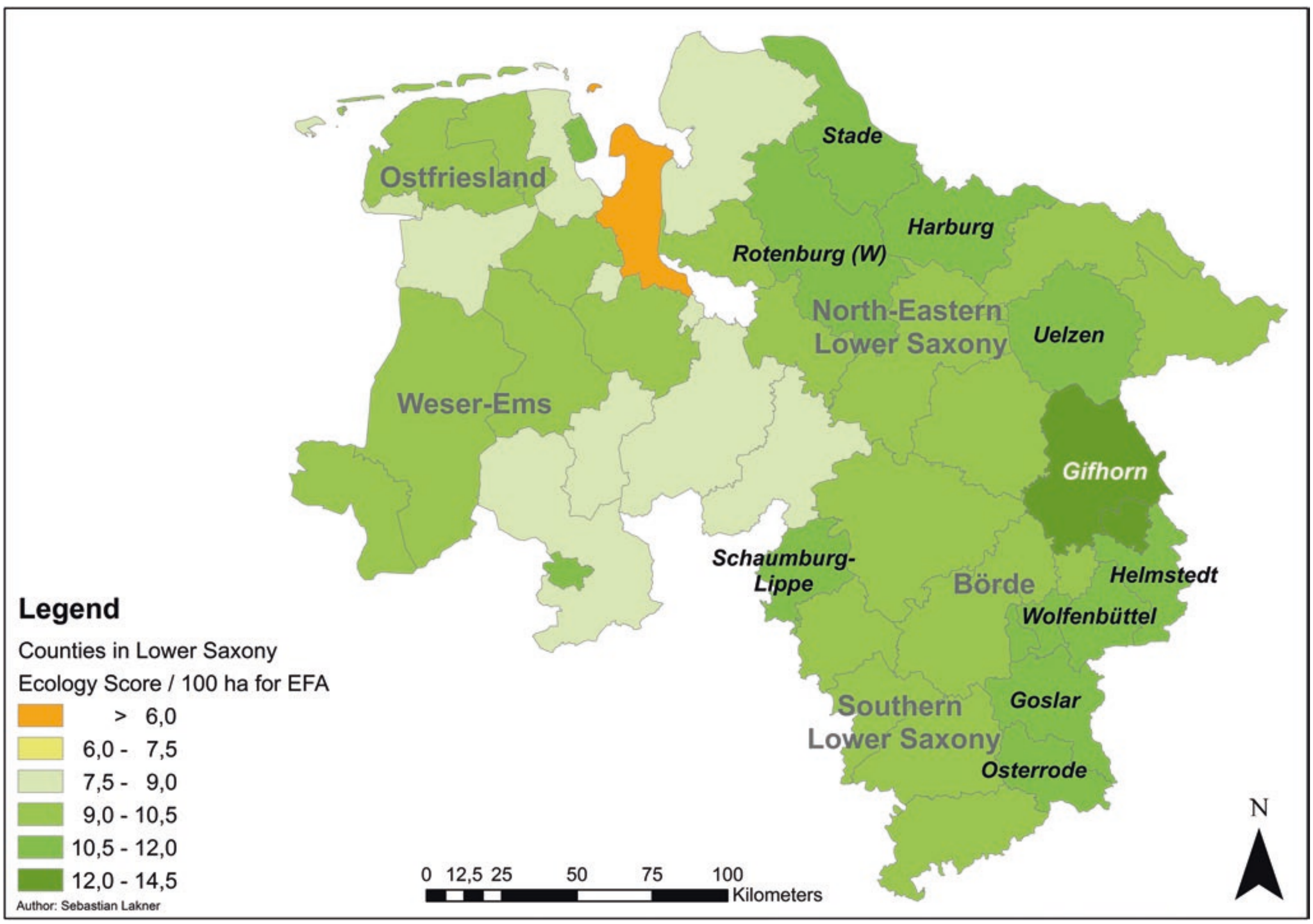

Fig. 58.3 Ecology Score per 100 ha of Ecological Focus Area (EFA) on arable land in Lower Saxony. Source: own calculation based on expert-scoring versus implementation data for all EFA-options eligible for arable land in Lower Saxony, 2016. Colours reflect anticipated impacts on biodiversity and ecosystem services from high (green) to low (red). Only score values above 5 are included; therefore the lower part of the scale is not presented. Shapefiles for administrative borders from FACG [6]

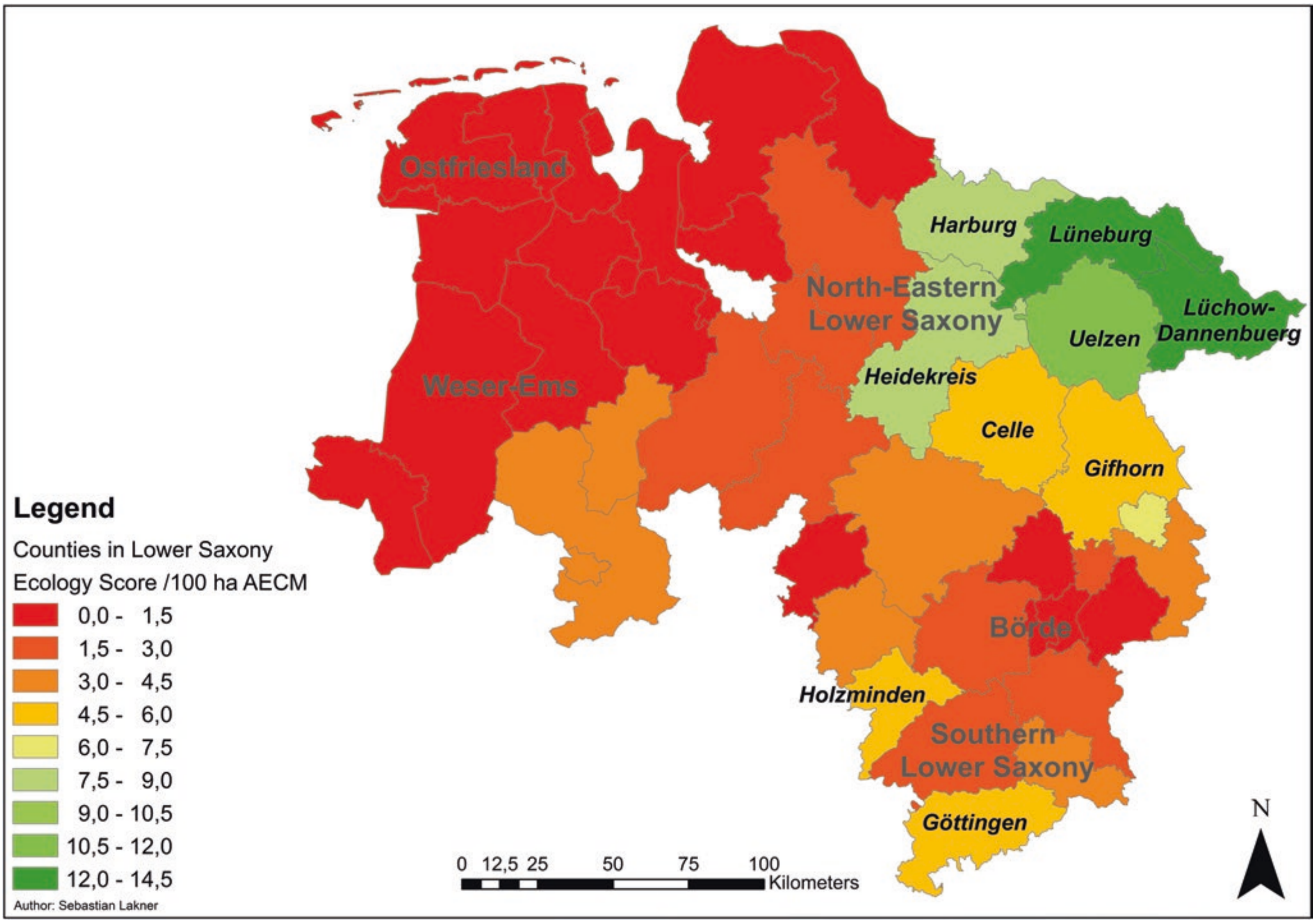

Fig. 58.4 Ecology Score per 100 ha for Agri-Environment and Climate Measures (AECM) on arable land in Lower Saxony. Source: own calculation based on expert-scoring versus implementation data for all AECM eligible for arable land in Lower Saxony, 2016. Colours reflect anticipated impacts on biodiversity and ecosystem services from high (green) to low (red). Shapefiles for administrative borders from FACG [6]

less favourable soil conditions (North East Lower Saxony) and in the Counties of Göttingen and Holzminden. Notably, the spatial distribution of the AECM-effect in the east is quite different than of the EFA, with high implementation of AECM in the north-east; and very low participation in the Börde region.

58.4 Economic Considerations

In addition to the critique on the low effectiveness of Ecological Focus Areas (EFAs), it has been suggested that this instrument has also low efficiency [12]. Table 58.2 shows area, score, and costs of EFA and Agri-Environment and Climate Measures (AECM).

The data clearly document that area and payments for EFA are substantially larger compared to the AECM on arable land. With Greening, the EU has strongly increased the efforts. However, the sum of ecology scores shows a low effectiveness: EFA-measures achieve only 0.62 scores/ha on average, whereas AECM achieve 0.79 scores/ha. AECM are the more effective measures given the chosen methodology.

Considering the costs of both measures divided by ecology scores, we see that for EFA, the taxpayer subsidizes each ecology score with 612 EUR, whereas for AECM only 307 EUR per ecology score are paid. This suggests that from a taxpayer's perspective, AECM measures are less expensive for the provision of biodiversity and ecosystem services. On the other hand, with increased area of AECM, the costs will also rise.

The AECM are voluntary. Farmers might therefore choose arable land for AECM, which has a lower yield potential and induces fewer costs for the farm. The ecology scores for EFA are mainly achieved on an average soil quality. In contrast to the AECM, which are rather on non-favourable locations, the EFA are produced with slightly higher costs. This might explain parts of the difference in the payments per ecology scores between EFA and AECM (+99.4%).

Table 58.2 Area, Costs and Impacts of EFA and AECM in Lower Saxony 2016[a]

	Ecological Focus Area (Pillar I)		Agri-Environment and Climate Measures (Pillar II)	
Area for biodiversity and ecosystem services	304,014.0	ha	84,879.0	ha
Total payment for measures on arable land[b]	114.7	Mio. EUR	20.6	Mio. EUR
Sum of ecology scores	187,259.0	scores	67,167.0	scores
Payment per ecology score	612.6	EUR/score	307.1	EUR/score
Ecology score per hectare	0.62	scores/ha	0.79	scores/ha

[a]The data for AECM were provided by the Ministry for Food, Agriculture and Consumer Protection of Lower Saxony. The data for EFA are from Dahl [11]

[b]For a conservative calculus, we assume that only 50% of the spending for Greening in Lower Saxony is used for EFA. The other 50% is used for the conservation of grassland and for crop diversification

58.5 Discussion and Conclusions

The analysis of the distribution of so-called effective Ecological Focus Areas (EFAs) and comparable Agri-Environment and Climate Measures (AECM) on arable lands indicates low effectiveness of both instruments in the western counties of Lower Saxony (Figs. 58.3 and 58.4). Yet there is some complementarity in the eastern counties, where areas with low effectiveness of EFA are somewhat complemented by higher effectiveness of AECM and vice versa.

Our results demonstrate the classical weakness of AECM, where farmers in productive regions tend not to participate in such programmes because opportunity costs on arable land are substantially higher than the actual payments for AECM. This explains deficits in addressing biodiversity declines in productive arable regions (Fig. 58.2). EFAs perform better in regions with high soil fertility because of their compulsory nature. However, as the Biotope Index in Fig. 58.2 indicates, in Western Lower Saxony there is still much room for improvement, as EFA do not sufficiently support biodiversity and ecosystem services (Fig. 58.3).

Farmers in Lower Saxony have chosen a high share (89%) of catch crops and nitrogen fixing crops [11], indicating that farmers first and foremost choose to implement productive options that are less effective for biodiversity and ecosystem services (see Zinngrebe et al. [13]). Thus, it is important to incentivise the uptake of EFA options that are more beneficial for biodiversity [1, 13].

When implementing EFA, the area the farmers must designate depends on the "weighting factor" (WF), which determines the effective area according to the EU, and accordingly, the minimum area one must register in order to comply with the required area. For instance, with a weighting factor of 0.3 for catch crops, farmers need to register 16.67 ha to fulfil a requirement of 5.0 ha EFA; whereas they have to fulfil only 5.0 ha of fallow land (WF: 1.0) or 3.33 ha of buffer strips (WF: 1.5). In consequence, the largest part of Ecological Focus Area (75%) has a comparatively low ecological score per effective hectare, but is applied on large areas due to a low weighting factor.

Fallow land and buffer strips score high for biodiversity, but cover less than 10% of the EFA in Lower Saxony. This is critical with respect to biodiversity [1]. In terms of ecosystem services, it is argued that catch crops can prevent soil erosion and nitrate leakage [14]; while nitrogen-fixing crops improve soil quality and biodiversity when introduced into the crop rotation. Thus, in terms of ecosystem services, EFA might not score as low as for biodiversity.

Combined with our results, showing deficiencies of AECM on arable lands, we join earlier recommendations to increase the budgets for AECM in Pillar II of the Common Agricultural Policy and/or to improve, focus and simplify EFA as suggested by Zinngrebe et al. [13] and Pe'er et al. [1].

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Social Mapping of Perceived Ecosystem Service Risks: Some Thoughts from a Belgian Case Study

59

Rik De Vreese

59.1 Introduction

There are few examples of social (or stakeholder-based) mapping of ecosystem services risks. For ecosystem services in general, mapping risks to ecosystem services supply is mainly based on geodatabases on land use, land cover, biophysical, or abiotic features. Socially perceived risks to ecosystem services are mainly studied in (preference) surveys, with the data not being geolocalised. However, we cannot presume that socially perceived risks to ecosystem services delivery are overlapping or similar to risks based on more "objective" data such as biophysical features, production functions, or ecological data: aspects such as traditions, personal use, sense of place and identity, but also personal experiences and personal values, co-shape perceptions on ecosystem services risks. For example, a flood buffering area can be modelled as being under high ecosystem services risk because of changing water regimes, but this fact can go unnoticed by stakeholders, or be judged as not at all important to them. In a contrasting example, a new management plan for increasing the ecosystem services delivery in an intensively visited urban woodland can be perceived as a high risk to ecosystem services and appreciated by stakeholders (e.g., socio-cultural ecosystem services). As a consequence, planners and managers lack insight about how users, stakeholders, and citizens perceive ecosystem services risks at specific places ("ecosystem services myopia" [1]). This is a rather uncomfortable situation, given the anthropocentric origin of the ecosystem services risk notion.

Social mapping facilitates integrating stakeholders' perceived ecosystem services risk into planning and management processes. We define social mapping as mapping subjective perceptions and the personal use of nature and landscape and intangible ecosystem services [2]. Social mapping is not only relevant for integrating perceptions. It can also be an appropriate tool for integrating lay knowledge and/or information about less tangible or less mapped ecosystem services into planning and management processes. To synthesize stakeholders' input on socially perceived ecosystem services risks, and to facilitate the integration of socially grounded perceived data with biophysical (and economic) information on ecosystem services and associated risks, we apply a social landscape "risk" indicator.

This chapter focuses on the social side of the ecosystem services framework, and mainly which risks, threats, and

Which ecosystem services are addressed? All ecosystem services that stakeholders perceive as being important. In practice, these are mainly cultural and provisioning ecosystem services.

What is the research question addressed? Which risks, threats, and conflicts related to ecosystem services delivery are perceived by stakeholders?

Can the social landscape indicator "risk" be applied in participatory landscape planning and management?

Which method has been applied? Social mapping: A stakeholder-based method in which stakeholders themselves sketch the locations of perceived ecosystem services-related risks, threats, and conflicts. GIS is used to process the data and to compute the social landscape indicator "risk."

What is the main result? The social landscape indicator "risk" has potential to capture stakeholders' perceived risks to ecosystem services. Perceived ecosystem services risks can be different from and can be located at other sites than ecosystem services risks located by experts (including ecosystem services risks based on ecological mapping or biophysical features).

What is concluded, recommended? We recommend that planners and managers apply social mapping and the associated social landscape indicator "risk" to gain insight about and appreciation of the ecosystem services and the associated perceived risks to ecosystem services. The social maps can be used to identify where there could be resistance from stakeholders to landscape restoration and nature development plans.

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conflicts related to ecosystem services delivery are perceived by stakeholders. Having an insight into the perceived ecosystem risks is a prerequisite to choosing the most appropriate type of societal response to ecosystem services risks or ecosystem risks. In this chapter we will showcase the social landscape risk indicator with results of a case study in central Belgium. We will discuss how the risk indicator can be applied in landscape planning and management.

59.2 Methodology

This chapter is based on a case study in a peri-urban study area close to Brussels (central Belgium, see Figs. 59.1 and 59.2; see De Vreese et al. [2, 3] for details on the study area and the stakeholders sample). The study area includes two municipalities in the Flemish region (Oud-Heverlee and Bierbeek) and two municipalities in the Walloon region (Grez-Doiceau and Beauvechain). We interviewed 38 stakeholders in land use and landscape management on perceived ecosystem services delivery in the study area. Respondents included active members of environmental NGOs (eNGOs, managers of nature reserves or board members, $N = 7$), farmers ($N = 4$), executive politicians (mayors and aldermen competent for nature, environment, agriculture, and/or spatial planning, $N = 7$), civil servants (similar competences, at municipal and regional level, $N = 8$), and 12 citizens from different backgrounds (e.g., socio-cultural work, culture, arts, sports, recreation). The respondents (1) scored the importance of local nature and landscape for supplying 32 ecosystem services (see De Vreese et al. [2] for an overview of the ecosystem services discussed), and (2) located the most important sites for ecosystem services delivery in their living and working environment. The sketched locations were digitised in GIS (polygons) and related ecosystem services were registered in a geodatabase (conflicts and synergies were based on discourse analysis of the transcripts; see

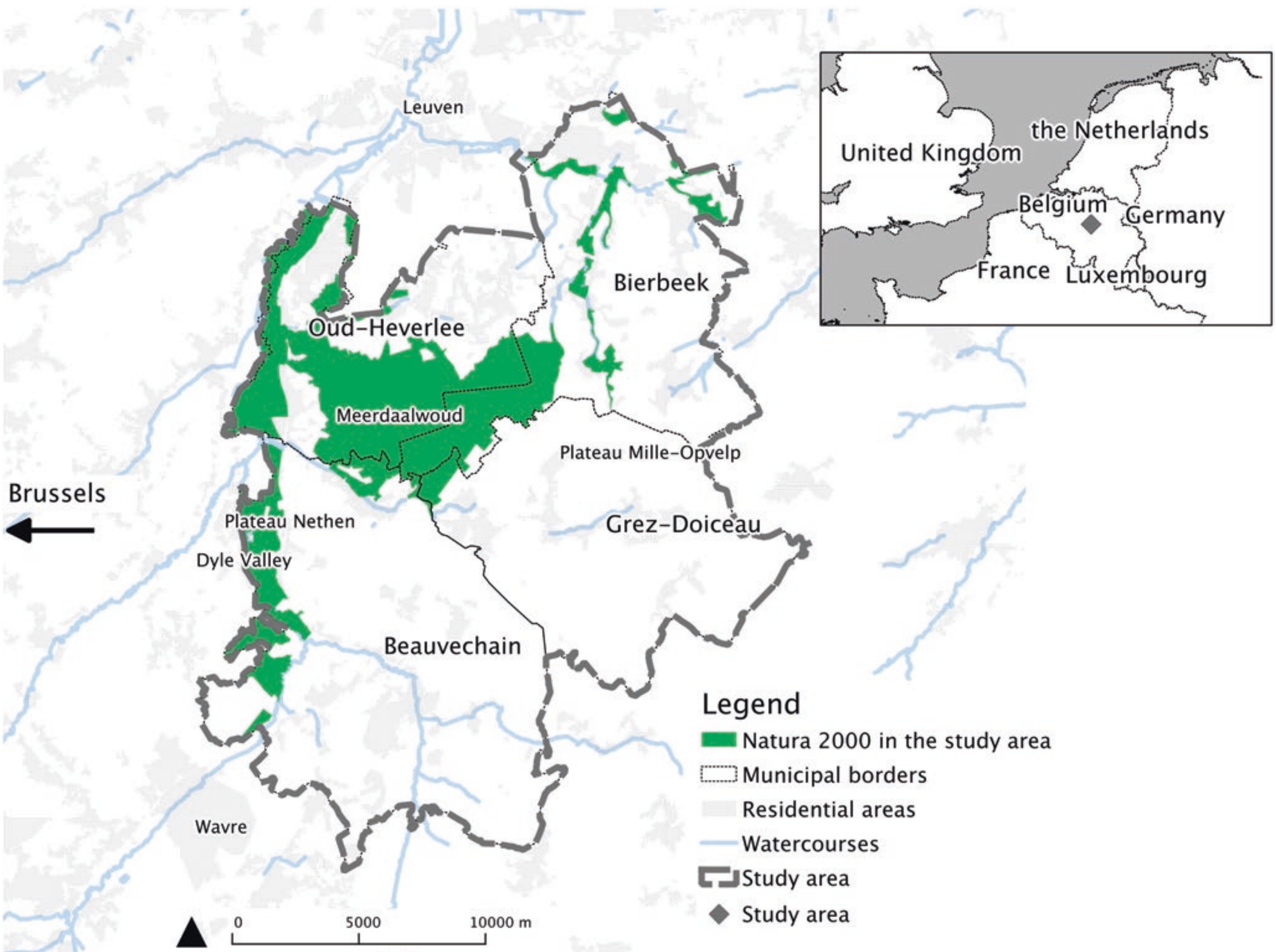

Fig. 59.1 Study area and situation of the Natura 2000 areas in the study area (*reprinted from* De Vreese et al. [2]; *with permission from Elsevier*). The Natura 2000 sites include forest areas (Meerdaalwoud), valley systems (Dyle valley from north to south along the west border of the study area), and a brook system in Bierbeek. Note that Natura 2000 is not present in the south-east part of the study area

Fig. 59.2 Pictures from the study area. (**a**) The Meerdaalwoud-Heverleebos (north-west part of the study area) forest is popular with mountainbikers, horseback riders and hikers. (**b**) The Dyle river and its associated brooks and ponds are Natura 2000 and home to specific avifauna and flora. (**c**) The loamy slopes are prone to erosion, resulting in mud slides and floods. (**d**) Meadows in agricultural use or managed as nature reserve are typical for the Dyle valley. (**e**) The area is criss-crossed with roads and residential areas. (**f**) Interview setting with stakeholders from Beauvechain. (**a**–**e**: *Images courtesy of* BOS+; **f**, *Image courtesy of* Ann Van Herzele)

Table 59.1 for the attributes registered). The polygons were intersected, and the intersected polygons were rasterised (with the frequencies of individual ecosystem services as attribute value). Figure 59.3 summarises the methodology.

In the next step, we calculated several social landscape indicators, including risk to ecosystem services delivery. The social landscape indicator "risk" describes the spatial coincidence of stakeholder-defined ecosystem services-related conflict areas with stakeholder-defined ecosystem services abundance (*normalised ecosystem services abundance * normalised conflict abundance*) [4]. We choose this indicator because it accounts as well for the frequency of important ecosystem services (as reported by stakeholders), as well the number of reported conflicts. This means that a site with a high number of reported conflicts, but with a low social importance for ecosystem services delivery (low ecosystem services abundance), will show a lower social ecosystem services risk index than if the site were considered as important for ecosystem services delivery (in their opinion) by more respondents. A site indicated by a limited number of respondents, but where all these respondents report the site as an area of conflict, will have a moderate risk index. The highest risk index is seen at sites with a high social importance (high ecosystem services abundance) and a high number of reported conflicts between different ecosystem services or between ecosystem services and urbanisation.

To compare social mapping with expert-based ecological mapping and biophysical mapping, we applied Jaccard coefficients J [5] to calculate the measure of similarity between the social hotspot (risk >0) and an ecological hotspot (site designated as Natura 2000); with $J = (area(A \cap B))/(area\ (A \cup B))$. Natura 2000 sites (see Fig. 59.1) were selected as proxy for expert-based ecological hotspots because Natura 2000 is designed to "halt the loss of biodiversity and the degradation of ecosystem services in the EU" [6]. Natura 2000 sites in the study area include a diversity of habitats (forest, agricultural land, valley), that deliver a wide range of ecosystem services [6]. Moreover, Natura 2000 is the only dataset that is available in both administrative areas involved in the study area.

Table 59.1 Attributes to the digitised polygons[a]

Name attribute	Description
ID	Polygon number
Interviewee	Name of the interviewee sketching the polygon
Primary ES[b]	ES for which the interviewee sketched the polygon
Secondary ES	ES in conflict (trade-off) or synergy with the primary ES (as mentioned by the interviewee)
Conflict/ Synergy	Is the relation between the primary and secondary ES, as mentioned by the interviewee, a conflicting or a synergetic relation?

[a] *Reprinted from* De Vreese et al. [2]; *with permission of Elsevier*
[b] *ES* ecosystem services

59.3 Social Landscape Metrics Describing Conflicting Ecosystem Services

Thirty-eight respondents indicated a total of 535 polygons with a cumulated surface of 159,486 ha (of which many overlapping polygons; total study area is 16,400 ha), covering 25 of the 32 ecosystem services discussed during the interview. The most important ecosystem services to the respondents were food and fodder production, natural water protection, protection against erosion, protection against flood, provision of habitats, conservation of local species, pollination, non-motorised recreation, aesthetic experiences, relaxation and therapeutic impact, learning and education, and provision of a good place to live. De Vreese et al. [2] offers a detailed overview of the results, including the description of the synergies stated by the respondents. In this chapter we focus on the conflictual relationships.

A content analysis of the respondent interviews (see De Vreese et al. [3]) found that ecosystem services risks in the area are related to (a) tensions between agriculture, nature conservation, and environmental protection (on the use of specific methods, on the need for farming land); (b) urbanisation; (c) recreation (motorised and non-motorised); and (d) discussions on how nature should be managed ("careless nature management approaches," e.g., not removing dead wood, not mowing the paths, not hunting game or managing invasive species). The threats were only partially recognised and localised by the respondents. They located 71 conflicts between ecosystem services or between ecosystem services and urbanisation (cumulated area 12,722 ha, 1–8 conflicts stated per respondent, see Table 59.2). Twenty-one respondents located negative impact of urbanisation on ecosystem services. Other conflicts located by those surveyed referred

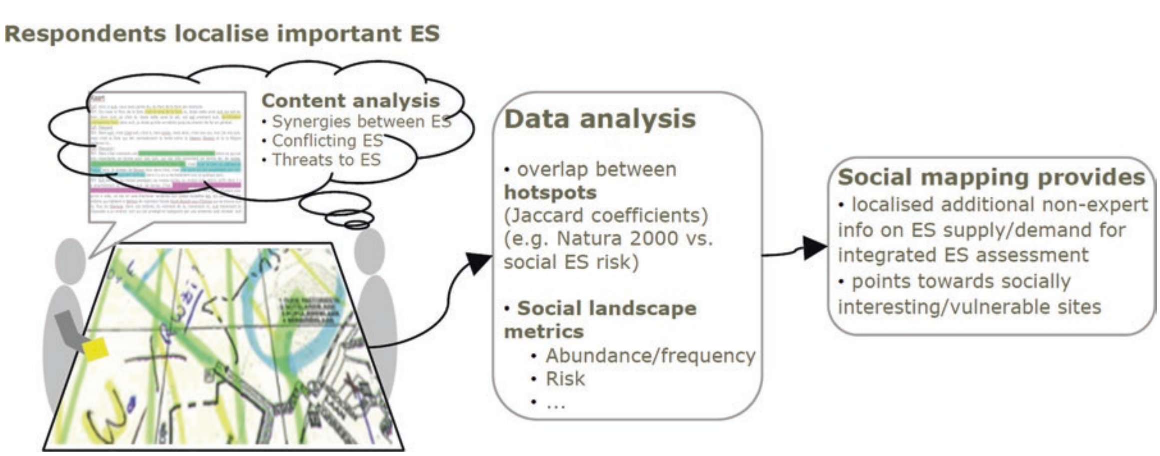

Fig. 59.3 Methodological flow of the research on ecosystem services (ES). (*Reprinted from* De Vreese et al. [2]; *with permission from Elsevier*)

Table 59.2 Conflicting ecosystem services as indicated by respondents[a]

Secondary (conflicting) ES	Air purification		Employment agriculture		Erosion		Food/ fodder		Habitat		Local species		Recreation (non-motorized)		Regional products		Urbanisation		Wood production	
Primary ES	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)	#	area (ha)
Aesthetics													1	58.8			16	2768.8		
Berry picking	1	89.5																		
Conflict only											1	50.0								
Flood protection									1	0.8							8	110.4	1	63.2
Food/fodder																	9	244.4		
Habitat			2	1526.7			1	8.8					2	1841.8	1	1.8	6	1104.7		
Historical landscape									1	7.6										
Noise protection																	2	853.5		
Recreation (non-motorized)					1	102.0	1	85.7	3	315.0							6	3294.1		
Regional products									3	8.4										
Research													1	133.4						
Sense of place																	1	9.0		
Water purification									2	43.5										
Total	**1**	**89.5**	**2**	**1526.7**	**1**	**102.0**	**2**	**94.5**	**10**	**375.2**	**1**	**50.0**	**4**	**2033.9**	**1**	**1.8**	**48**	**8384.8**	**1**	**63.2**
Grand Total	**71**	**12,721.6**																		

Note: areas are cumulated and include overlapping polygons indicated by single or multiple respondents. # is the number of polygons
Reprinted from De Vreese et al. [2]; *with permission of Elsevier*
[a]This table also includes the negative impact of urbanisation on ecosystem services (ES) delivery, as these are the main ecosystem services risks mentioned by the respondents

to the negative impact of recreation on aesthetics, erosion, food production, research, and habitat provisioning; they also noted the negative impact of nature conservation and habitat provisioning to (employment in) agriculture.

The risk indicator (Fig. 59.4) illustrates which conflicted areas are socially more important than others by taking the abundance of the sites into account. Interesting to note is that the overlap between the social risk hotspot (risk index >1) and Natura 2000 is rather limited: the Jaccard coefficient is 38%, which means that the common area of conflict and Natura 2000 is about one-third of the total area of Natura 2000 and sketched conflict areas. Or, as noticeable in Fig. 59.4, ecosystem services risks are not bound to ecological hotspots (in Natura 2000), but also include other areas such as agricultural areas or even residential areas (e.g., the picturesque villages of Biez and Nethen, where ecosystem services such as aesthetical experiences are under threat of urbanisation).

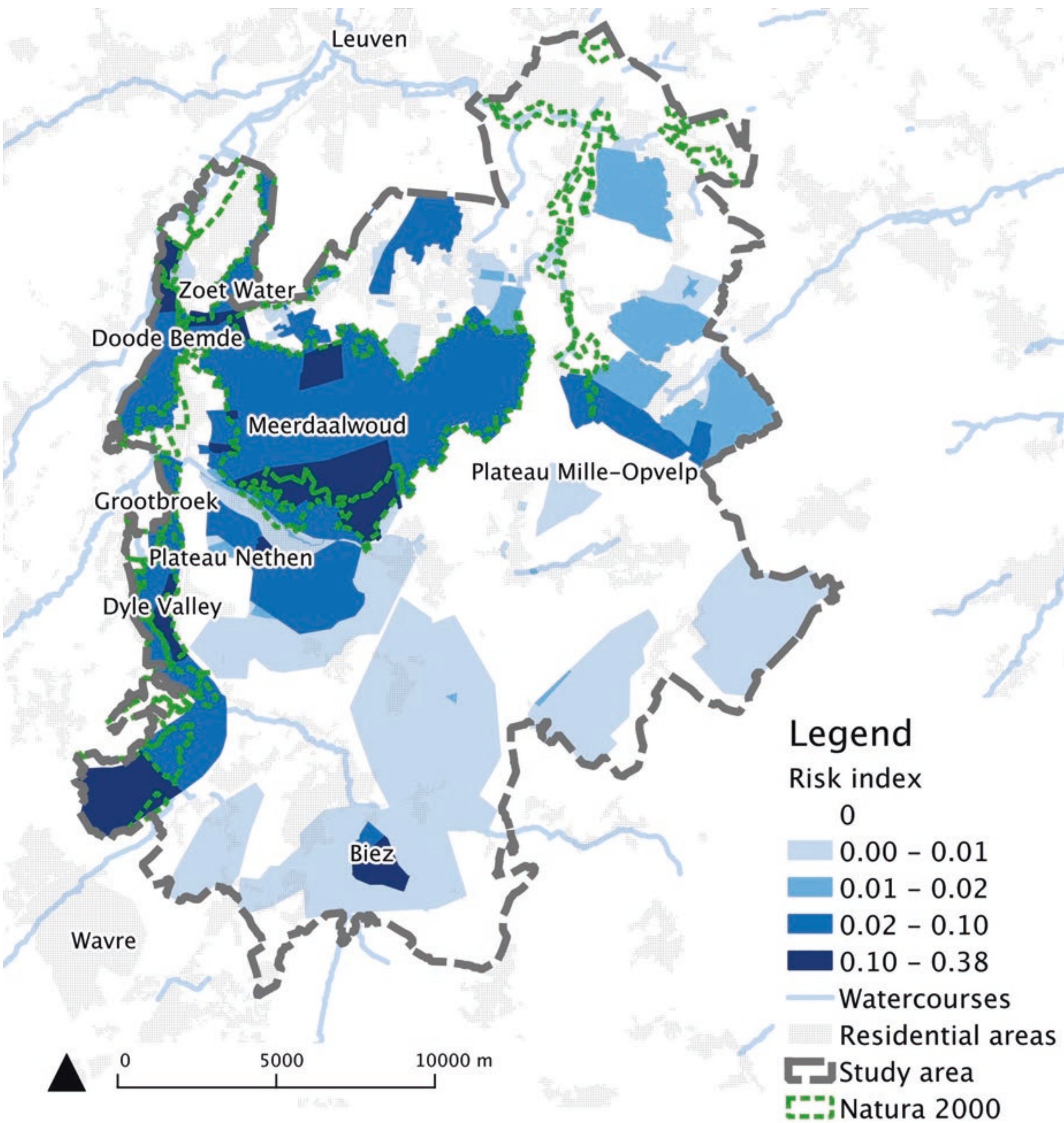

Fig. 59.4 Risk index (based on perceived threats to ecosystem services) and the overlap with Natura2000 areas (*reprinted from* De Vreese et al. [2]; *with permission from Elsevier*). The higher the risk index, the more important the risk is to the stakeholders (more stakeholders mention the risk and/or more stakeholders locate ecosystem services at the spot)

59.4 Discussion

We noticed that socially perceived risks are diverging from the risks indicated by ecosystem service mapping (see the rather limited overlap between the social risk indicator and the Natura 2000 areas in the study area). The risk indicator points to areas where respondents perceive conflicts and trade-offs between socially important ecosystem services and where negative impacts of future development on ecosystems and their services are expected by the respondents. The reasons for the diverging perceptions can be multiple, including differences in perception as well as different notions of nature and ecosystem services between and experts and non-expert stakeholders. Non-experts are less aware of regulating ecosystem services, and stakeholders give larger importance to cultural ecosystem services than do experts. Most cultural ecosystem services are difficult to measure, monitor, and model, and as a result less recognized by common expert mapping methods. It is important to acknowledge and respect the different notions by lay people and experts.

Social mapping can facilitate a social and participative turn in ecosystem services risk mapping: It broadens the knowledge base and is a method to integrate people's perceptions in planning and management of nature and landscape [7]. The results of social mapping help to identify sites with high perceived societal value for ecosystem services supply (social hotspots). Planners, managers, and decision-makers should avoid measures with (potential) negative impact in these sites, or they should at least discuss the planned operations with stakeholders.

The main drawbacks of the social mapping method are (1) the potential collectivisation of individual perceptions, priorities, and conflicts, due to the inherently limited number of respondents involved; (2) the time and effort needed for the approach; (3) the lower interest for and more limited knowledge of stakeholders on less visible and less known ecosystem services (such as regulating ecosystem services); and (4) the undocumented uncertainties related to social data and spatially combining social with biophysical mapping and geographical data [8].

59.5 Conclusion

We illustrated the potential of a social landscape indicator "risk" to capture stakeholders' perceived risks to ecosystem services. Perceived ecosystem services risks can be different from and localised at different sites than the ecosystem services risks localised by experts (including ecosystem services risks based on ecological mapping or biophysical features). We suggest that planners and managers apply social mapping and the associated social landscape indicator "risk" to get insight into the ecosystem services appreciated by people and the associated perceived risks to ecosystem services. The social maps can be used to identify where there could be resistance from stakeholders against landscape restoration and nature development plans.

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Part V

Synthesis and Conclusions

60 Ecosystem Services: Understanding Drivers, Opportunities, and Risks to Move Towards Sustainable Land Management and Governance

Matthias Schröter, Aletta Bonn, Stefan Klotz, Ralf Seppelt, and Cornelia Baessler

The *Atlas of Ecosystem Services* has collected knowledge on drivers, trade-offs, and synergies of ecosystem services and biodiversity, as well as societal responses. It presents case studies from various fields to demonstrate concepts of sustainable land management and governance. In this final chapter, we identify important open questions to sketch avenues for future research in the field (see also Grunewald and Bastian [1]).

60.1 Which Variables and Data Do We Need to Better Quantify, Assess, and Monitor Ecological and Societal Aspects of Ecosystem Services?

Several assessment approaches have been developed, starting with biological monitoring to measure the effects of stressors on biological systems [2], followed by the closely related Essential Biodiversity Variables [3] and indicators of ecological integrity [4]. Promising steps have been taken [5], but the task of linking biodiversity to ecological systems functioning is still a challenge. It is apparent, however, that the role of biodiversity underpinning multiple ecosystem functions and services is not fully understood. Many contributions to this volume hence employ proxies, such as land cover, to assess ecosystem services. Progress has been made to use remote sensing to assess different entities of ecosystem services [6]. The actual realisation of ecosystem services, however, often depends on the demand of different beneficiaries. Here, general measures identifying societal interest and demand as well as impacts on human well-being need to be further developed.

60.2 What Are the Main Driving Forces for Ecosystem Service Change?

This Atlas provides an overview of drivers and pressures on ecosystem services and demonstrates these with different case studies. The Driving-forces-Pressures-States-Impacts-Responses (DPSIR) framework developed by the European Environmental Agency, based on former UN and OECD approaches [7, 8], has been employed in many case studies. Such frameworks must be further developed and implemented in studies on ecosystem service risks [9, 10]. Drivers of ecosystem risk (first order) and ecosystem service risks (second order) can be manifold. Among the diverse drivers covered in this Atlas are the loss of genetic diversity, disturbance of ecological processes, invasions affecting the provision of services to society, pollution, land use, and climate change. The relationship between dynamic anthropogenic pressures and ecosystem functions needs to be better understood, and a process understanding needs to be integrated into ecosystem service valuation [11]. Furthermore, global commodity trade may affect and potentially displace pressures to ecosystems elsewhere. Rising societal demands triggered by, e.g., consumption patterns, demographic challenges and political agendas, may lead to inter-regionally coupled drivers for ecosystem service provision. These drivers may be exacerbated in the coming decades by climate change and associated socio-economic pressures. It is therefore important to not only assess current provision of ecosystem services, but also future changes.

60.3 What Are the Main Spatial and Temporal Patterns of Ecosystem Services?

Spatial scales and hierarchies must be differentiated in the analysis of ecosystem services. The chapters in this volume present a series of studies at different spatial scales and discuss the importance of spatial patterns including the amount and size of different ecosystems and their configuration within a landscape context for the provision of ecosystem services. Open questions relate to the co-appearance of ecosystem services in bundles across landscapes or administrative units. Ecosystem services depend strongly on a given time span with unique patterns of pressures and societal needs. Historically, ecosystems have been formed by a char-

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acteristic set of specific societal needs, cultural preferences, and technological abilities. Looking at the past can facilitate understanding of present situations, patterns of change, and future potentials. Scenario developments are needed to evaluate both the capacity of ecosystems to provide services and their actual use [12].

60.4 Which Trade-Offs and Synergies Occur Between Different Ecosystem Services?

Different types of relationships between ecosystem services have been studied: trade-offs (negative relationships) and synergies (positive relationships) among ecosystem services and with biodiversity and other societal goals [13]. Bundles, i.e., sets of services spatially co-appearing, may result from these relationships, or may develop owing to simple coincidence. There are knowledge gaps on how bundles of services can change over time, and how they differ across large regions. Contributions of the Atlas pointed out the crucial relevance of spatial analyses for analysing relationships between ecosystem services. These can help to identify hotspots, in which conflicts arise that need specific management solutions. To foster advances in this field, research needs to be based also on better regionalized data and on development of metrics and indicators that help to understand the underlying causes of ecosystem service relationships. Such indicators could be used to track changes in ecosystem service relationships over time. An important question in this context is how society can overcome the problem of singular and often competing interests of different land uses for different services, and those impacting on future opportunities. Hence, land use conflicts are a core subject of current and future research. To increase societal relevance of ecosystem service science, studies need to assess socio-ecological systems in an integrative fashion, bridge across scientific disciplines, and include different interest groups and decisionmakers in co-creating research questions.

60.5 What Is the Importance of Different Societal and Political Contexts?

Contributions to the Atlas have pointed to different societal response strategies, including the mitigation of drivers of ecosystem service change, adaptation to a changed ecosystem service provision, and consideration for proactive transformation of ecosystems through management approaches. Different policies and policy mixes need to be considered. When creating and implementing policy instruments, local contexts as well as different stakeholders need to be considered through, e.g., engagement in participatory research approaches. Overall, incentives need to be developed to foster more sustainable land use options. Within this endeavour it is necessary to take a comprehensive approach, i.e., addressing several drivers, to foster policy and management cross-coherence and avoid shifting pressures. There is a strong need to better understand ecological complexity to be able to create suitable policy instruments. Concerning the dynamics of ecosystem service provision, the equitable distribution of benefits derived from ecosystem services needs to be analysed in terms of distributive and procedural equity. There are differences between stakeholders with regard to needs and preferences of ecosystem services, and there are differences in power relationships, which has the potential to lead to inequitable distributions of these benefits. When making decisions, multiple values in society should be considered and a comprehensive understanding of human well-being is needed—one that embraces considerations on, e.g., shared social values and health.

60.6 How Can We Integrate Concepts, Methods, and Models from Different Disciplines for Future Studies of Ecosystem Services?

The study of ecosystem services needs contributions from different scientific fields [14] and the involvement of civil society. Many contributions to this Atlas point directly or indirectly to the need for interdisciplinary studies across natural and social sciences. Some of the studies in this volume analyse drivers of change but do not yet comprehensively address the societal response side, while others analyse societal responses but do not yet fully address the drivers of or relationships between ecosystem services. The challenge ahead for the field is to develop avenues for integrative studies to cover several elements of our framework (Schröter et al., Chap. 1, this volume). Overarching general approaches that could guide future research might be ecosystem integrity or resilience [15].

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