João C. Azevedo · Ajith H. Perera
M. Alice Pinto *Editors*

Forest Landscapes and Global Change

Challenges for Research and Management

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Preface

Climate change, urban sprawl, abandonment of agriculture, intensification of forestry and agriculture, changes in energy generation and use, expansion of infrastructure networks, habitat destruction and degradation, and other drivers and pressures of change are occurring at increasing rates globally. They affect ecological patterns and processes in forest landscapes and modify ecosystem services derived from those ecosystems. Consequently, the landscapes that are rapidly changing in response to these pressures present many new challenges to scientists and managers. Although it is not uncommon to encounter the terms “global change” and “landscape” together in the ecological literature, there has been no adequate global analysis of drivers of change in forest landscapes and their ecological consequences. Providing such an analysis is the goal of this volume: an exploration of the state of knowledge of global changes in forested landscapes, with an emphasis on their causes and effects, and the challenges faced by researchers and land managers who must cope with these changes.

This book was based on the IUFRO Landscape Ecology Working Group International Conference that took place in Bragança, Portugal, in September 2010 under the theme “Forest Landscapes and Global Change: New Frontiers in Management, Conservation and Restoration”. The event brought together more than 300 landscape ecologists from almost 50 countries and 5 continents, who came to expand their knowledge and awareness of global changes in forest landscapes. We hope that the syntheses in this book, prepared by a diverse group of scientists who participated in the conference, will enhance the global understanding of a range of topics relevant to change in forest landscapes and stimulate new research to answer the questions raised by these authors.

First, we introduce the broad topic of forest landscape ecology and global change. This is followed by chapters that identify and describe major agents of landscape change: climate (Iverson et al.), wildfire (Rego and Silva), and human activities (Farinaci et al.). The next chapters address implications of change for ecosystem services (Marta-Pedroso et al.), carbon fluxes (Chen et al.), and biodiversity conservation (Saura et al.). A subsequent chapter describes methodologies for detecting and monitoring landscape changes (Gómez-Sanz et al.) and is followed by a chapter that highlights the many challenges facing forest landscape managers amidst global change (Coulson et al.). Finally, we present a summary and a synthesis of the main points presented in the book (Azevedo et al.). Each chapter was inspired by the research experience of the authors, augmented by a review and synthesis of the global scientific literature on relevant topics, as well as critical input from multiple peer reviewers.

The intended audience for this book includes graduate students, educators, and researchers in landscape ecology, conservation biology, and forestry, as well as land-use planners and managers. We trust that the wide range of topics, addressed from a global perspective by a geographically diverse group of contributing authors from Europe, North America, and South America, will make this volume attractive to a broad readership.

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Chapter 1
Forest landscape ecology and global change: an introduction

João C. Azevedo, Maria Alice Pinto, and Ajith H. Perera

Abstract Forest landscape ecology examines broad-scale patterns and processes and their interactions in forested systems and informs the management of these ecosystems. Beyond being among the richest and the most complex terrestrial systems, forest landscapes serve society by providing an array of products and services and, if managed properly, can do so sustainably. In this chapter, we provide an overview of the field of forest landscape ecology, including major historical and present topics of research, approaches, scales, and applications, particularly those concerning edges, fragmentation, connectivity, disturbance, and biodiversity. In addition, we discuss causes of change in forest landscapes, particularly land-use and management changes, and the expected structural and functional consequences that may result from these drivers. This chapter is intended to set the context and provide an overview for the remainder of the book and poses a broad set of questions related to forest landscape ecology and global change that need answers.

1.1 A brief history of forest landscape ecology

Before we can discuss landscape ecology, it is necessary to define what we mean by a *landscape*. Although this term has been given different interpretations by authors from different backgrounds, in the ecology literature, a landscape is most often considered to mean an area that is heterogeneous in at least one factor of interest

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(Turner et al. 2001). A landscape perceived by land and natural resources managers, for example, is usually a broad-scale mosaic of land-use and cover types that are strongly interconnected and are functioning as a single unit. Watersheds represent a good example of a landscape because, despite the diversity of ecosystems and land uses within any given watershed, all components of the watershed are interconnected, such that changes in one component affect all other components either directly or indirectly for a change is transmitted through intermediate components.

Landscape ecology emerged in central Europe in the 1930s following the development of ecology as a separate branch of science. At the beginning of the twentieth century, forests in Europe often consisted of fragments or remnant woodland patches in landscapes dominated by other land uses, typically agriculture and urban, but were nonetheless important in the functioning of ecosystems and the landscape, particularly in terms of water, soil, and wildlife conservation. Through recognition of these functions, forests became key units in land-use planning, which was one of the major applications of landscape ecology in Europe during the twentieth century (Naveh and Lieberman 1994). Historically, the study of forests within a landscape context has also been addressed from the perspective of plant ecology in terms of plant communities with inherent temporal and spatial patterns (Turner 1989).

It was only in the 1980s that forests were addressed explicitly as landscape systems—as dynamic mosaics of interacting landscape units, patches, tree cohorts, or stands. Several factors contributed to this evolution. First, the establishment of a conceptual framework for the science of landscape ecology (Zonneveld 1990) provided the theoretical grounds for formally addressing and testing scientific hypotheses about landscapes, including forested landscapes. Second, the increasing availability of technology for data collection, storage, and analysis made it possible not just to process the large amounts of data associated with extensive and heterogeneous land areas, but also to incorporate spatially explicit methodological approaches, including spatial modeling of landscape structures and functions, into research (Mladenoff and Baker 1999b, Turner 1990). Third, many recent developments in landscape ecology occurred in regions where landscapes were predominantly forested, such as North America and Australia, which resulted in a high proportion of landscape ecology studies being conducted in forested landscapes (Perera et al. 2000). Fourth, forests are particularly interesting to ecologists because of their high spatial and vertical heterogeneity and the resulting complexity and high levels of species diversity they contain. They became particularly attractive for ecologists with an interest in the relationships between landscape patterns and biological diversity (Hunter 1990, Lindenmayer and Franklin 2002, With 2002).

Last, and particularly important in the context of anthropogenic landscapes, forests have a high social and economic value, both from a traditional timber-based economics perspective and from the more contemporary perspective of a sustainable, multifunctional ecosystem that provides crucial services for society. Globally, forests are now managed to ensure a sustainable production of commodities that combine conventional forest and non-forest products and ecosystem services. The landscape scale has become a required component of planning to address sustainable forest management, and forestry professionals started incorporating a landscape approach (Schlaepfer and Elliott 2000). As a reaction to this paradigm shift, a landscape perspective was also incorporated into silviculture by explicitly developing silviculture models and practices at this scale (Boyce 1995, Oliver and Larson 1996).

From 9366 publications selected by searching the Web of Science database (http://thomsonreuters.com/web-of-science/) in 2013, with all years included and with "landscape" and "ecology" as search terms, we found that 3290 (35 % of all landscape ecology publications) dealt with forests. This proportion is only approximate, since publications dealing with landscape ecology or forest landscape ecology do not necessarily include these terms in the abstract or keywords, but it shows the relative importance of forests within the field of landscape ecology. In the journal *Landscape Ecology*, 46 % of all the publications since 1987, the journal's first year of publication, included the term "forest" in the text. The number of forest landscape ecology publications has grown considerably over time, from 5 in 1988 to 316 in 2012. Around 15 % of the publications are from a forestry perspective, with the remainder focused on ecology and conservation. In addition to publications in scientific journals, numerous books have been devoted to forest landscape ecology, including collections that resulted from forest landscape ecology conferences (Lafortezza et al. 2008, Li et al. 2011) and works dedicated to the ecology of forest landscapes; to the application of principles and methods of landscape ecology to the practice of forest planning, management, and conservation; and to a broad range of closely related theoretical and applied subjects (e.g., Hong et al. 2007; Kohm and Franklin 1997; Lindenmayer and Fischer 2006; Lindenmayer and Franklin 2002; Perera et al. 2000, 2004, 2006).

It appears that the majority of forest landscape ecology research has occurred in North America, since 58 % of the publications stem from the United States and Canada. However, this field is rapidly expanding to new regions and new forest systems around the world, where it is contributing to a better understanding of landscapes and is supporting sounder forest management. This geographical shift creates challenges for the science of landscape ecology because it addresses the dynamics of highly complex and insufficiently understood systems and their responses to drivers of change. The new frontiers of forest landscape ecology include countries such as Brazil, Spain, and China. In Brazil, for example, the study of forest landscape ecology has grown rapidly in recent years, mainly within the fields of conservation biology (Lantschner et al. 2012, Tabarelli et al. 2004, Zanella et al. 2012), landscape dynamics (Freitas et al. 2010, Laurance et al. 1998, Lira et al. 2012), and forest management (Amaral et al. 2009, Brockerhoff et al. 2013).

1.2 A definition of forest landscape ecology

Despite the importance of forest landscapes in the development of landscape ecology and the emphasis on a landscape scale in research on forest conservation and management, forest landscape ecology has not become an independent field.

For the most part, it is still mostly landscape ecology in a forest context or a subset of landscape ecology that addresses relationships “in spatial geometry among forest elements” (King and Perera 2006) and how patterns and interactions affect forest processes and dynamics in heterogeneous forested areas. No major conceptual distinction is usually made between forest landscape ecology and the ecology of other types of landscapes, and the approaches, scales, and methods used are similar to those used in any other field of landscape ecology (e.g., Chen et al. 2008). Forest landscapes are also often defined in terms of conventional landscape ecology concepts, frameworks, and indicators, with an emphasis on the large extent of the landscape, the dominance of forest land-cover types (despite the potential presence of non-forest elements), and high heterogeneity that produces a mosaic-like structure (King and Perera 2006, Perera et al. 2006, Perera and Euler 2000).

Although forest landscape ecology is part of the broader science of landscape ecology, it has a very well-defined context and distinctive research issues and concerns. Forest landscape ecology has gained its own identity from the nature, type, and scales of the subjects of study and the issues and questions about the ecology of forest mosaics, within which management is a central component. One major element of this identity relates to the fact that landscapes with contiguous forest differ from landscapes where the forest cover exists only as patches in a matrix dominated by other types of land use or cover. Dynamics in contiguous landscapes, although preserving structural stability at the landscape scale, cause changes in ecosystems at the local scale. Patches and edges are, therefore, not spatially fixed structures in contiguous forest landscapes, since they change over time. As a result, fragmentation is often temporary, except when it is associated with a long-term trend of landscape change. This has caused the conceptual basis of forest landscape ecology to be supported by systems ecology, percolation theory, and disturbance or resilience perspectives more than in other fields of landscape ecology.

Most forests are managed, and for that reason, forest landscape ecology has commonly dealt with managed forests and management-related issues, with an emphasis on the causes and effects of management (Perera and Euler 2000). This is possibly one of the most distinctive aspects of forest landscape ecology. It is, therefore, not surprising that forest landscape ecology has gained attention outside of academic ecological circles, such as in the forest industry and in national and regional administration of forests. Forest products companies address the landscape scale in forest management that is performed under sustainable forest management certification programs (Ferraz and Ferraz 2009). Federal and national agencies have incorporated the landscape scale in forest policy and management since the 1990s, following the emergence of novel management concepts such as ecosystem management and adaptive management (Rauscher 1999).

Many of the research issues in forest landscape ecology address either how management affects landscapes or how landscape-level patterns or processes affect forest management. The field can therefore provide a solid background to inform forest and landscape management based on landscape ecology principles (Gustafson and Diaz 2002). Management must also be accounted for in the context of other drivers of change that affect the structure, processes, and responses of the landscape at a variety of scales.

1.3 Major research topics in forest landscape ecology

Wu and Hobbs (2002) proposed the following as the top ten research issues in landscape ecology: ecological flows in landscape mosaics; the causes, processes, and consequences of land-use and land-cover change; nonlinear dynamics and landscape complexity; scaling; development of new methods; relating landscape metrics to ecological processes; integrating a description of humans and their activities into landscape ecology; optimizing landscape patterns; landscape sustainability; and data acquisition and accuracy assessment. These have all been addressed in forests as well as in other landscape types. In all journal publications concerning forest landscape ecology that have been published since 1987, the terms most frequently used are habitat (52 % of publications), pattern (45 %), scale (41 %), management (38 %), change (37 %), conservation (36 %) land use (28 %), fragmentation (26 %), patch (22 %), disturbance (21 %), edges (11 %), heterogeneity (11 %), and connectivity (8 %). Of these, we consider edges, fragmentation, connectivity, disturbance, and biodiversity to be essential topics in forest landscape ecology, and in the rest of this section, we will briefly discuss why they are important and will provide links to the other chapters of the book, where relevant.

1.3.1 Edges

Edges have attracted more attention from forest landscape ecologists than from other ecologists (Donovan et al. 1997, Harper et al. 2005). Edges, created by disturbance and patterns in the distribution of resources, affect the physical environment (Chen et al. 1995), the composition and distribution of communities (Fraver 1994), and many ecological processes (Chalfoun et al. 2002) that function across adjacent patches. Although edge effects are local, they have cumulative effects through their influences on the abundance and spatial pattern of interior forest habitats and associated species (Gustafson and Crow 1996). In the literature, forest edges have been considered mostly from the perspective of biodiversity conservation based on their effects on the availability and quality of forest habitat and the spatial distribution of species (Ries et al. 2004).

Forest edges first became a relevant issue in the context of harvesting and management of pristine or other forests, particularly on the western coast of North America (Chen et al. 1992, Franklin and Forman 1987). This focus spread to other parts of the world (Alignier and Deconchat 2011, Tabarelli et al. 2004, Williams-Linera et al. 1998). The seminal paper of Franklin and Forman (1987) addressed how the size and pattern of harvesting units potentially affected landscape structure and key processes related to edge effects and biodiversity, and this remains an important field of research, as edges remain dominant features in managed landscapes. Knowledge generated since Franklin and Forman's paper was published has supported the development of management guidelines for forest landscapes (e.g., FSC 2010).

1.3.2 Fragmentation and connectivity

Fragmentation has shaped landscapes in many parts of the world for thousands of years due to the effects of land-use conversion and land degradation once humans became major drivers of landscape change. However, it is the ongoing fragmentation in forest landscapes in North and South America, Asia, and Oceania that raises concerns among scientists and conservation and management authorities, given the potential of this complex process to cause a loss of species and degradation or loss of key ecosystem functions (Hill et al. 2011, Laurance et al. 2011, Riitters et al. 2002, Saunders et al. 1991, Skole and Tucker 1993). As a subject of research, fragmentation has created a common ground for the integration of disciplines such as ecology, management, and social sciences within a common framework, in which the search for relationships between social and ecological patterns and processes at multiple scales has become a major goal.

Research on fragmentation is challenging given the multiple interactions among the structural components, such as habitat area, patch size, number, shape, perimeter–area ratio, edge abundance, distance or isolation, and connectivity in fragmented landscapes, as well as between these factors and ecological processes that are affected by the degree of fragmentation. On the other hand, this research provides fundamental support for efforts to halt fragmentation and to ensure that essential ecological functions are maintained in fragmented landscapes. Knowledge of fragmentation, in both structural and functional terms, is abundant and has solid theoretical support (Fahrig 2003, Forman 1995, Forman and Godron 1986, Lindenmayer and Fischer 2006).

Connectivity is a major goal in landscape systems, particularly when they are managed, as connectivity is necessary to provide pathways for movement between habitats for animal and plant species and to contribute to the maintenance of biodiversity at all scales (Lindenmayer and Fischer 2006). Connectivity is, therefore, a general component of sustainable landscapes (Forman 1995) and is now considered to be an essential target in forest management and conservation (Lindenmayer and Fischer 2006, Lindenmayer and Franklin 2002, Loyn et al. 2001). Connectivity has been traditionally considered from a structural perspective, although the concept was originally formalized as a process-oriented factor (With et al. 1997). The analysis of connectivity has evolved towards a more functional approach based on the traits of particular species of interest (Taylor et al. 2006). Attempts to combat the effects of fragmentation often rely on the creation or maintenance of structural connectivity between particular ecosystems in the landscape, usually through corridors, “stepping stones”, or “green infrastructures” (Franklin et al. 1997, Lindenmayer and Franklin 2002, Zanella et al. 2012). In sustainable forestry, riparian management zones and wildlife corridors, for example, are used to provide habitat that permits movement of organisms among habitat patches and across landscapes driven by forest management. These features are fundamental for complying with sustainable forestry and certification programs (e.g., FSC 2010).

Connectivity research has been an important component of landscape ecology and has produced a set of theoretical, methodological, and application tools for evaluating this attribute and testing hypotheses concerning its role in ecological processes (e.g., Saura 2008, Saura et al. 2011, With 2002). Fragmentation and connectivity and their effects on biodiversity are analyzed in detail in Chap. 7 of this book.

1.3.3 Disturbance

Given its significance in landscape dynamics and forest management, disturbance is another important factor in landscape ecology research. Forest management has been defined as "the management of disturbance and succession to achieve specific vegetation and ecological conditions that in turn support the products and benefits sought by the manager" (Gustafson and Diaz 2002). Management focuses on broad-scale processes based on the temporal and spatial dimensions of disturbance, which affect the configuration and functioning of the forest landscape. Much of forest landscape ecology research has dealt with disturbance, whether natural or anthropogenic in origin. Disturbance generated by natural causes (e.g., fire, hurricanes, pests) or by clearcutting and other silvicultural models (Liu et al. 2012, Perera and Buse 2004) is a major source of patterns, processes, and dynamics in forest landscapes. Disturbance regimes or management plans determine the composition and configuration of forest landscapes (Mladenoff et al. 1993, Wallin et al. 1994) and affect the processes that shape the distribution of populations and communities, genetic flows, water yield, soil erosion, and productivity, among other factors, at stand and landscape levels (Burton 1997, Saura et al. 2011). On the other hand, the frequency, intensity, and extent of disturbances are affected by the structure of the landscape (Cumming 2001).

Efforts to integrate natural disturbance patterns into forest planning and management include several approaches; Perera et al. (2004) and North and Keeton (2008) provide an overview of the roots, principles, methods, and applications of emulating natural disturbance. However, this approach is based on the idea that the spatial and temporal attributes of natural disturbance events can provide a template for forest management and can guide the definition of management strategies and practices. For example, in a forest management plan, clearcut size could be defined based on the statistical distribution of the size of burned areas, and rotation length could be combined with size based on a consideration of the fire recurrence interval. This would contribute to maintaining the structure and functioning in a managed forest landscape such that it resembles that of a natural landscape. This is assumed to result in a sustainable forest landscape. Chapter 3 in this book covers the specific case of fire at the landscape level.

1.3.4 Biodiversity

Biodiversity, whether in natural systems or in the context of forest management, has become a major component of forest landscape ecology (Fahrig 2003, Zavala and Zea 2004). Species diversity is a key component of ecological systems and is fundamental for providing most ecosystem services (see Chaps. 5 and 7). Ecosystem diversity is an important element of landscape structure and complexity and is one of the most commonly measured landscape attributes, usually through indices based on information theory. Principles, guidelines, frameworks, strategies, and practices proposed for the conservation of biodiversity in forests often are applied at the landscape scale (Gustafson and Diaz 2002, Hunter 1990, Lindenmayer and Franklin 2002, Lindenmayer et al. 2006, With 2002). In addition, the landscape level is a fundamental requisite for biodiversity conservation in the context of sustainable forest management programs (Montréal Process 1999) as well as in the context of land-use and climate change (Araujo et al. 2011).

1.4 Forest landscape ecology and change

Change is an intrinsic characteristic of landscapes, which is why change is part of the definition of landscape ecology (e.g., Forman and Godron 1986). Since both patterns and processes evolve over time, this dimension must be directly or indirectly considered in research methods and applications. Change has attracted additional interest in recent years due to the rapid transformations that many landscapes are exhibiting and to the consequences that these changes are expected to have on ecosystem services and human well-being (Hassan et al. 2005).

In forest ecology, change has been historically addressed mainly from an aspatial, community or stand, perspective and has been frequently based on the concepts and theories of ecological succession, climax communities, and disturbance. Although disciplines within geography and ecology, such as phytosociology and phytogeography, deal with the distribution and temporal patterns of plant communities at broad spatial scales (Turner et al. 2001), such changes in forest systems were not explicitly addressed until the 1980s (Bormann and Likens 1994, Mladenoff and Baker 1999a, Sprugel 1991, Turner et al. 1993). Since then, several methods and models that account for changes in forest landscapes have undergone rapid development (Mladenoff 2004, Xi et al. 2009), making possible not just the modeling of spatial patterns and processes but also the application of these tools in management-oriented simulations.

1.4.1 Landscape dynamics

All landscapes are dynamic, since both their structure and how they function change over time. However, under many natural conditions, these dynamics are relatively stable over time, with the landscape reaching and maintaining an equilibrium state. For instance, see the shifting mosaic steady-state concept of Bormann and Likens (1994) and the review by Turner et al. (1993). Increasingly often, however, forest landscape change is driven by anthropogenic disturbances such as harvesting (Gustafson and Diaz 2002), by human-mediated disturbances such as fire (Moreira et al. 2011), or even by land-use change through the expansion of agriculture or urban areas (Meyer 1995). These changes often push the landscape dynamics away from a more stable condition (equilibrium).

Whereas landscape dynamics occur mostly at a microscale (1 to 500 years; 1 to 10^6 m^2) in the conceptual temporal and spatial ecological framework of Delcourt et al. (1982), some landscape change events occur at a mesoscale (500 to 10 000 years; 10^6 to 10^{10} m^2). Mesoscale processes relevant for forest landscape change include long-term changes in vegetation cover and are driven by anthropogenic factors and by climate change (e.g., land-cover changes throughout the Holocene). Major proximate drivers of forest landscape change have also acted at this scale, including historical land-use and cover change, habitat loss and degradation, and habitat fragmentation.

At the microscale, forest landscapes are affected both by physical environmental change, particularly through climate cycles (temperature and precipitation) or climate change (see Chap. 2), and by disturbance in the form of major proximate and anthropogenic drivers of forest landscape change (e.g., land-use and cover change, habitat loss, degradation, fragmentation, introduction of invasive species). Other major change events include disturbances such as fire (Turner et al. 1994), pest or pathogen outbreaks (Kelly et al. 2008), and windthrow or timber extraction (Bormann and Likens 1994, Delcourt et al. 1982).

Many of these processes are directly or indirectly driven by human activities. However, the ultimate driver of change in modern forest landscapes is human population growth (Groom et al. 2006, Meyer and Turner 1992). The world's population has grown from 1×10^9 inhabitants in 1800 to more than 7×10^9 today (UN 2011). Population will continue to increase in most regions of the world except Europe and Japan, where significant decreases are expected. The future population may be as high as 11×10^9 in 2050 and 16×10^9 in 2100 (UN 2011). Less-developed regions, which currently host 82 % of the world's population, are growing at a much faster pace (UN 2011).

Landscapes of regions with fast population growth will likely suffer more drastic changes, mainly through land-use change, degradation or destruction of forest ecosystems, and increased forest fragmentation. These processes, combined with climate-driven change, may have disastrous consequences for a wide range of ecosystem services and, eventually, for human well-being (Leadley et al. 2010). Although there are many examples of changes in forest landscapes driven directly by recent population growth (Bradshaw 2012, Zhao et al. 2013), rural depopulation and the concentration of populations in cities may contribute more strongly to forest landscape change than was previously expected (DeFries et al. 2010).

1.4.2 Drivers and consequences of landscape change

Though most of the drivers discussed in this chapter affect landscapes at the microscale, long-term land-use change occurs at the mesoscale. Forest management simultaneously deals with the microscale (forest stands, planning units) and the mesoscale (forest estates, planning regions), but the majority of changes in forests, their causes, and their effects are increasingly addressed at the latter scale. In particular, landscape-dynamics processes contribute to the long-term stability of landscapes; that is, they do not significantly affect landscape structure and functioning over time, at least in comparison with natural trends driven by large-scale factors such as climate. However, most processes that occur at multiple spatial and temporal scales drive change towards new states, with new patterns and functions, and this has implications for the services provided by forest landscapes. In this section, we will analyze two recent (or recently studied) factors that are both drivers of landscape change and major research and management challenges: land-use change and changes in forest management concepts and practices.

1.4.2.1 Land-use change

Driven by social, economic, or political factors and influenced by environmental constraints, land uses have been profoundly modified throughout history in most parts of the world. These changes are an ongoing process that is continuing to affect ecological processes (FAO 2012), making this topic of interest both generally and in landscape ecology (August et al. 2002). The study of land-use change is complex because of the many factors (drivers) and the interactions among them that operate at multiple temporal and spatial scales and because of the diversity of physical and biological factors that are affected by the changes (Lambin et al. 2003). In addition to the direct local effects of land-use change, large-scale effects on both patterns and processes are expected to occur at landscape, regional, and global scales. Moreover, the relationship between land-use change and changes in landscape patterns and processes is not linear. One of the consequences of this nonlinearity is that changes in land use can have larger-than-expected effects on the structure, and concomitantly on the functioning, of forest landscapes. We will briefly discuss several land-use change processes that have affected forest landscapes in recent decades: agricultural expansion and intensification, agricultural abandonment, deforestation, and forestry intensification.

The expansion of agriculture has affected forest landscapes more strongly than just about any other factor during the last 10 000 years. The majority of agricultural land has been established on forest soils, leading to a decrease in forest area from 6×10^9 ha to the current level of 4×10^9 ha (FAO 2012). In Europe and parts of North America and Asia, this transition occurred in historical times and is largely finished, but the process continues in the rest of the world. Although the rate of expansion of agriculture is decreasing (FAO 2002), the pressure from agriculture on forest ecosystems remains high. Agricultural areas are expected to increase by 120×10^6 ha in developing countries by 2030, mostly due to the establishment or expansion of intensive cultivation of major food crops (FAO 2002). In regions such as East Asia, South Asia, the Near East, and North Africa that have already reached full use of their existing arable soil, agriculture will expand into forest landscapes that have survived previous expansion cycles. In the coming decades, the predicted expansion and intensification of agriculture is expected to affect the atmosphere, climate, soil, water, and biodiversity, and these effects may be cumulative.

The process of agricultural abandonment usually affects areas with low crop-production potential (e.g., low soil fertility, difficult topography, and climatic constraints) and low human density. Together with other drivers of change such as depopulation, incentives from markets, industrialization, poor adaptation of agricultural systems to local conditions, and land mismanagement, significant abandonment of agricultural activities has occurred in several regions in the world, but most frequently in Europe (Benayas et al. 2007). The effects of abandonment on the landscape structure depend on the matrix in which abandonment occurs and on the magnitude of the associated change. Abandonment in agriculture-dominated areas increases landscape heterogeneity by increasing landscape richness, diversity, evenness, and edge abundance and diversity, as well as increasing the variability in the sizes and shapes of landscape units. Functionally, landscape processes tend to be strongly influenced by the new systems that become established in the abandoned areas. Given that abandoned land generally becomes dominated by woody plants, agricultural landscapes often revert to forests within a few decades following abandonment. In landscapes where the matrix is mostly composed of natural or seminatural cover types, agricultural abandonment leads to a loss of heterogeneity and the potential loss of local diversity (Navarro and Pereira 2012). Fire, which is usually absent from agriculture-dominated landscapes, is promoted in these more natural landscapes by local accumulation of fuel and increasing continuity of highly flammable units within the landscape (Moreira and Russo 2007).

Deforestation is another complex land-use change process associated with the conversion of forest to a different land-use or cover class. The annual net loss of forests during the last decade was nearly 5.2×10^6 ha, which is the rate after accounting for the positive effects of afforestation (FAO 2010). Deforestation rates have been decreasing worldwide, but on different trajectories. Temperate regions reached their maximum deforestation rates prior to 1700, whereas tropical regions reached their maximum rate from 1950 to 1979 (FAO 2012). Although deforestation in temperate regions is currently balanced by reforestation, net deforestation remains high in tropical regions (FAO 2012).

Although agricultural expansion is a major cause of deforestation, there are many other causes, including unsustainable logging related to the demand for fiber and fuel, cattle grazing, infrastructure construction, urbanization, and interactions among these factors (FAO 2012). In addition, ancient agricultural systems such as slash-and-burn cultivation are still in use in many tropical regions. Deforestation is also associated with processes that act at multiple scales, such as urban growth, road construction, and climate change, in complex feedback loops, making the prediction of landscape change and its effects a difficult task (Freitas et al. 2010, Lambin et al. 2003). Deforestation can also result from habitat degradation. In this case, processes such as selective logging, insect pests and diseases, natural disasters, and invasive species affect the conservation of forest ecosystems (FAO 2010).

Deforestation is a typical landscape-level process. In addition to changing the vegetation composition, it affects other structural features of the landscape such as patch size (Bélanger and Grenier 2002); isolation, fragmentation, and connectivity (Lira et al. 2012, Riitters et al. 2002); edge dynamics (Laurance et al. 1998, Numata et al. 2009); and landscape stability (Metzger 2002). The ecological processes affected by deforestation include fire occurrence and intensity (Armenteras et al. 2013), species dynamics (Laurance et al. 1998), water yield (Sahin and Hall 1996), and ecosystem degradation and biodiversity loss (Bradshaw 2012, Brook et al. 2003).

In some parts of the world, forestry intensification is as significant as agricultural expansion and intensification in terms of its effects on forest landscapes. Whereas agricultural intensification requires good soil, weather, and terrain conditions, forestry, even when intensive, is less demanding. Thus, forests can grow over larger areas, including less-fertile soils and rough terrain, and intensification affects a diverse set of ecosystems, including native forests. Although planted forests represent just 7 % of the world's forests, they are concentrated in East Asia, Europe, and North America (FAO 2010), where they affect landscapes strongly. The landscape-level effects of this process vary with the land-use history and the tree crop species. Plantations have been established under intensive management regimes based on exotic species such as *Eucalyptus* in South America and other regions of the world. When such plantations are established in close contact with native forests or instead of local forests (Cossalter and Pye-Smith 2003), their main effects are land-use change and the creation of edge effects. Bamboo plantations in Africa or Central America and rubber plantations in South and Southeast Asia and West Africa are crops with a potential effect on native forests. The established plantations are ecologically simpler and are managed to maintain that structural and functional simplicity; they therefore cannot support rich plant and animal communities, leading to impoverishment of local and eventually regional diversity. Under certain circumstances, however, plantations can provide habitat connectivity at the landscape level, despite their poor habitat quality, and thereby help to maintain population processes and diversity (Barlow et al. 2007).

In some regions, forest plantations are established in degraded areas, usually after previous deforestation and intensive agriculture or in areas where forest cover has been historically replaced as a result of land-use and cover change. In each case, the landscape prior to afforestation was dominated by non-forest land uses. Degraded land is particularly common in tropical regions with impoverished soils that are fragile and have poor resistance to disturbance and in semiarid and arid areas, such as much of northern China, that are also highly vulnerable to degradation. In such cases, even intensive forestry based on exotic species can have positive impacts in terms of organic matter inputs, energy and nutrient cycling, and providing habitat, and these changes can influence broader areas than just the local direct effects of the plantations. For example, see the thorough review of ecosystem functions and services associated with plantations in Brazil and elsewhere by Brockerhoff et al. (2013). Restoration of these areas based on the establishment of forest plantations, including landscape-scale measures, has been proposed (Lamb et al. 2005). However, afforestation programs in degraded areas have also been associated with negative ecological and socioeconomic effects (Cao et al. 2011).

Historical land-use and cover change has been particularly common in Europe and North America. In Europe, forests account for 34 % of the land area, compared to 80 % around 2000 years ago (FAO 2012). This relatively high coverage is only because of afforestation campaigns conducted during the last 100 to 150 years. Many of these planted forests are managed in relatively intensive ways, such as *Eucalyptus* plantations in Spain and Portugal, poplar (*Populus* spp.) plantations in Italy, or willow (*Salix* spp.) coppices managed for energy production in Sweden. As in Europe, the rapid deforestation that took place in North America during the nineteenth century was rapidly followed by reforestation or natural regeneration of abandoned agricultural land. For example, the United States and Canada have planted an annual average of 371 000 ha of forest since 2000 (FAO 2010).

1.4.2.2 Changes in forest management

Forest planning, management paradigms, and forestry practices have changed dramatically during the twentieth century, not just in terms of the concepts and objectives (e.g., the move from sustained timber yield to ecosystem management) but also in terms of the scale at which forest management is addressed, which expanded from the cohort or stand to large-scale heterogeneous landscapes (Brunet et al. 2000). This shift in scale was influenced by the development of landscape ecology in the 1980s, which provided the conditions and a theoretical framework for the application of forest landscape ecology within forestry. Landscape-level sustainability criteria and indicators are used today to support decisionmaking in forest management to ensure the sustainable provision of forest products and the maintenance of ecological functions. Clearcut size, the abundance of edges, connectivity, and the presence of corridors, among others, are important landscape-level variables that are relevant in today's forest management because of their relationship with ecological processes. Forest landscape ecology informs management not only by supplying knowledge of the interactions between patterns and functions in forest landscapes but also by providing conceptual and methodological tools to support planning and management.

The expansion of emerging concepts such as sustainable management, ecosystem management, multifunctionality, and adaptive management in forestry during the late twentieth century has resulted from changing public perspectives towards forests and natural resources and increased scientific knowledge. These novel approaches have affected decisionmaking, forest management, and forest product markets, as well as the structure and functioning of forest landscapes.

Sustainability has become the most important goal in planning and management of forests. Emerging from the "Statement of Forest Principles" and the "Convention on Biodiversity" that were agreed to at the United Nations Conference on Environment and Development (UNCED) held in Rio de Janeiro in 1992, the concept rapidly became the background for major international initiatives directed at defining principles and practices for sustainable forestry as well as for certification programs. Schlaepfer and Elliott (2000) and Burley (2001) provide a detailed history of sustainable forestry.

Broad scales are required to address sustainability in natural and managed systems in the fields of land planning, nature conservation, and land management, including forest management (Cary et al. 2009, Christensen et al. 1996, Forman 1995, Lubchenco et al. 1991). Sustainable forestry initiatives and certification programs address landscape patterns and processes in several ways. Many of the criteria and indicators of the Montréal Process (1999) and the Pan European Forest Certification (http://www.pefc.org/) require that large scales be defined and applied. Criteria such as water conservation, habitat and species conservation, maintenance and encouragement of the productive functions of forests, and maintenance of ecosystem health or integrity rely strongly on a consideration of the spatial attributes of ecosystems at broad scales. At the national level, the program's guidelines also require a landscape-scale approach, which includes the establishment and management of riparian buffer zones and wildlife corridors, defines the size of harvested areas, and prescribes adjacency rules. In addition, compliance with sustainable forestry programs involves the application of landscape ecology concepts and methods (e.g., FSC 2010). In the Sustainable Forestry Initiative in the United States, for example, several biodiversity- and water-related processes, criteria, and indicators can only be addressed from a landscape perspective (Azevedo et al. 2008).

Ecosystem management is the designation of the management policy adopted by the USDA Forest Service and other federal agencies of the United States in the 1990s based on the application of ecological principles in forest management (Rauscher 1999, Schlaepfer and Elliott 2000). Despite the numerous interpretations of the concept, it deals with many complex ecological and management themes, including holism and a consideration of cross-scale interactions among a system's components, defining the ecological boundaries of systems at multiple scales, maintaining diversity of patterns and processes at all scales, research, managing and using existing data, monitoring, adaptive management, and accounting for interactions between ecosystems and humans (Christensen et al. 1996, Franklin 1993, Grumbine 1994, Szaro et al. 1998). Its ecological foundations and the prerequisite for addressing large temporal and spatial scales make this approach intimately related to forest landscape ecology (Crow 1997, Franklin 1993). A full range of applications in forestry is provided by Kohm and Franklin (1997).

Multifunctionality in forestry refers to the delivery of multiple outputs from the process of forest management that are obtained by taking appropriate measures to optimize their production (i.e., multifunctional management). Although multifunctionality is related to the concept of multiple use and may have evolved from the Multiple-Use Sustained Yield Act of 1960 in the United States to simultaneously address timber, range, water, recreation, and wildlife values, the term has been expanded to encompass a broader range of ecological functions and services. Multifunctional management also overlaps considerably with the concept of sustainable management, as defined by Farrell et al. (2000), since the objectives of sustainable forest management consider multiple roles, functions, and outputs of forest systems, and the terms are often used interchangeably. Spatial multifunctionality is an extension of the concept to the landscape level, by considering multiple outputs from the diverse land-use and cover types present within a given landscape

(Carvalho-Ribeiro et al. 2010). Landscapes, by their intrinsic heterogeneity and complexity, are consequently seen as multifunctional systems (Willemen et al. 2010).

Ecosystem services are not, in a strict sense, inherently a management concept, but their impact on the management of ecosystems and landscapes will nonetheless be massive in the coming years. The use of the ecosystem services concept in management provides a quantitative and objective methodological framework to support decisionmaking in complex systems that are being managed, simultaneously, for multiple targets and objectives. This is particularly relevant in defining management strategies and technical solutions for the application of philosophies such as multifunctionality or sustainability in forestry. It is also relevant for landscape ecology, since many of the ecosystem services usually associated with forests are actually landscape services, including water-related services (e.g., yield and quality regulation), disturbance regulation services (e.g., fire, flooding), or cultural services (e.g., esthetics).

Although the ecosystem services framework is based on human needs, it is a long-term, ecologically based approach to management because it relies on a holistic perspective that requires the maintenance of fundamental ecosystem patterns and processes, including biodiversity. Chapter 5 provides a thorough discussion of ecosystem services and their valuation in forest systems.

Adaptive management is the process of adjusting management practices as more knowledge is gathered through research, monitoring, or experience and as the system's behavior changes in response to management (Holling 1978). The concept has been developed and adapted to the management of natural resources, where predictability is low and uncertainty is high, particularly when available knowledge is limited, and falls within the scope of sustainability (Walters 1986). Adaptive management is a growing component of strategies to adapt natural resources to climate change. It has been addressed in silviculture as an operational null hypothesis for the management of unstudied forest systems (Oliver and Larson 1996) and in ecosystem management as a way of dealing with complex unknown and changing systems (Kessler et al. 1992). Most proposals for the application of this concept, however, come from the field of forest biodiversity conservation (Lindenmayer and Franklin 2002, Lindenmayer et al. 2006).

Our discussion in this section indicates that forest management has changed significantly during the past century, both conceptually and in practice, and it is likely that these changes will increasingly affect the patterns and processes of forest landscapes. Previous evaluations of these concepts have included the changes observed in the structure of forested landscapes that have undergone different management practices (e.g., Crow et al. 1999, Spies et al. 1994) or the use of modeling and simulation to predict changes in structure as a function of management practices such as the choice of regeneration method, harvest and regeneration scheduling, and the spatial pattern of harvested areas (e.g., Baskent 1999, Crow et al. 1999, Franklin and Forman 1987, Gustafson and Crow 1996, Radeloff et al. 2006, Shifley et al. 2000, Spies et al. 1994). The effects of forest policy and management objectives on spatial patterns have also been analyzed through simulations (Cissel et al. 1998, Gustafson and Loehle 2008, Hagan and Boone 1997).

The effect of management-caused changes in structure on landscape functioning has been addressed through modeling and simulation for wildlife habitat suitability (Hansen et al. 1992, Larson et al. 2004, Li et al. 2000, Shifley et al. 2006), plant succession and disturbance (He et al. 2002, Kurz et al. 2000), metapopulation dynamics (Akcakaya et al. 2004), and hydrological processes (Azevedo et al. 2005).

Despite these advances, the application of landscape ecology approaches or methods to real-world forest management, particularly by the forest industry and the private sector, has been limited. Although the forest industry tends to consider landscape issues in management, mostly due to the desire to achieve certification, the implementation of a forest landscape approach has mostly been superficial. The management of *Eucalyptus* plantations by some forestry companies in Brazil is a possible exception. These companies have applied sustainable forestry principles and methods using GIS-based software that was developed to assess and monitor landscape diversity (with stands defined in terms of clone and age), water balance, and the size, edges, core areas, proximity, vegetation diversity, and value of conservation areas based on conventional and customized landscape metrics (Ferraz and Ferraz 2009). Insufficient implementation of forest landscape ecology in forestry practices is discussed in detail by King and Perera (2006) and will be addressed in the final chapter of this book.

1.5 Trends and roles of forest landscape ecology in the context of change

1.5.1 Why is forest landscape ecology essential within the framework of global change?

Land-use change in forest landscapes that is driven by the major processes described in this chapter creates forest expansion or forest fragmentation, which are particularly relevant processes that require further attention from forest landscape ecologists, particularly to support forest management. Forest cover is expected to keep increasing in some parts of the world, such as Europe and North and South America. The expansion of planted and naturally established forests associated with rural abandonment and the intensification of forestry are causing fast changes in the landscapes of the southern and eastern parts of Europe (Benayas et al. 2007, Keenleyside and Tucker 2010, Navarro and Pereira 2012, Proença et al. 2012). The prospects for further forest expansion in the coming decades are high given the availability of abandoned farmland (Keenleyside and Tucker 2010). Trends include forest intensification (south), multifunctional forestry (in other areas), and the transition of previously rural landscapes back to wilderness (Navarro and Pereira 2012). In other parts of the world, abandonment of agricultural land (North America) and reestablishment of forest in previously degraded land (South America and China) are also expected to expand forest cover.

In each of these cases, a landscape perspective and the associated theory, methods, and tools are required in any attempt to manage forest landscapes that are undergoing development (or a return to previous forested states) to ensure multi-functionality or sustainability or just to ensure a smooth transition between “system states”. Particularly relevant is the management of landscapes for the sustained provision of ecosystem services, as this is only possible at landscape scale. Landscapes that became dominated by forest develop different processes. In most areas where forest expansion is occurring, fire is a major disturbance and might become a major driver in subsequent change; therefore, a landscape’s fire regime is an important reference for both landscape and stand management.

In contrast, many forest landscapes around the world are expected to experience habitat loss and fragmentation in the future. These processes attract attention from society in general, since they will affect some of the world’s most biodiverse regions and will have significant effects on global biological diversity and ecosystem services. Forest fragmentation is one of the most serious threats to biodiversity, but fortunately, it is also one of the major research subjects covered in landscape ecology, and particularly in forest landscape ecology.

In addition, all forest landscapes are being influenced by climate change. Since many processes affected by this type of change occur at the landscape scale or are affected by other processes that occur at this scale, forest landscape ecology has an increasingly relevant role in the context of global change. Through changes in the temporal and geographic patterns of temperature and precipitation and changes in their uncertainty (e.g., interannual variability, the frequency of extreme events), climate change will affect species distribution and ecosystem productivity, disturbance regimes, biological invasions, and resilience (Hansen et al. 2001) and, most importantly, will affect the role of ecosystems as service providers (Schroter et al. 2005). In some cases, these effects have already been detected (see review in Hannah 2011). In return, landscape changes will affect climate, both locally and globally, through changes in albedo, evapotranspiration, and emissions of greenhouse gases through complex feedback processes. Interactions among climate change, land-use and cover change, management, and large-scale disturbance (e.g., fire) will be complex and difficult to forecast (Dale et al. 2001).

Changes in an ecosystem’s species composition due to extinction or shifts in distribution ranges may influence forest ecosystem function and the survival of numerous plant and animal species, with subsequent effects on landscape-level systems (e.g., see Chap. 2 of this book). In terms of conservation, measures adopted since the nineteenth century, such as the establishment of protected areas and conservation networks, will need to be adjusted to account for the expected changes in species distribution during the present century (Araujo et al. 2011, Hannah et al. 2007). For example, see Chap. 2 for a discussion of the need for “assisted migration” of species in response to climate change. On the other hand, current landscapes play a fundamental role in species redistribution in response to climate change through the effect of landscape structure (and the key role of connectivity) on the spread of organisms. Changes in the distribution of vegetation types are also occurring. In Europe, climate change will increase the dominance of forest landscapes in the near future, thereby positively influencing ecosystem services such as productivity and esthetics but negatively influencing water availability and vulnerability to forest fires or the ability to sustain high-quality timber production (Hanewinkel et al. 2013, Schroter et al. 2005). In other parts of the world, such as North America (Bachelet et al. 2001) and Asia (Weng and Zhou 2006), the trend is also an expansion of forest systems, but the effects remain unknown. Pest outbreaks and changes in fire regimes will become active drivers in landscapes affected by climate change. Increasing frequency or severity of pest outbreaks and disease epidemics or of forest fires is predicted in response to changes in climate that affect the composition of forest ecosystems and the landscape pattern (Dale et al. 2001, Westerling et al. 2006). Climate change will therefore affect both processes that occur at the landscape scale and management of the ecosystems and landscapes affected by these processes, and this will require the integration of climate change into future planning and management of forests. Approaches to deal with climate change will be increasingly based on concepts such as adaptation and resilience, which are being studied more intensely from both theoretical and applied perspectives and from a landscape perspective (Heller and Zavaleta 2009, Opdam et al. 2009).

1.5.2 Roles of forest landscape ecology in contemporary forest science and management

Based on what we have discussed thus far, forest landscape ecology has contributed and will continue to contribute to helping researchers and managers to deal with change and its complexity. The major goal of forest landscape ecology is to minimize the risks and the effects of change on ecological sustainability and human well-being by providing a better understanding and description of change and its effects from a theoretical perspective while, as an applied science, simultaneously informing the management and planning of forest landscapes.

From a scientific perspective, landscape ecology offers the foundations (theory, approach, scale, research methods and tools, knowledge) to provide the following: (1) full understanding of the drivers of change and their nature, scale, complexity, and interactions; (2) full understanding of the effects of change on patterns, processes, and services; and (3) full availability of methodological and practical tools to monitor landscape change. Understanding the drivers, processes, and effects of change is a rather difficult task considering the inherent complexity of the systems under analysis, and the difficulty is increased by the complexity of change, particularly when that change results from interactions at multiple scales. For this reason, monitoring of landscape change isolating the weight, scale, and mechanisms of different drivers of change and understanding of interactions at multiple scales are of utmost importance in the development of this field.

From a management perspective, the potential roles of forest landscape ecology are to (1) inform the planning, management, and design of forest landscape systems under changing conditions; (2) support the multifunctionality and sustainability of forest landscapes under changing conditions; (3) integrate change into the disturbance regimes that result from management and planning; (4) ensure sustained provision of ecosystem services, particularly those related to biodiversity conservation; and (5) support the definition of adaptation strategies, approaches, and outcomes. Forest landscape ecology must, therefore, provide the foundations, the methods, and the tools to deal with change in a management context. It must also support the incorporation of generally accepted concepts such as sustainability, multifunctionality, ecosystem services, and adaptive management into more conventional management approaches. It should also provide support for changes in management in order to adjust forest landscapes and the management methods to, for example, account for changes in disturbance regimes (e.g., fire, pests, storms), biological invasions, drought, human pressures, and other change processes. Given the potentially high species extinction rate that will occur under the projected fast environmental change and the irreplaceability of biodiversity in sustaining ecosystem structure and functioning, biodiversity conservation planning at broad scales under future land, climate, and disturbance conditions should be our top priority.

1.5.3 How this book addresses forest landscape ecology and change

In this book, we have addressed change in forest landscapes from both theoretical and practical perspectives. Based on the existing management traditions in forest landscape ecology and the need to contribute more and better solutions to deal with change and its effects in real-world situations, we developed the book outline by simultaneously considering the underlying processes of change (climate, human activities, disturbance regimes), the effects of change on ecosystem and landscape processes (carbon, biodiversity, disturbance), the methods to monitor and assess change (landscape monitoring), the approaches to deal with changes in management (ecosystem services), and the integration of knowledge in forest management at the stand and landscape scales (forest management and change).

After our introduction, in which we discuss the analysis of change in forest landscapes and the role of forest landscape ecology in a changing context (Chap. 1), Louis Iverson and his colleagues provide a detailed analysis of the consequences of climate change on the distribution of tree species and on the interactions of plants, populations, and ecosystem processes with landscape patterns (Chap. 2). Next is a discussion of the processes that simultaneously drive change and are the result of other drivers, and the complex interactions among them. Francisco Rego and Joaquim Silva explore the case of fire as an agent of disturbance based on the Portuguese experience (Chap. 3), and Juliana Farinaci and her colleagues explore the transition from deforestation to forest restoration in São Paulo, Brazil, and Indiana, United States, emphasizing their causes and consequences (Chap. 4). Chapter 5, by Cristina Marta-Pedroso and her colleagues, is directed towards changes in socioeconomic perspectives related to ecological and social processes and functions and explores the application of the ecosystem services concept and related methodologies in forestry decisionmaking. In Chap. 6, Jiquan Chen and his colleagues analyze the processes of carbon sequestration and storage and their dynamics in forest ecosystems and landscapes, as well as their interaction with climate change. Chapter 7, by Santiago Saura and his colleagues, is dedicated to the major effects of landscape change on biodiversity at multiple scales, with an emphasis on habitat amount, quality, fragmentation, connectivity, and heterogeneity. Chapter 8, by Valentín Gómez-Sanz and his colleagues, is dedicated to the theoretical and technical aspects of procedures for monitoring and assessing changing landscapes. The implications of changes in forest management approaches and methods are discussed in detail by Robert N. Coulson and his colleagues in Chapter 9, which explores the author's contributions towards better management of forest landscapes in response to the several sources of change that are currently affecting forest landscapes or that will affect them in the future. Common to most of the chapters in this book is the objective of providing knowledge transfer from the scientific sphere to the sphere of real-world management (e.g., monitoring techniques, adaptation to fire regimes, adaptation to climate change, biodiversity conservation, carbon sequestration and storage, and valuation and evaluation of human values and ecosystem services). We conclude by summarizing the main achievements in this book, discussing the challenges that forest landscape ecology faces in the future, and describing the next steps that are required to advance this field (Chap. 10).

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Chapter 2
Climate as an agent of change in forest landscapes

Louis R. Iverson, Anantha M. Prasad, Stephen N. Matthews, and Matthew P. Peters

Abstract Climate is the primary force that controls forest composition and the broad-scale distribution of forests. The climate has always been changing, but the changes now underway are different—they are faster and they are intermingled with other disturbances promoted by increasing human pressures. The projected climate change during the twenty-first century will alter forest habitats—dramatically for some species. These pressures will simultaneously affect the survival, growth, and regeneration of a species. Here, we present an approach to visualizing the risk to individual tree species created by climate change by plotting the likelihood of habitat change and the adaptability of trees to those changes. How will the forests actually respond? Many factors play into the final outcomes, including the vital attributes and abundance of a species, its migration potential, the fragmented nature of the habitats in the landscape into which the species must move, and other factors. Our research is attempting to address each of these factors to inform a more realistic picture of the possible outcomes by the end of the century. We describe three programs that have been developed to support this analysis: DISTRIB, which empirically models the distribution of suitable future habitats under various climate-change scenarios; SHIFT, which is a cell-based spatial model that simulates species migration across fragmented landscapes; and ModFacs, which accounts for the impacts of 9 biological traits and 12 disturbance factors on final species fates. We conclude with a discussion of research needs and how humans can potentially assist forests in their adaptation to climate change.

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2.1 Climate change

The climate has always been changing. However, our increased use of fossil fuels has made the anthropogenic component more prominent than ever before, and unusually rapid change is projected to occur by the end of the twenty-first century (IPCC 2007). Globally, temperatures (and especially nighttime minimums) have risen, while many places are getting wetter, albeit as a result of more frequent extreme events (Alexander et al. 2006). A recent study by the Berkeley Earth Surface Temperature Project (http://www.berkeleyearth.org; Fig. 2.1) has provided reliable evidence of a rise in the average global land temperature by approximately 1 °C since the mid-1950s (Rohde et al. 2012).

IPCC (2007) has determined that climate change is accelerating and that changes will continue. Many institutions have been modeling future climates, and all

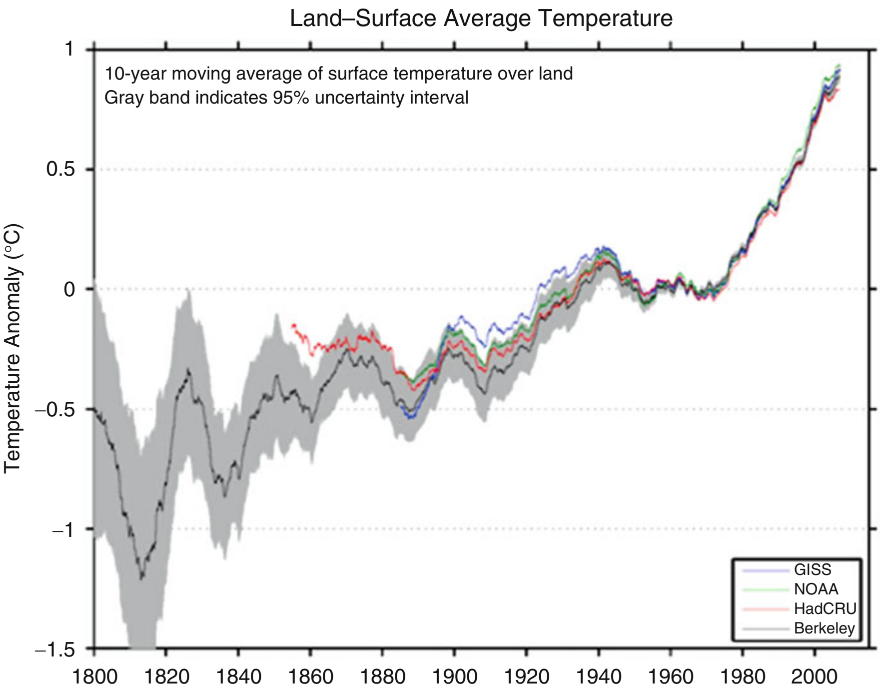

Figure 2.1 Changes in land-surface temperatures since 1800. Reprinted with permission from the Berkeley Earth Surface Temperature Project (2012, http://berkeleyearth.org/analysis/). The graph shows the global mean annual land-surface temperature using a 10-year moving average. Anomalies are expressed relative to the mean from January 1950 to December 1979; positive values represent a temperature increase. The *grey band* indicates the 95 % confidence interval. GISS, Goddard Institute for Space Studies; NOAA, National Oceanic and Atmospheric Administration; HadCRU, Hadley Centre of the U.K. Meteorological Office; Berkeley, Berkeley Earth Surface Temperature Study

scenarios predict a warmer world in the coming decades, particularly in the latter part of this century (IPCC 2007). Future precipitation patterns are less consistent, with some parts of the globe showing net increases and others showing net decreases. What is consistently projected, though, is a more vigorous hydrologic cycle because of the greater heat energy in the atmosphere. Thus, it is projected that heavy precipitation events (storms) will increasingly provide a larger proportion of the total annual precipitation, resulting in more runoff and floods (Lenderink and van Meijgaard 2008, Milly et al. 2002), but also more and longer periods without rain and droughts (Burke et al. 2006, IPCC 2007, Seidel et al. 2008). Indeed, a recent study showed strong evidence linking the extraordinary number and impact of disastrous heat and precipitation events that occurred between 2000 and 2011 to the human influence on climate (Coumou and Rahmstorf 2012). Another study pointed to the amplified heating of the Arctic as a key factor responsible for the elevated number of extreme events in the northern hemisphere (Seminov 2012). Coumou and Rahmstorf (2012) clearly describe how we might think about this pattern:

"Many climate scientists (including ourselves) routinely answer media calls after extreme events with the phrase that a particular event cannot be directly attributed to global warming. This is often misunderstood by the public to mean that the event is not linked to global warming, even though that may be the case—we just can't be certain. If a loaded dice [sic] rolls a six, we cannot say that this particular outcome was due to the manipulation—the question is ill-posed. What we can say is that the number of sixes rolled is greater with the loaded dice (perhaps even much greater). Likewise, the odds for certain types of weather extremes increase in a warming climate (perhaps very much so). Attribution is not a 'yes or no' issue as the media might prefer, it is an issue of probability."

2.2 Forests and a changing climate

At a coarse scale, climate is the primary driving force for the location, composition, and productivity of forests (Shugart and Urban 1989, Woodward and Williams 1987). Therefore, changes in climate will yield changes in forests. These changes have also always occurred in response to climate change (e.g., Davis and Zabinski 1992, Delcourt and Delcourt 1987), and the combination of species that comprise a forest also changes through time (Webb 1992). A mounting number of studies provide evidence that such changes continue to occur (Bolte et al. 2010, Woodall et al. 2009). Although there is empirical evidence of tree species moving to higher altitudes (Beckage et al. 2008, Holzinger et al. 2008, Lenoir et al. 2008), there is minimal evidence documenting a progression of tree species in a poleward direction in this century (Zhu et al. 2012). However, some case studies have shown changes in species composition over time, with more recently arrived species arriving from lower latitudes (Schuster et al. 2008, Treyger and Nowak 2011). In addition, meta-analyses have provided increasing evidence of species movements from a large suite of taxa (Chen et al. 2011, Parmesan and Yohe 2003). The mean extinction risk across all taxa and regions has been estimated at 10 to 14 % by about 2100 (Maclean and Wilson 2011) despite the "Quaternary conundrum", which relates to a lower-than-expected rate of extinction during the Quaternary ice ages (Botkin et al. 2007).

The paleoecological record shows a remarkable change in tree distributions. In eastern North America, for example, the pollen record shows massive migrations since the last glaciation (ca. 18 000 years before the present). These migrations have been matched to concomitant changes in temperature (Davis 1981). Spruce (*Picea* spp.) and fir (*Abies* spp.) in the northeastern United States have shown particularly great changes in their distribution during the last 6000 years and appear to be destined to retreat northwards back into Canada as the climate warms (DeHays et al. 2000). The same phenomenon has been observed in Europe, where the glacial history and climate have acted as key controls on tree distribution and species richness (Svenning and Skov 2005, 2007).

Thus, suitable habitats for tree species appear to be changing, but many models predict that these changes are likely to accelerate throughout this century. The models of several groups show these potential trends (Crookston et al. 2010; Delbarrio et al. 2006; Dobrowski et al. 2011; Iverson et al. 2008b; Keith et al. 2008; McKenney et al. 2007, 2011; Morin et al. 2008; Ravenscroft et al. 2010; Scheller and Mladenoff 2008), and a recent report suggests these studies may be underestimating the actual change (Wolkovich et al. 2012). Uncertainty and extraordinary challenges will continue to confront species modeling (Araújo and Guisan 2006, Pearson et al. 2006, Thuiller et al. 2008, Xu et al. 2009), although multiple approaches are being developed in attempts to improve projections (Araújo and Luoto 2007, Elith et al. 2010, Franklin 2010, Iverson et al. 2011, Matthews et al. 2011, Morin and Thuiller 2009).

2.3 Climate-related drivers for forests and forest changes

Climate constraints interact with the physiological and ecological attributes of trees to produce the broad-scale characteristics of forest composition and productivity. These forces, along with broad-scale land-use and management manipulations, are the primary determinants of the forests we see today. At a finer scale, topography, local climate, and soil conditions play a primary role in determining forest characteristics, and many features such as species composition, productivity, and regeneration success are strongly determined by slope position and aspect along with the soil's water-holding capacity (Iverson et al. 1997, Kabrick et al. 2008, McNab 1996). Thus, scale is important, especially in climate and climate impact models. The spatial resolution of the original general circulation models was coarse, with cells spanning 1° to 4° (Tabor and Williams 2010). Thus, downscaling of these data is required, and though such efforts will be very helpful, they will by their nature be imprecise at a fine scale (Tabor and Williams 2010).

Many drivers of forest change are also related to climate, either directly or indirectly. Obviously, land-use change, management or mismanagement, herbivory, pest outbreaks, and other impacts are critically important forces of change at certain times and places, but climate provides the overall conditions that create constraints on a forest's characteristics. Climate change creates two primary, and interrelated, categories of impacts for trees: *maladaptation* and *disturbance* (Johnston 2009). Maladaptation refers to a situation in which the local conditions to which a species is adapted begin to change faster than the species can move or adapt. Examples include a reduction of moisture availability, the CO_2 fertilization effect, permafrost melting, drying or creation of wetlands, and changes in snow depth. Disturbance refers to the suite of biotic and abiotic onslaughts that occur as a result of climate change or that are in some way encouraged by climate change. Many disturbance regimes that directly alter forests are expected to increase in frequency, intensity, or both as a result of climate change (Dale et al. 2001). Evidence is mounting that such climate-linked disturbances are increasing, including an increase in fire frequency in the western United States and elsewhere (Littell et al. 2009, Liu et al. 2010, Westerling 2006), an increased northward prevalence of mountain pine beetle (*Dendroctonus ponderosae*) outbreaks in western North America (Bentz et al. 2010, Hicke et al. 2006, Kurz et al. 2008, Sambaraju et al. 2012), an increasing risk from invasive species (Dale et al. 2009, Dukes et al. 2009, Hellmann et al. 2008, Jarnevich and Stohlgren 2009, Mainka and Howard 2010), and an increasing evidence of drought-induced mortality (Adams et al. 2012, Allen et al. 2010, Hanson and Weltzin 2000, Peng et al. 2011).

Although some of the disturbance characteristics may be subtle, they may eventually reach a "tipping point" at which the change is enough to shift the competitive balance between species or to overwhelm a forest's compensatory mechanisms, leading to a change in the forest's composition. For example, an insect species may be able to overwinter just enough that its population levels gradually increase until they become sufficient to kill trees that were not previously at risk. This phenomenon has been shown for the mountain pine beetle in whitebark pine (*Pinus albicaulis*) forests in the Greater Yellowstone Ecosystem (Logan et al. 2010), the southern pine beetle (*Dendroctonus frontalis*) in the New Jersey pine barrens (Tran et al. 2007), and the hemlock woolly adelgid (*Adelges tsugae*) on hemlocks (*Tsuga* spp.) of the eastern United States (Fitzpatrick et al. 2012, Paradis et al. 2008). Some of these climate-induced or climate-enhanced factors have been shown to quickly alter forest characteristics, but even when the impact is more gradual, these factors can still greatly alter the biodiversity of an area. Though few or no single events can be attributed to climate change, the overall trend tends to support the hypothesis that the impacts of climate change are increasing.

2.4 Forest adaptation to climate change

Uncertainties abound in forest management, even in the absence of climate change (Bolte et al. 2009, Long 2009, Seidl et al. 2011). Most of the large uncertainties associated with climate change will remain, regardless of research progress, owing to the impossibility of predicting the impact of the next major pest, of the next hurricane, or of the series of climatic events needed to create a "perfect storm" for fire or drought mortality at a location where such events had rarely occurred or had not previously occurred. It is known, though, that one of the primary outcomes of climate change is a more vigorous hydrologic cycle and that extreme events with direct and indirect impacts on forests will therefore become more likely (Solomon et al. 2007). This increase in extreme events is already occurring, and their frequency is expected to increase substantially in the future (Coumou and Rahmstorf 2012, Huber and Gulledge 2011). Therefore, the direct and indirect risks to forests are expected to increase throughout the century. We therefore need methods to evaluate changes in the risk for a given species over time and to evaluate and implement policies and procedures that would allow or promote adaptation to the new climate and disturbance regimes. Essentially, any tree species whose microclimate changes enough for it to be growing outside its preferred niche has three options: move, adapt, or be extirpated (Aitken et al. 2008). In the rest of this chapter, we address the first two options after evaluating the concept of risk.

2.4.1 Strategic assessment of species adaptation requirements through risk matrices

To help forest species adapt to changing conditions, where this is appropriate, it is first necessary to evaluate the risk and develop appropriate strategies that respond to that risk (Millar and Stephenson 2007, Yohe and Leichenko 2010). Increasing a forest's resistance to climate change and its resilience is a key elements of adaptation, but triage may be necessary if a species cannot be protected in situ without incurring a cost that society is unwilling to bear.

We developed a visual tool called a *risk matrix* to assess risk and compare risks among species and among locations to "organize thoughts" around risk and forest adaptability for a particular region (Iverson et al. 2012). The tool was developed for the United States National Climate Assessment and is intended to provide an easily understood visual tool for focusing the conversation on management strategies at all levels. The intention is to use the tool for areas small enough that they do not have major disjoint habitats or species that gain and lose from climate change simultaneously within the same region, but not so small that they have too few cells for analysis. First, we defined "risk" as the product of the likelihood of an event happening and the consequences if it happens. We then categorized the matrix into three zones: (1) *watch*, which involves a relatively low risk but the need to remain vigilant; (2) *evaluate further and perhaps develop strategies*, which involves an intermediate level of risk; and (3) *develop strategies to cope with the risk*, which involves the highest level of risk (Yohe and Leichenko 2010). For forest trees, we interpret this risk (likelihood) as a potential for change based on the adaptability or resistance of the species to the impacts of climate change (consequences), and in this chapter, we will demonstrate this form of analysis for a species whose habitat is likely to

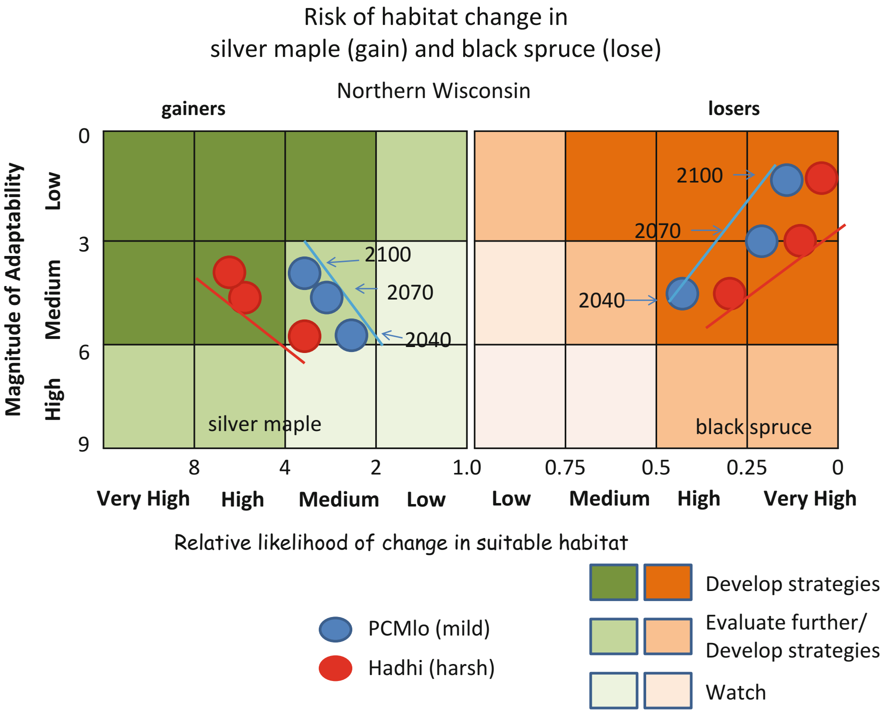

Figure 2.2 A risk matrix for a species whose habitat is likely to decrease (black spruce, *Picea mariana*) and a species whose habitat is likely to increase (silver maple, *Acer saccharinum*). The analysis is conducted for northern Wisconsin (United States). The trend lines are hand-drawn to approximate the trends modeled throughout the twenty-first century. PCMlo represents the PCM B1 scenario and Hadhi represents the Hadley A1fi scenario (IPCC 2007). Relative likelihood represents the ratio of future habitat in a given year to the current habitat

decrease, black spruce (*Picea mariana*), and for a species whose habitat is likely to increase, silver maple (*Acer saccharinum*). We conducted the analysis for northern Wisconsin (United States) between now and 2100 (Fig. 2.2).

In the context of changes in the amount of suitable habitat in response to climate change, we modeled the potential for an area to have suitable habitat for the selected species in the future relative to its current amount of suitable habitat. The *x*-axis is thus based on the difference in suitable habitat (i.e., the sum of importance values for all 20 × 20 km cells within the region of interest) between the current date and three future time intervals, which end around 2040, 2070, and 2100. In addition to these three dates, we also include predictions based on two widely differing scenarios for modeled climate change, PCM B1 and Hadley A1fi, to extract a range of potential risks associated with the IPCC projections of future climates (IPCC 2007, Nakicenovic et al. 2000). We view the ratio of future habitat to current habitat as being related to the likelihood of an impact on the amount of suitable habitat—the greater the potential change in habitat, the greater the likelihood of an impact.

For species that show a loss of habitat, the *x*-axis ranges from +1 (no change in habitat with time) to 0 (complete loss of habitat over time). In this analysis, black spruce shows a substantial future habitat loss by 2100, especially under the more severe Hadley A1fi scenario, and the species thus has a large likelihood of change on the "loser" side of the risk matrix (Fig. 2.2). On the other hand, silver maple shows a positive ratio of future to current habitat and is therefore in the "gainer" section of the risk matrix.

The *y*-axis is related to the adaptability of the species under climate change, based on a literature review to assess the biological traits of the species and its capacity to respond to various disturbances that may increase in frequency or severity in the coming century, compounded (or not) by climate change. We thus scored the adaptability of the species to cope with climate change; the lower the capacity to cope, the greater the risk of habitat loss and the greater the consequences of this loss. The data for this species-level analysis comes from an evaluation of the literature for 12 disturbance factors and 9 biological factors (Matthews et al. 2011). Relative scores were averaged for the biological and disturbance factors, then plotted to yield a composite modification factor score that was also modified for plotting on the *y*-axis in 2070 and 2100 based on the disturbance factors that were estimated to increase throughout the century. Further details are provided by Iverson et al. (2012).

In summary, we quantified the estimated risk for each species using the bounds of a harsh (Hadley A1fi) and a mild (PCM B1) scenario for the future climate and extrapolated the trends to 2040, 2070, and 2100 (Fig. 2.2). The matrix shows contrasting trends for the two species, but in both cases, managers will increasingly be required to develop strategies to cope with the risks created by the climate change that is currently underway—one set of strategies for silver maple, a species that may or may not need to be encouraged to become established, and one for black spruce, for which it may be necessary to establish protected refugia, enhance or maintain corridors that will permit poleward migration, or possibly even assist in this migration.

2.4.2 The need for species migration

Migration of species will be necessary over the long term as species reshuffle their distribution to adapt to their new climatic niches. Most species-distribution models show that the habitats for many species will often move large distances by 2100 (Iverson et al. 2008b, McKenney et al. 2011). Based on studies of pollen distributions during the Pleistocene, when forest cover was nearly complete across eastern North America, migration rates per century appear to range from 10 km (McLachlan et al. 2005) to 50 km (Davis 1981, Huntley 1991). With the modern fragmentation of forested land, estimates of migration rates are generally much lower (Schwartz 1993). Thus, there is little evidence to support the belief that migration by natural means will be able to keep up with the expected rate of change in habitats.

In one study, less than 15 % of new suitable habitat would have even a remote chance of being colonized by 2100 (Iverson et al. 2004b).

Various aspects of forest management will therefore become important to assist migration or encourage an increased rate of migration. Two primary modes include increasing the connectivity of forested land (i.e., to provide migration corridors) and assisting in the migration (e.g., by artificial distribution of seeds and other propagules). In addition to facilitating species movements, forest management can also play a large role in adaptation through techniques that increase the resistance of the current forest stands to environmental and other stresses, thereby increasing their resilience.

2.4.3 Enhancing adaptation through stand and landscape manipulation

On their own, the extent to which tree populations will be able to adapt to a changing climate depends upon the amount of phenotypic and genotypic variation, the natural selection intensity, fecundity, degree of interspecific competition, and a range of biotic interactions (Aitken et al. 2008). We may be able to intervene in the latter three via silvicultural management. There have been several publications that thoroughly describe the suite of possibilities to enhance adaptation through stand- and landscape-level management (e.g., FAO 2012, Johnston 2009, Spies et al. 2010). For example, Spies et al. (2010), working in the Pacific Northwest of the United States, provided the following ideas to enhance adaptation at the stand and landscape levels:

1. To promote resilience and vigor and to promote diversity of species and stand structures, use variable-density thinning in dense young stands to provide more resources to the surviving individuals.
2. Maintain mature stands where possible, because older, well-established individuals (at least before senescence begins) are usually more resistant and resilient to disturbances and climate change.
3. Increase the proportion of the landscape devoted to providing critical habitats and resilient ecosystem types, so that any single disturbance event has a decreased probability of destroying the habitat.
4. Manage wildfire to protect habitats or species that are at risk by suppressing fire where critical habitats exist, treating stands to reduce fuel loads, increasing spatial heterogeneity to create more resilience against fire or pests, or implementing tactical treatments that create fire breaks. (However, these interventions will have trade-offs with the requirements of some species. For example, some boreal species such as jack pine (*Pinus banksiana*) require periodic high-intensity fires to ensure their persistence within a landscape; Rohde et al. 2012.)

5. Alter the landscape structure to facilitate migration of species, to impede the spread of fire and pathogens, or a combination of the two. (Again, this may lead to mutually exclusive outcomes for some species.) Here, it is helpful to identify "pinch points" where species movement is constrained by the landscape, so that managers can alter the landscape structure accordingly and most efficiently. Tools to assist in this landscape analysis include Conefor Sensinode (Saura and Rubio 2010) and Circuitscape (McRae and Shah 2011).

2.4.4 Enhancing adaptation through managed relocation

Spies et al. (2010) proposed an additional idea to encourage adaptation to climate change: "Establish new genotypes and species to create communities that are adapted to current and future climates"—in other words, to assist species migration, which is also referred to as "managed relocation" or "assisted colonization". Here, we will use the definition presented by Hoegh-Guldberg et al. (2008): *intentionally moving species to sites where they do not occur or have not been known to occur in recent history*. The use of assisted migration has elicited controversy within conservation circles and must be used with caution because of potentially serious trade-offs (Hoegh-Guldberg et al. 2008, Richardson et al. 2009). Opponents cite many cases in which intentional relocations resulted in myriad environmental issues (Davidson and Simkanin 2008, Ricciardi and Simberloff 2009, Seddon et al. 2009) because of unanticipated risks, such as runaway invasions, that surface only after it is too late to turn back. Proponents point out that assisted migration is a key option that must remain available in the face of unprecedented global change (Minteer and Collins 2010, Sax et al. 2009, Schwartz et al. 2009, Vitt et al. 2010). Several groups have developed frameworks to evaluate the risks and benefits of assisted migration so that decisionmakers have solid approaches they can use (Hoegh-Guldberg et al. 2008, Lawler and Olden 2011, Richardson et al. 2009, Seddon 2010).

When the discussion shifts to common trees, rather than endangered species such as *Torreya taxifolia* (Schwartz 2005), the discussion changes. Trees have been planted in places where they previously did not occur for centuries. In the context of commercial forestry operations, managed relocation has been proposed as a means to maintain forest productivity, health, and ecosystem services under a rapidly changing climate (Gray et al. 2011, Kreyling et al. 2011). Pedlar et al. (2012) thus distinguish forestry-assisted migration from rescue-assisted migration (the latter being the context of much of the debate) based on the intended outcomes, target species, movement logistics, potential risks, science-based feasibility, scope, cost, and practice. We believe that if practiced cautiously and with the focus on moving species within their current broadly defined range to encourage "filling in" at the margins where a species is less common, forestry-assisted migration holds promise as a relatively low-risk tool for adaptation to climate change.

2.5 Putting it all together: a case study of potential forest responses to climate change in the eastern United States

In this section, we present a case study that attempts to capture the key aspects of the discussion thus far in this chapter—specifically, we describe the results of our last 17 years of research, which has been devoted to understanding and modeling potential changes in forests of the eastern United States in the face of a changing climate. Though details on these efforts have been published, we will present a brief synopsis accompanied by links to representative papers for readers who want to learn more. Figure 2.3 provides a flow chart of the overall process.

We have used a series of species-distribution models to assess habitat suitability for 134 tree species across the eastern United States, under both current environmental conditions and predicted future conditions. The methods used in these models, which were created with DISTRIB, have been published (e.g., Iverson et al. 2008a, 2008b, 2011; Prasad et al. 2007, 2009). In summary, the procedure is as follows: (1) collect data on the forests using more than 100 000 forest inventory and analysis plots (Miles et al. 2001) and data on 38 predictors, including soil, climate, and landscape variables; (2) aggregate all data to a 20 × 20 km grid across the eastern United States, including estimates of the importance of a species based on the numbers and sizes of individuals of the 134 tree species; (3) use a decision-tree ensemble method of statistical modeling (including regression-tree analysis, bagging, and random forests) to establish contemporary relationships between the 38 predictor variables and the importance values determined in step (2), and then use the model to create

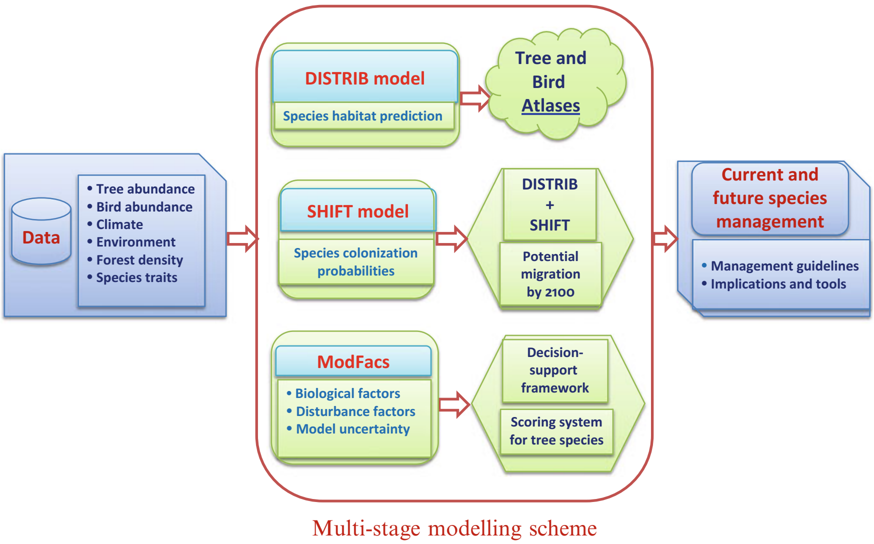

Figure 2.3 Flow diagram for the approach used to model potential responses of forests and tree species to climate change

a wall-to-wall map of importance values that resembles the current situation (Prasad et al. 2006); (4) use a series of future climate scenarios, varying according to greenhouse-gas emission scenarios and general circulation models, to replace current values of the seven key climate variables used by the models with their potential values at each of the three future time steps; and (5) map, chart, and tabulate the outputs. The outputs, consisting of more than 20 maps, 11 charts, and 4 tables for each of 134 tree species, are available in our Climate Change Tree Atlas (http://www.nrs.fs.fed.us/atlas/tree/tree_atlas.html). The atlas provides a suite of landscape-scale ecological information for each species under both current and future conditions, including details on the current species–environment relationships, maps of species abundance, life-history information, relative importance of the 38 predictors, potential habitat changes according to three general circulation models and two emission scenarios, and tables of potential changes by ecoregion, state, national park, or national forest (Prasad et al. 2009).

One feature of advanced data mining and modeling procedures such as those used in this analysis is that some distinction of scale can be made for key drivers of the model through the model outputs. For example, in our model for white ash (*Fraxinus americana*), we used a regression-tree tool, “random forest” (Prasad et al. 2006), to show that at the distribution level (i.e., the range of the species), climate variables such as the January temperature were most relevant for this species, whereas soil permeability was the single most important variable for identifying the most suitable habitat within white ash’s distribution. For most species, though importantly, not for all species, we can discern the scale of influence for each driver and distinguish differences among drivers by means of the regression-tree analyses. For example, we often see a distinction between climate-level versus landscape-level drivers, such that initial, broad-scale variables (often climate) fall out at the top of the regression tree, whereas fine-scale (often edaphic) variables fall out farther down in the tree’s structure (Iverson et al. 2011). Thus, these tools provide additional detail about the workings of the models and insights into why species occur where they do.

We then used the SHIFT model (Fig. 2.3) in conjunction with the outputs of DISTRIB to model the possible colonization of new suitable habitats within the next 100 years (Iverson et al. 1999, 2004a, 2004b; Prasad et al. 2013; Schwartz et al. 2001). SHIFT is a spatially explicit simulation model based on 1×1 km cells that simulates the dispersal of individual species propagules as a function of the current abundance of suitable habitats in surrounding cells, the proportion of the land covered by forest in the region to which the species is migrating, and the probability of long-distance dispersal using an inverse-power function of distance (so that long-distance dispersal also occurs occasionally). The rate of dispersal was calibrated to approximately 50 km per century through unfragmented areas of forest, which is towards the high end of the Holocene migration rates. Even so, the “advancing front” of the migrating species is likely to be concentrated near the boundary of the current distribution of the species and is not likely to keep pace with projected rates of warming and changes in habitat availability (Iverson et al. 2004a).

Another important interpretation of these SHIFT outputs is that the source strength (i.e., the abundance of a species near the boundary of its distribution) appears to be more important for migration than the sink strength (the proportion of forest cover in the destination cells). The combination of SHIFT with DISTRIB therefore predicts how much of the newly suitable habitat may be colonized over a 100-year period (in the absence of human-assisted migration); typically, this is only a small fraction of the available habitat.

Because of scale issues, it is difficult to translate the potential climate effects on the model into specific management activities for forest stands. With DISTRIB at a cell size of 20 × 20 km, we believe that multiples of at least 20 cells should be used for interpretation of regional trends, such as developing lists of species that are likely to increase or decrease their distribution. At a local management scale, managers must consider potential species shifts as only one of several inputs when they plan and implement management actions (Swanston et al. 2011). SHIFT, despite its finer 1 × 1 km cell size, presents a probability map in which general patterns (not specific single-cell probabilities) emerge within the larger landscape. The local factors of soils, topography, past silvicultural treatments, and the current species composition and forest structure remain the primary factors to consider in management, but overlaid on that picture is the potential for the distribution of certain species to decrease and that of certain other species to increase over time. Therefore, management can potentially provide refugia for declining species and new habitat for expanding species or even new migrant species through assisted migration in the forestry context (Pedlar et al. 2012).

Modeling the responses of a comprehensive suite of biological and disturbance characteristics that interact in myriad ways is extremely difficult—irrespective of whether statistical or process-based mechanistic models are used. We therefore developed a way to use modification factors to improve predictions. The ModFacs system (Fig. 2.3), a nonspatial scoring system, uses life-history traits obtained from a literature review (12 disturbance factors and 9 biological factors) and three post-modeling assessments as a method to increase the usefulness and practicality of the model for managers and researchers (Matthews et al. 2011). The biological characteristics attempt to assess the capacity of a species to adapt to predicted future conditions, such as a higher capacity to regenerate after a fire, to regenerate vegetatively, or to disperse; these are all positively associated with the adaptability of a species in response to expected climate change. Similarly, the disturbance characteristics assess the resilience of a species in terms of its capacity to withstand disturbances (e.g., drought, fire, floods), many of which are likely to increase in frequency or severity. To score each characteristic for each species, we reviewed the key literature to arrive at a modification factor score ranging from −3 to +3 (respectively, very negative to very positive influences in the context of expected climate change and the associated disturbance impacts). We also scored each of the characteristics in terms of their relevance in the context of the future climate (i.e., whether the changing climate will potentially increase the risk of this disturbance), with scores ranging from 1 to 4 in order of increasing relevance, and in terms of their uncertainty (e.g., our confidence in the data supporting our scoring), with scores ranging from 0.5 to 1.0 in order of increasing certainty. ModFacs also provides a means to assess each species in terms of its adaptability to the impacts of climate change. We have summarized, synthesized, and validated these modification factors as best we can based on the available information, and the overall information is then passed through management filters that adjust the results for local conditions, if necessary.

The goal is to finally arrive at appropriate information and potential tactics to support the management of a species (Fig. 2.3). Our intention is to provide the best information possible, under the uncertainty limitations imposed by the state of our knowledge, for decisionmakers to consider in their efforts to account for climate change.

2.6 Research needs

There is still plenty of research needed to better understand the relationship between climate and forests, and especially how the changing climate will affect forests.

Modeling studies have progressed a great deal in the last decade. The advent of advanced nonparametric statistical methods has greatly benefited the modeling of species distributions (Elith et al. 2006, Franklin 2009). Mechanistic modeling has also come a long way (Ravenscroft et al. 2010, Tague and Band 2004). Each approach brings its own advantages and drawbacks, and when both approaches arrive at similar answers, confidence in the predictions increases; where the approaches predict different outcomes, focused research may uncover the reasons for the discrepancy and allow improvement of the models (Morin and Thuiller 2009, Swanston et al. 2011). In addition, models that incorporate both approaches are now attempting to achieve the best of both worlds (Iverson et al. 2011). Nonetheless, there will always be trade-offs between using complex mechanistic models versus simpler empirical models to assess possible changes in species habitats (Thuiller et al. 2008). Myriad tough questions still remain to be answered (Iverson and McKenzie 2013, McMahon et al. 2011).

To improve our understanding of climate–forest relationships, much basic research must be done to understand the biological, ecological, and physiological attributes of individual species and to predict how multiple species will interact under various environmental situations.

Historically, provenance studies have assessed seed sources and genotypes. These data are being mined even now, decades later, to provide clues about the adaptability of a species under future climate change (Carter 1996). However, there is a need for competition experiments to see how seedlings will fare, for example, if their propagules travel northwards into an established forest community.

Tests of assisted migration will also be necessary to begin the process of understanding how we can help forests adapt to the new conditions created by climate change.

2.7 Conclusions

Climate change may be a more insidious agent of change than fire or anthropogenic land-use change, but it affects all forests in certain ways. Climate change will also interact with various factors and modify outcomes in unique ways, such as by increasing the frequency and severity of extreme climatic events or disturbance events, whether directly or indirectly.

Humans are largely responsible for modern climate change and must therefore decide whether and how to reduce carbon emissions to mitigate the coming changes. Humans must also decide to improve our understanding of forests and other ecosystems, including human-dominated ecosystems, and, where practical and scientifically prudent, help them adapt to the changing conditions. Part of this effort can be to simply promote healthy ecosystems via sound management. Artificially moving species also may become more and more part of the equation.

Climate is an important agent of change for forests. As the climate changes, so do the forests. In light of the increased stressors that are currently being observed, it is up to us to manage our forests in ways that will best suit the needs of a rising human population and the needs of our forests.

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Chapter 3
Wildfires and landscape dynamics in Portugal: a regional assessment and global implications

Francisco C. Rego and Joaquim S. Silva

Abstract Wildfire is an important and complex factor that both shapes landscapes and is shaped by landscapes. In this chapter, we discuss some of the factors that have shaped wildfire frequency and size in Portugal from a landscape perspective and describe the expected changes that will result from a combination of the predicted future climate change and socioeconomic changes such as the abandonment of agricultural land. Some landscapes, such as shrublands, are more vulnerable to fire than others, and the frequency and size of wildfires depend in complex ways on the proximity to humans, who provide both the major source of fire ignition (humans are responsible for more than 95 % of all wildfires in Portugal) and the major agent for fire suppression. Based on the results of our analysis in Portugal, we propose some generalizations that are likely to apply to other regions around the world, such as the need to manage and coexist with fire rather than adopting a strategy based exclusively on fire suppression. This will become particularly important in the context of global climate change, which is expected to increase wildfire frequency.

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3.1 Introduction

Portugal is part of the Iberian Peninsula and extends over approximately 89 000 km^2. It is bounded on the west and south by the Atlantic Ocean and on the north and east by Spain but has a Mediterranean climate with several localized variants according to the Köppen classification. There is a wide climatic variation across the country, with average annual precipitation ranging between 500 and 1500 mm. Portugal has an average population density of 121 residents km^{-2}, but in rural districts this decreases to 15 residents km^{-2} (Nunes 2012). Portugal is mainly covered by forests (around 33 % of the total area), followed by agricultural land (around 30 %) and shrublands (around 25 %). More than 80 % of the forest area is dominated by only four species: maritime pine (*Pinus pinaster*), eucalyptus (*Eucalyptus globulus*), cork oak (*Quercus suber*), and holm oak (*Quercus rotundifolia*) (Marques et al. 2011). The shrublands are mostly dominated by species in the Ericaceae, Cistaceae, and Fabaceae.

More than half a million forest fires were registered in official databases from 1980 to 2009. During this period, fires burned around 3.2 million ha, which amounts to around one-third of mainland Portugal. In recent decades, there has been an increase in the number of ignitions, reaching an average of 26 000 per year from 2000 to 2009 (Nunes 2012). According to remotely sensed data covering the period from 1975 to 2007, the area that burned in a single year (including all fires ≥5 ha) ranged from 15 500 ha in 1977 to 440 000 ha in 2003. The largest fire occurred in 2003, extending over about 58 000 ha (Marques et al. 2011). These figures show a very high incidence of fire in Portugal, even in comparison with other areas that have similar climatic and landscape characteristics within the Mediterranean region (San-Miguel and Camia 2009). For this reason, it is not surprising that there have been many studies of the interaction between wildfires and the landscape in Portugal.

Landscape ecology can provide valuable insights into this interaction by examining the relationships between fire, the ecosystems that sustain it, and human activities at different scales. In this chapter, we will characterize and discuss these relationships based on studies from Portugal. This regional perspective of the interaction between fire and landscapes includes a review of our present knowledge of wildfires and a prediction of future trends based on likely scenarios for human activities and climate. In the final sections, we extend our discussion to a more general basis, focusing on the main problems we have discussed and suggesting possible solutions and implications.

3.2 Landscapes and fire ignition

Wildfires are often ignited by a point source, whose location at the time of ignition is commonly not precisely known. However, despite the difficulty of predicting the time and location of this ignition, it is widely recognized that managers must understand the spatial and temporal patterns of fire ignition, since this is an essential element in analyzing and assessing wildfire danger (e.g., Finney 2005).

The primary causes of ignition vary in different parts of the world. Although lightning is the primary cause of fire in several regions (e.g., Rorig and Ferguson 1999), such as the world's boreal forests, most contemporary wildfires are of human origin in other regions, including the most populated areas. Because of the variability among different regions of the world, different studies have reached different conclusions about the primary factors that influence the spatial patterns of fire ignition (e.g., Badia-Perpinyã and Pallares-Barbera 2006, Cardille et al. 2001, Yang et al. 2007).

However, in the Iberian Peninsula, official statistics for both Portugal and Spain indicate that around 97 % of all investigated wildfires were human-caused (DGRF 2006, MMA 2007). Accordingly, most studies in this region have concluded that human-related factors are the most important factors that determine the spatial and temporal patterns of ignition (Catry et al. 2007, Romero-Calcerrada et al. 2008, Vasconcelos et al. 2001, Vega-Garcia et al. 1996).

In Portugal, the number of fire ignitions is high in comparison with other European countries with a similar population density (San-Miguel and Camia 2009). Using a database of fire ignitions, Catry et al. (2009) were able to illustrate the importance of various factors associated with human presence and activity in predicting spatial patterns of fire ignition. They evaluated the importance of population density, proximity to roads, land use, and elevation by comparing the locations of more than 127 000 ignitions between 2001 and 2005 to a random selection of points throughout the country. The comparison of the frequency of the ignition for each class of the factors that they analyzed with a purely random distribution revealed the main factors involved in determining the spatial pattern of ignitions (Fig. 3.1).

The authors concluded that human activities were the primary cause of wildfires because about 60 % of the ignitions were observed in areas with a population density greater than 100 persons km^{-2} (Fig. 3.1a). In addition, around 60 % of the ignitions occurred within 500 m of the nearest road (Fig. 3.1b). Different land uses were also found to be associated with different levels of ignition (Fig. 3.1c). Approximately 25 % of all ignitions occurred in the area classified as interspersed urban–rural. About 60 % of the wildfires started in agricultural areas, possibly because of the traditional practice of burning to eliminate agricultural residues. Elevation was also considered, since the authors hypothesized that burning for the renovation of mountain pastures to improve conditions for livestock and lightning-caused ignitions would both be more common at higher elevations (e.g., Vazquez and Moreno 1998), but this factor was not as significant as the authors expected; the observed ignition frequencies were not significantly different from a random distribution (Fig. 3.1d).

Analyses of the factors associated with ignition sources make it possible to develop predictive spatial models. Most studies have used logistic regression models (Catry et al. 2009, Preisler et al. 2004, Vega-Garcia et al. 1995), but other authors used different approaches, such as artificial neural networks (Chuvieco et al. 2003,

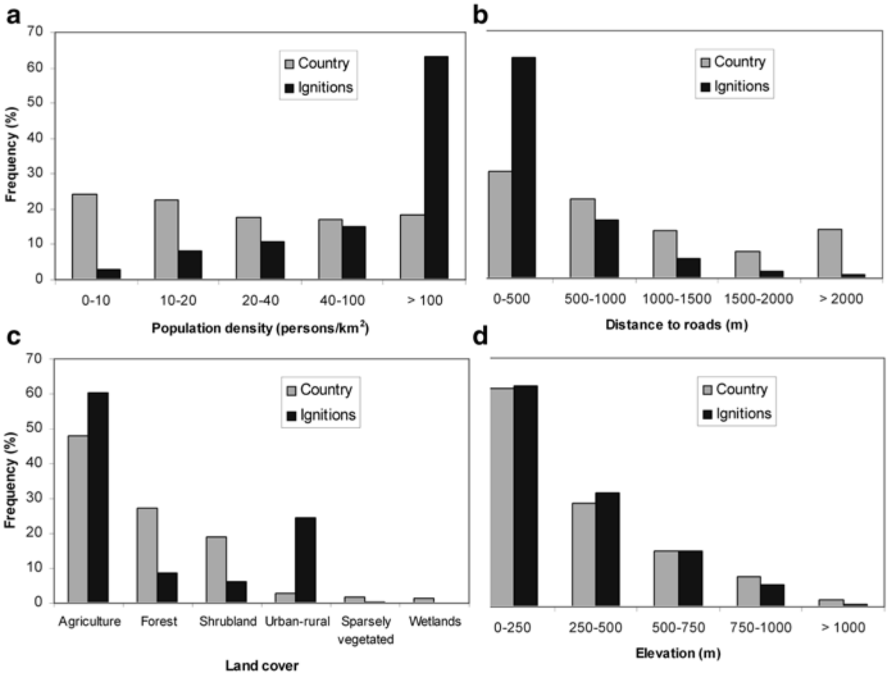

Figure 3.1 Comparison of observed ignition frequencies to frequencies expected under a random distribution for the country as a whole as a function of different variables: (**a**) population density, (**b**) distance to the nearest road, (**c**) land-cover type, and (**d**) elevation (Catry et al. 2009). "Country" represents the results for a randomized sample throughout the country; "ignitions" represents the actual recorded fires

Vasconcelos et al. 2001) or classification and regression-tree algorithms (Carreiras and Pereira 2006). In many of these models, human-related variables (e.g., population density) were included. Where land-use and cover type characteristics were included, most models in southern Europe (e.g., Badia-Perpinyã and Pallares-Barbera 2006) and in some other regions of the world (e.g., Cardille and Ventura 2001) indicated that fires, independently of the resulting fire size, were much more likely to start in non-forested areas than within forests, even though forests provide an environment that promotes the spread of fires.

Wildfire ignitions result in burned areas of different sizes. The geographical distribution of ignitions that resulted in large fires differed from the distribution of ignitions for all fire sizes combined, as can be observed in data from Portugal (Fig. 3.2). To explore the size-dependent pattern of fire ignitions in Portugal, Moreira et al. (2010) assigned each fire to one of several size classes: 5, 50, 100, 250, and 500 ha. They then modeled the probability of an ignition resulting in a different burned area by means of logistic regressions using the three main explanatory variables previously used by Catry et al. (2009): population density, distance to the nearest road, and land use. They then compared the coefficients of the variables across the models for the different size classes (Fig. 3.3).

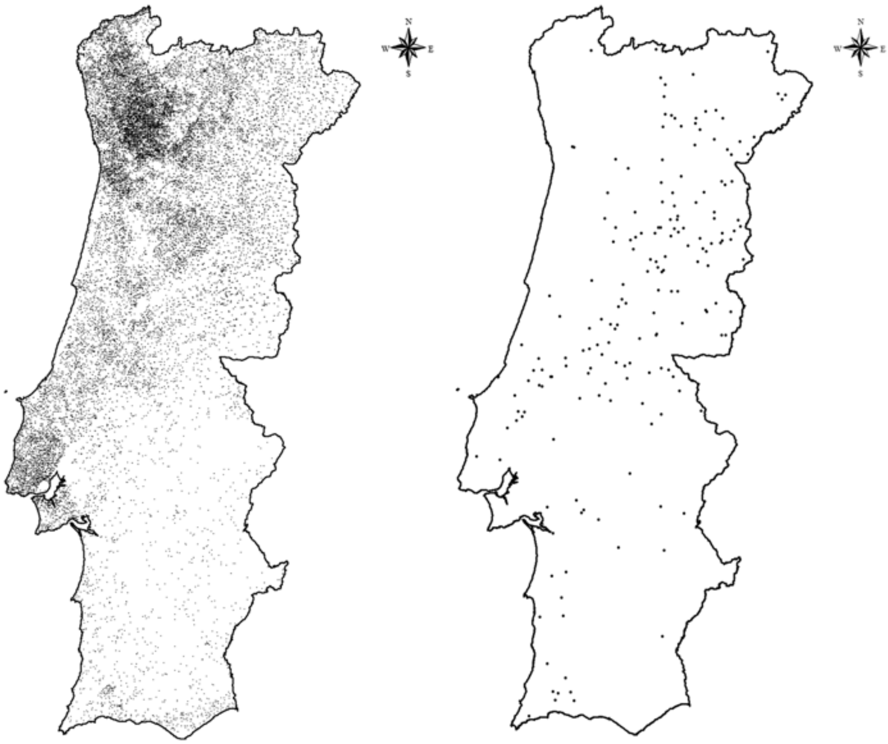

Figure 3.2 Geographical distribution of (*left*) all fire ignitions in Portugal between 2001 and 2003 and (*right*) ignitions that resulted in burned areas greater than 500 ha (Moreira et al. 2010)

The regression coefficients for population density were always negative, and were more negative for larger burned areas, indicating that once ignition occurs, the likelihood of a large fire decreases with increasing population density. For distance to the nearest road, the coefficient was positive for medium-sized fires (5 to 250 ha), indicating that fires of moderate size are more likely to occur farther from roads, whereas the coefficients for the largest fires were not significant, indicating that distance was not a significant factor for the largest fires. In contrast, land-use and cover type seemed to be important for some types (i.e., had a larger coefficient) but not for others in terms of increasing the likelihood of larger fires. The coefficients for shrublands and forests became increasingly positive with increasing fire size, particularly when compared with agriculture, suggesting that the transition to a larger fire is increasingly easy in the former land-cover types (Moreira et al. 2010).

From these previous studies, we can conclude that the same factors (population density, distance to the nearest road, and land cover) are responsible for the patterns of fire ignition and the final burned area but that their effects are quite different.

Population density was positively correlated with the number of ignitions in many studies (e.g., Cardille et al. 2001, Catry et al. 2009, Mercer and Prestemon 2005, Yang et al. 2007) but was simultaneously negatively associated with the

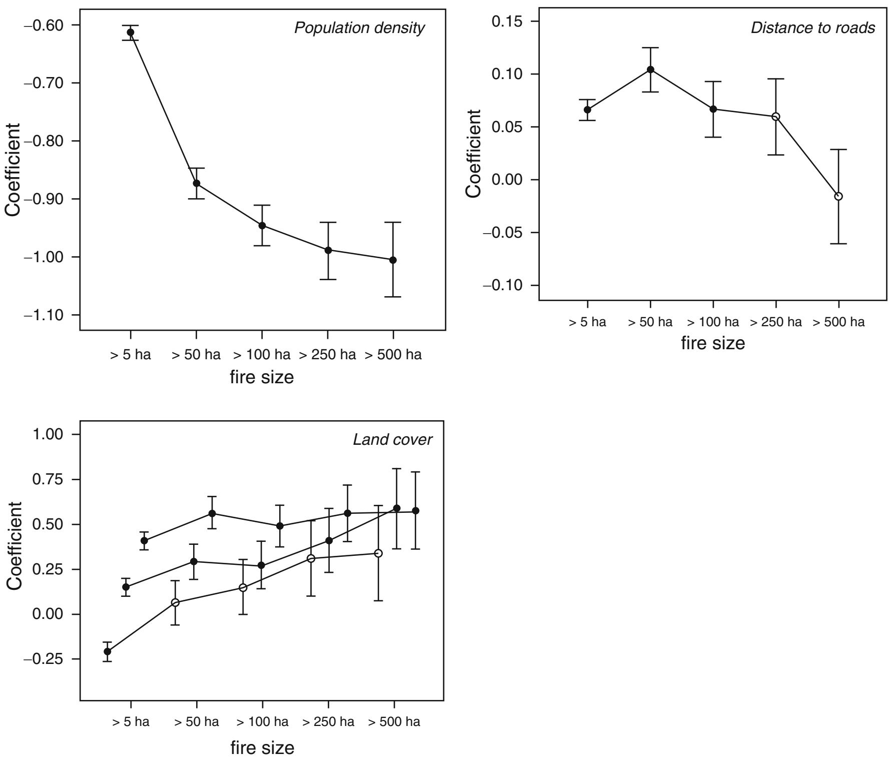

Figure 3.3 Regression coefficients (mean ± SE) for fire size as a function of population density, distance to the nearest road, and land-cover type in different logistic regression models that expressed the likelihood of an ignition resulting in a burned area of a given size (Moreira et al. 2010). Land cover is a categorical variable, with the reference category being agricultural land. The *upper*, *middle*, and *bottom lines* represent, respectively, shrublands, forests, and interspersed urban–rural areas. *White dots* indicate nonsignificant regression coefficients

burned area, indicating that population density plays a dual role in defining fire patterns: simultaneously, it represents a source of ignition and a higher likelihood of controlling the size of the burned area, probably as a consequence of earlier detection and more effective suppression (Moreira et al. 2010).

Proximity to roads was clearly associated with ignition probability in several studies (e.g., Catry et al. 2009, Romero-Calcerrada et al. 2008, Vega-Garcia et al. 1996), and it seems that medium-sized fires were more likely to occur farther from roads, possibly because of more difficult detection and a greater distance that suppression crews must travel from the road. However, larger fires seem to develop independently of the distance to the nearest road (Moreira et al. 2010).

Land cover is known from various studies around the world to be an important factor that determines fire ignition (e.g., Cardille and Ventura 2001, Yang et al. 2007), and the study by Catry et al. (2009) in Portugal confirmed these findings: they concluded that the vast majority of ignitions were concentrated in agricultural areas and in interspersed urban–rural areas. Nevertheless, subsequent studies indicated that very large fires are much more likely to spread in forests and shrublands, probably because these are areas with low population density, which delays detection, and with higher and more continuous fuel accumulation, which makes fire fighting more difficult (Moreira et al. 2010).

3.3 Landscape types and fire probability

Several papers have addressed the relationships between landscape types and fire spread. For example, Nunes et al. (2005) studied the patterns of fire spread in Portugal during the 1991 fire season and found the highest probability of fire in shrublands, followed by forests. In Sardinia, a nearby Mediterranean region, Bajocco and Ricotta (2008) also found that fires burned the landscape selectively, following a pattern similar to that in Portugal.

In another study to estimate the fire probability as a function of the land-use or cover type, Moreira et al. (2009) used data from 5591 fires that burned in Portugal between 1990 and 1994 to compare the land-use and cover type composition before the fire in a buffer surrounding (and including) each burned patch (land-cover availability) with the composition within the patch (land-cover use). If a given land-use or cover type burned more or less often than the relative abundance of that type within the regional landscape, different land-use or cover type compositions would be expected to appear within the burned patch and in the buffer surrounding the patch, and fire would therefore be considered to be selective.

This approach used selection ratios to characterize the patterns of land-use or cover type selection by fire (as in Moreira et al. 2001) and was analogous to studies of habitat preferences by animals (Manly et al. 1993). The selection ratio for a given land-use or cover type was estimated as the ratio of the proportion of that type in the burned patches to the proportion of that type in the surrounding landscape. The results of this study at a national scale indicated that annual crops, permanent crops, and agroforestry systems were the least likely to burn, with fires occurring at less than half of the rate expected based on their proportions of the landscape (Fig. 3.4). Shrublands were clearly most at risk of fire and burned twice as often as expected. Forests as a whole showed intermediate behavior, with some variation among coniferous, eucalyptus, broadleaved, and mixed forests.

Agricultural crops are recognized by various authors as being the least fire-prone cover type, possibly because of the lower fuel loads and the generally higher moisture contents (e.g., Sebastián-López et al. 2008), but also because cultivated land is usually closer to houses, making fire detection faster and firefighting both easier and a high priority (Moreira et al. 2009).

Shrublands are clearly the most fire-prone land-cover type in Portugal (Marques et al. 2011, Nunes et al. 2005), which agrees with similar findings in other parts of the Mediterranean region (González and Pukkala 2007, Wittenberg and Malkinson 2009). These results have been explained by a combination of both the special fuel

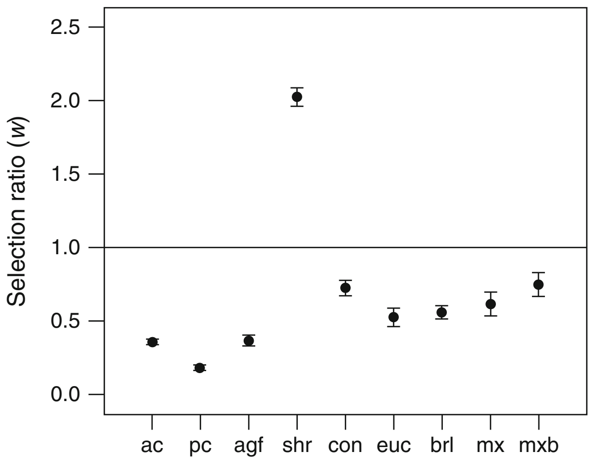

Figure 3.4 Mean selection ratios (*w*; mean ± 95 % confidence interval) for the land-cover types that burned in Portugal between 1990 and 1994. Land-cover types in this analysis were annual crops (ac), permanent crops (pc), agroforestry land (agf), shrublands (shr), coniferous forests (con), eucalyptus forests (euc), broadleaved forests (brl), mixed coniferous and eucalyptus forests (mx), and mixed forests of broadleaved and coniferous or broadleaved and eucalyptus trees (mxb). Data are from Moreira et al. (2009)

characteristics of shrublands (e.g., a dense and continuous supply of fuel, located close to the ground; high contents of flammable volatile compounds) and a potentially lower firefighting priority, as they are generally perceived to be a low-value land cover (Moreira et al. 2009).

Forests vary in their probability of fire but are generally at a level of fire selectivity intermediate between agricultural crops and shrublands. However, there is also regional variation depending on the characteristics of the different forest types. To study this specific issue in Portugal, Silva et al. (2009) used different approaches to assess fire probability and used the results to rank fire probability in the following order: greatest for maritime pine (*Pinus pinaster*) forests, followed by eucalyptus (*Eucalyptus globulus*) forests, unspecified broadleaved forests, unspecified coniferous forests, cork oak (*Quercus suber*) forests, chestnut (*Castanea* spp.) forests, holm oak (*Quercus rotundifolia*) forests, and stone pine (*Pinus pinea*) forests.

However, despite these general patterns, tree cover had an important influence on the fire probability of the different forest types. Silva et al. (2009) developed a fire probability model by means of logistic regression that related fire probability to a cumulative cover index that was computed from the vegetation cover values for the different vegetation strata. This index represented a measure of the degree of light extinction across seven forest layers, with a value of 0 representing no vegetation and 1.0 corresponding to complete shade at the soil surface. Figure 3.5 shows the results for the five main forest types.

Unspecified broadleaved forest is a diverse category, but many of these stands typically have low height, reduced dominance by trees, and high fuel continuity and

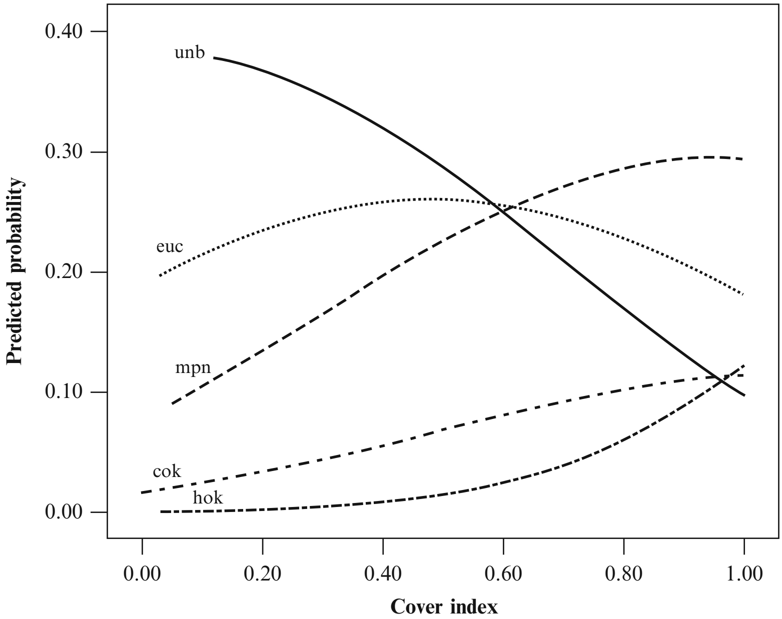

Figure 3.5 Probability of fire occurrence (from 1998 to 2005) as a function of a cumulative cover index, for which a value of 0 represents full sunlight at ground level and a value of 1 represents complete shade (Silva et al. 2009). Values are for the five main forest types in Portugal: *cok* cork oak (*Quercus suber*), *euc* eucalyptus (*Eucalyptus globulus*), *hok* holm oak (*Quercus rotundifolia*), *mpn* maritime pine (*Pinus pinaster*), and *unb* unspecified broadleaved trees

therefore resemble shrublands (Godinho-Ferreira et al. 2005). In this forest type, an increase in cover corresponds to a decrease in the probability of ignition, which suggests that as these stands age, they shift from shrubland-type plant communities (which are at high risk of fire) to closed broadleaved stands, which are much less likely to burn. Many studies have confirmed this pattern for various regions of the world (e.g., González et al. 2006, Mermoz et al. 2005, Wang 2002) and have confirmed that a lower probability of fire results from the lower flammability of the associated fuels.

Cork oak and holm oak stands have the lowest fire probability (Fig. 3.5). These stands are commonly managed using an agroforestry system named *montado*, which includes the presence of pastures and crops that maintain a low cover of scattered trees, with low fuel accumulations in the understory. As we have noted previously, agricultural land (including agroforestry systems) is less likely to burn than most other land-use types. However, when the vegetation cover increases, especially in the understory, the fire probability increases to values similar to those of unspecified broadleaved forest (Acácio et al. 2009).

Eucalyptus and maritime pine stands showed sharp increases in fire probability with increasing cover (Fig. 3.5) although the former showed a decreasing trend for very high densities. This can be explained by the fact that eucalyptus tends to have

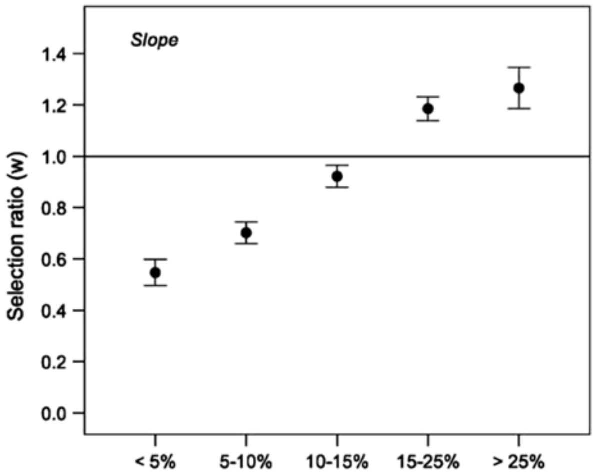

Figure 3.6 Average selection ratios (w; mean ± 95 % confidence interval) for different slope classes in northern Portugal (Carmo et al. 2011)

a high content of flammable volatile compounds, whereas there has been a general lack of management of maritime pine stands recently, resulting in increased vegetation cover due to invasion of these forests by shrubby understory vegetation that increases the probability of fire. In general, these conclusions agree with the findings of other authors, who also concluded that eucalyptus and pine stands are more flammable than other forest types (e.g., Wittenberg and Malkinson 2009, Xanthopoulos et al. 2012).

In addition to land-use and cover type, it is important to consider other landscape features that may strongly affect fire spread. One important aspect to consider is topography, which plays an important role in terms of both ignition and subsequent fire propagation. The relationship between topography and fire has been well established from experimental evidence (Rothermel 1983), but until recently, there were no approaches based on landscape analysis that examined the role of topography in fire spread in Portugal.

The existence of a relationship between topography and fire was hypothesized by Carmo et al. (2011), who analyzed the influences of land-use and cover type and topography on wildfire occurrence in northern Portugal, using the selection ratio approach to evaluate the fire probability for different topographic categories (based on slope and aspect). To do so, they characterized 1382 wildfires larger than 5 ha that occurred in 1990 and 1991. They found that a given type of vegetation in different aspect classes largely burned in proportion to its abundance within the landscape (i.e., aspect had little effect on the probability of fire). They found that the probability of a fire increased with increasing slope and that slopes steeper than 15 % were at particularly high risk of fire (Fig. 3.6). However, the problem can be more complex than this analysis suggests, as topography is often linked with land-use and cover type. For example, agricultural areas may be preferentially located in flat areas, whereas forests or shrublands may be most common in sloping land; this may

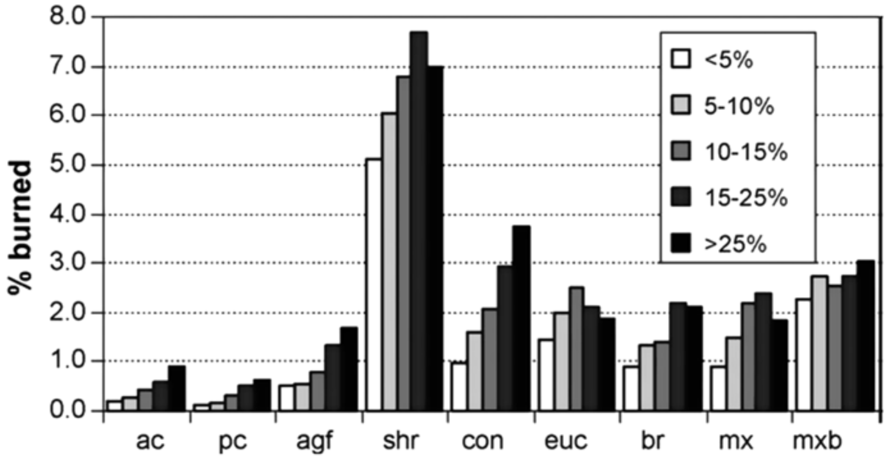

Figure 3.7 Proportion of the area of a given land-cover type that burned as a function of slope in northern Portugal (Carmo et al. 2011). Abbreviations for land-cover type: *ac* annual crops, *pc* permanent crops, *agf* agroforestry systems, *shr* shrublands, *con* coniferous forests, *euc* eucalyptus forests, *br* broadleaved forests, *mx* mixed coniferous and eucalyptus forests, *mxb* mixed broadleaved and coniferous or broadleaved and eucalyptus forests

have confounding effects on the factors that govern fire spread. It was therefore necessary to understand whether the effect of slope was independent of the land-use and cover type. Carmo et al. concluded that due to the physical effect of slope on fire behavior, the fire probability increased similarly with increasing slope for all land-use and cover types (Fig. 3.7). These results have important implications for landscape planning, since they can support the definition of landscape-scale fuel breaks. For example, areas that are prioritized for protection should include agricultural (or agroforestry) areas on shallow slopes (Carmo et al. 2011).

3.4 The dynamic interactions between landscapes and wildfires

The relationships between land-use changes and wildfires have been discussed for a long time, and general relationships have been proposed (Rego 1992). In a recent review of the interactions between landscape and wildfire in southern Europe, Moreira et al. (2011) concluded that socioeconomic factors were driving the abandonment of agricultural land and other land-use changes, contributing to more frequent and larger wildfires that promoted the development of more homogeneous landscapes covered by fire-prone shrublands; these, in turn, promoted fire spread and future fires. This trend seems to be common in many regions, particularly those with a Mediterranean climatic or cultural influence (Chuvieco 1999, Lloret et al. 2002, Loepfe et al. 2010, van Leeuwen et al. 2010, Viedma et al. 2006).

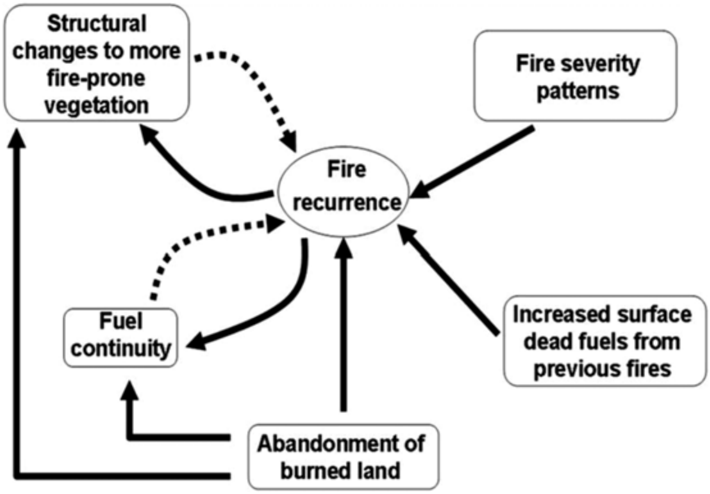

Figure 3.8 Possible feedback mechanisms that may lead to increasing fire probability in Mediterranean landscapes (Moreira et al. 2011)

This feedback mechanism is enhanced by the fact that there is a general public perception that the shrub-dominated landscapes created by wildfires are less valuable than other landscapes. This, in turn, makes them more prone to abandonment, leading to a higher probability of fire that can further promote the development of shrubland (Espelta et al. 2008, Vazquez and Moreno 2001), as shown in Figure 3.8.

The implications of wildfires for landscape dynamics can be assessed at various scales. In a detailed study at a local level in Bragança (in northeastern Portugal), Silva et al. (2011) assessed landscape changes from 1990 to 2005. The role of fire in these land-cover dynamics was assessed by building separate transition matrices for burned and unburned areas (Fig. 3.9). This example confirmed the results of other studies, which showed that fire is associated with a higher persistence of shrublands, simultaneously reducing the area of vegetation types that are vulnerable to fire and favoring transitions of all other land-use and cover classes into shrublands. It is particularly important to note the dramatic differences between the transitions for unburned and burned areas in coniferous and mixed forests: fire converted these forest types almost completely into shrublands, possibly because the persistence of maritime pine depends on a seed bank in serotinous cones, and this seed bank might not develop if the stand burns before reaching reproductive maturity (Fernandes and Rigolot 2007). In all cases, fire seems to have caused a decrease in the transitions to agricultural uses, which can be interpreted as a two-way association between fire and land abandonment: agricultural abandonment increases the probability of fire, and fire increases the probability of agricultural abandonment.

Silva et al. (2011) simulated future landscapes using transition matrices in a Markov-chain analysis to project the future landscape composition. These projections

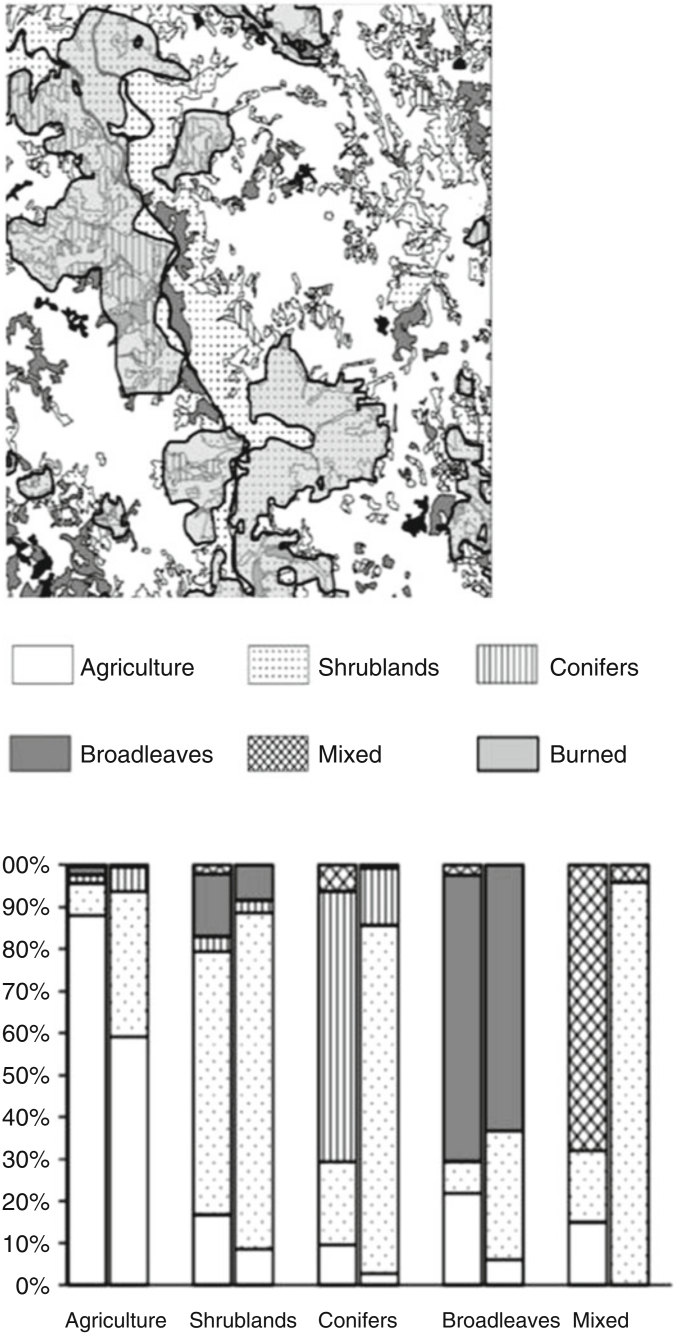

Figure 3.9 (*Top*) Map of the Bragança study area in Portugal (11 500 ha), showing the mosaic of land uses and cover types in 1990 and the areas burned from 1990 until 2005. (*Bottom*) Graphical representation of the transition matrices for unburned areas (*first bar*) and burned areas (*second bar*). Within each bar, the different proportions represent the proportion of the initial land cover in 1990 (classes on the horizontal axis) that transitioned to a different land cover in 2005 (patterns inside each bar), as shown by Silva et al. (2011)

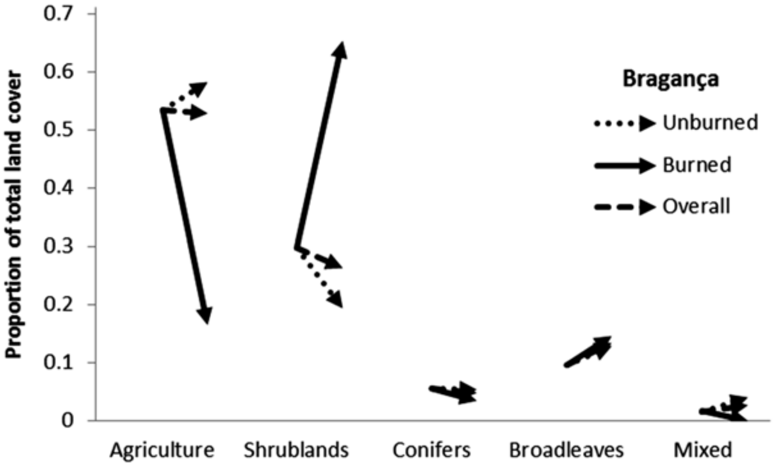

Figure 3.10 Long-term projected trends (from 2005 to 2095) based on three fire scenarios: Overall, the current fire regime; Unburned, a regime without fires; and Burned, a regime in which the whole area burned completely in each 15-year period (Silva et al. 2011). The percentage of each land cover in 2005 is represented by the start of each *arrow*. The percentage of each land cover in 2095 is represented by the tip of each *arrow*

allowed us to assess the effects of wildfires on landscape dynamics and predict the expected landscape pattern if the modeled land-use and cover type transitions were maintained. Projections were compared based on the current (Overall) fire regime with projections based on a regime without fires (Unburned) and a regime in which the whole area burned completely in each 15-year period (Burned). Figure 3.10 shows the results. Based on the overall transition matrix, the projected landscape would still be dominated by agriculture in the Overall scenario, with a decrease in shrublands and an increase in broadleaved and mixed forests. Similarly, the Unburned scenario would create a landscape with more agriculture, less shrubland, and an increase in the proportions of broadleaved and mixed forests. In contrast, fire-driven transition dynamics (the Burned scenario) would create a landscape strongly dominated by shrubland, with greatly reduced areas of agriculture but with an increased broadleaved forest component at the expense of coniferous and mixed forests (Silva et al. 2011).

3.5 A broader perspective on relationships between fires and the landscape: main problems and proposed solutions

Similarly to the situation in Portugal, most regions of the world include landscapes where fire is an important element of change (Pausas and Keeley 2009). As a consequence, there has been considerable effort to study and understand the relationships between fire and landscapes around the world. Based on these studies, it has been widely argued that plant species have evolved and adapted to natural fire regimes and that landscapes may reach a *quasi* equilibrium under any given fire regime (e.g., Pausas and Keeley 2009, Vogl 1982). However, human activities have modified fire regimes for millennia in many parts of the world. Alterations in land use, fire use, fuel patterns, and human-caused ignitions, as well as recent efforts in wildfire suppression, have all strongly influenced fire regimes at a range of scales (Bowman et al. 2009, Marlon et al. 2008, Pausas and Fernández-Muñoz 2012). Furthermore, observed and anticipated shifts in climate and weather patterns are expected to cause further alterations in fire regimes at many scales—from continental to regional or even local (e.g., Moritz et al. 2012, Raymond and McKenzie 2012).

As our discussion of the Portuguese situation showed, studies of relationships between fires and the landscape must address the spatial distribution of wildfires. Wildfires do not ignite and spread randomly across the landscape (e.g., Mermoz et al. 2005, Pezzatti et al. 2009, Verdú et al. 2012). Understanding the nonuniform distribution of wildfires is essential to understand why different regions of the world and even different parts of a region with the same climate may have completely different fire regimes. In the case of wildfire ignition, we can start by distinguishing regions where fire is still mostly a natural phenomenon from regions where it is mainly human-caused. In the latter case, the distributions of human activities within the landscape and of the associated human infrastructures are crucial aspects that define the distribution of fire ignitions across a study region. Moreover, wildfire databases show that the size of wildfires follows a markedly skewed distribution (Li et al. 1999, Pausas and Fernández-Muñoz 2012), with a few large burned areas that account for most of the total area that is burned annually. This leads to the problem of learning how fires spread across a landscape. This problem, which can be described as fire selectivity (see Sect. 3), can be tackled at different levels, depending on the study's scale and the problems for which answers are needed. An assessment of fire selectivity for coarse land-use and cover classes is suitable for a regional analysis of fire probability. However, at a more local scale, we might instead be interested in knowing the differences in fire probability for subtypes of the main landscape categories. This is clearly the case for forests, which have the highest probability of fire in some regions (e.g., Bajocco and Ricotta 2008, Cumming 2001). If we can understand which forest types burn most often, we can use this information to drive or at least influence management decisions that may determine the future fire regime.

Although the characteristics of fires are strongly determined by the characteristics of the landscape, fires may also change certain characteristics of the landscape, making the landscape more or less likely to support new fires; that is, feedbacks may occur. Under the influence of natural fire regimes, this dynamic interaction assumes different characteristics (e.g., exhibits different fire frequencies) in different ecosystems, eventually leading to dynamic steady states, such as those that develop in predator–prey relationships (Bond and Keeley 2005). However, these processes may not be balanced under the influence of human impacts, which may lead to drastic changes in the fire regime and therefore in the resulting landscape.

The results of such scenarios are not straightforward to forecast (Silva and Harrison 2010). Particularly in recent decades, there have been considerable changes in many human societies (e.g., the rapid rates of urbanization and socioeconomic development in China) that have further increased the unpredictability of relationships between fires and the landscape (Bowman et al. 2011). Hence, it is of paramount importance to assess the fate of burned landscapes and the resulting feedback on fire regimes at both regional and more local scales.

In addition to changes in human societies, we should consider the issue of global climate change. Fires are driven by climate, since climate directly affects fuel moisture content and vegetation development. The predicted climate change is therefore likely to have a strong influence on future fire regimes, on future landscapes, and on the resulting interactions (Brennan 2010, Robinson 2009). This influence adds to the unpredictability that results from social change, but most scenarios predict a higher likelihood of large wildfires due to global warming (Liu et al. 2010). In addition to the consequences of this change for landscapes, an increased occurrence of large wildfires will pose a strong threat to people and their livelihoods. Because of this problem, much effort is being devoted around the world to improve our ability to fight fires. However, the results have been discouraging in at least some regions, as the occurrence of large wildfires is increasing (Montiel and Kraus 2010). As a result, a new vision about the fire–landscape relationship has arisen, with the goal of mitigating this problem. This vision considers fire to be an intrinsic element of landscape dynamics that can be managed and not seen only as a threat. This ecological view of fire has contributed to the development of knowledge and expertise in “fire management”. Fire management has proven to be an efficient way of preventing the occurrence of large wildfires (Silva et al. 2010) and must therefore be considered an important strategy for coping with global warming (Robinson 2009).

The role of wildfires in shaping landscapes has been studied in various parts of the Mediterranean region, including Greece (Arianoutsou 2001) and France (Trabaud and Galtié 1996). Mazzoleni et al. (2004) provide an excellent review of examples of landscape change in this region, most of them related to a growing migration of rural populations away from rural areas towards coastal or heavily urbanized regions. Many regions of the world face a continuing population decline in rural areas. The so-called *rural exodus syndrome* has decreased the area of agricultural land and increased vegetation biomass over wide areas (e.g., MacDonald et al. 2000). The implications of these changes, including the possibility of an increased fire probability, have been addressed in several studies. For example, Moreira et al. (2001) estimated a 20 to 40 % increase in fuel accumulation at a landscape level in northwestern Portugal between 1958 and 1995. The combination of fuel accumulation and the current climatic trends of less rainfall and warmer summers (Santos and Miranda 2006) indicates that large wildfires will become more common in Portugal. In fact, studies both at a local scale (Moreira et al. 2011) and a global scale (e.g., Pausas and Keeley 2009) indicate that there is a clear trend in many regions for increased fuel accumulation due to changes in land use and a clear trend for more extreme weather due to climate change. The combination of these changes in fuel and weather will create favorable conditions for a higher frequency of large wildfires in many regions of the world (Robinson 2009). These predictions make it crucial for us to understand how the role of fire in the landscape could change.

An approach that has been suggested (and sometimes applied) is based on the use of fire to solve the problems caused by fire. It is well known that when land is abandoned and biomass use decreases, policies based only on fire exclusion and suppression result in large fuel accumulations. Under extreme weather conditions, this buildup of fuels may create conditions suitable for catastrophic wildfires. Several authors (e.g., Birot and Rigolot 2009, Myers 2006) have reached the conclusion that the best option is to learn how to live with fire. Therefore, the reduction of wildfire hazard and the sustainable management of ecosystems in Europe and elsewhere may require new management practices, such as "prescribed burning", which has been defined as a "controlled application of fire to vegetation in either their natural or modified states, under specified environmental conditions, which allow the fire to be confined to a predetermined area and, at the same time, to provide the intensity of heat and rate of spread, which are required to attain planned resource management objectives" (FAO 1986). Prescribed burning has been studied and developed in some European countries. The European Fire Paradox project (http://www.fireparadox.org/), which was conducted from 2006 to 2010, was essentially dedicated to studying and developing the potential of fire management through fire use (Fernandes et al. 2011, Montiel and Kraus 2010, Rego et al. 2010, Silva et al. 2010).

The effectiveness of prescribed burning has been reviewed by Fernandes and Botelho (2003), who concluded that significant reductions in the area burned by wildfires could be achieved by strategic use of this fuel management technique. Figure 3.11 illustrates the use of prescribed burning to disrupt the continuity of shrubland in Portugal. Successful use of prescribed burning has been described in several situations, but its use is often discontinued due to poor decisionmaking when managers fail to account for the benefits of this process of living with fire in a sustainable way. One of the few long-term and large-scale programs is in southwestern Australia, where a prescribed fire program has been applied successfully for decades (Burrows 2008). The future of forest landscapes and of fire seems to depend on improving our understanding of the relationships between the two and of the underlying processes. Instead of viewing fire only as an enemy to fight (wildfire), it should be viewed as a tool in vegetation and landscape management (prescribed fire). After all, we should remember the traditional Finnish proverb that "fire is a bad master but a good servant" (Fig. 3.12).

3.6 Concluding remarks

Through the studies we have described, we have tried to illustrate the relationships between fire and landscapes by providing examples from a particular area of the Mediterranean Region. This regional perspective, focused on examples from Portugal, reveals important conclusions about the role of fire in landscapes and the role of landscapes in shaping the characteristics of wildfires. We demonstrated a

Figure 3.11 Two images of the use of prescribed burning to create fuel breaks in mountain shrublands in northern Portugal (photo: P. Fernandes, University of Trás-os-Montes and Alto Douro)

close relationship between the ignition of fires and the type of landscape. Fires are more likely to start in agricultural and interspersed urban–rural areas as a consequence of human actions, and the likelihood of fire increases at higher population densities, particularly close to roads.

However, not all ignitions result in the same probability of large fires. Again, the type of landscape plays a crucial role, since large fires are more likely to occur in areas of shrubland or forest. These results agree with the results of other studies, which showed that fire spreads faster through shrublands and forests than through agricultural and agroforestry landscapes. In Portugal, different forest types present different fire probabilities, and changes in vegetation cover have different effects in different forest types. However, the complex interactions between forest composition and forest structure are not well understood, and additional research

Figure 3.12 The use of fire as a "good servant" in Lousã, Portugal (Photo by Liliana Bento, CEABN/ISA)

will be required to allow a more accurate assessment of the susceptibility of forested landscapes to fire.

The interaction between land-use and cover types and the topography is increasingly evident; for example, agriculture is most common in flat land and forests are most common on steep slopes. This is important because slope strongly affects fire spread in the different landscape types of Portugal. The research literature highlights the complex feedback mechanisms that lead to mutual influence (feedback mechanisms) between fire and the landscape. Fire seems to play an important role in the present trend of abandonment of agricultural land in Portugal, which is in turn making the landscape more vulnerable to future wildfires. This is creating feedback mechanisms that increase the probability of fire and, when combined with the predicted global warming, creates strong concern and the need for new solutions and new approaches.

These new approaches should include a change from the present paradigm, which is primarily reactive (fire fighting), to a proactive attitude based on fire prevention and management. This attitude should account for the vast knowledge acquired in recent years about the ecological role of fire in the landscape and its potential use as a management tool. The use of prescribed fire is far from being a panacea to solve fuel management problems, but it nonetheless has an immense potential that has not been fully explored. Therefore, despite the concerns being raised over trends that are leading to an increased risk of fire in areas such as Portugal, we hope that future policies will include a more comprehensive and sustainable view of the relationships between fire and the landscape.

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Chapter 4
Humans as agents of change in forest landscapes

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Abstract Forest systems play a crucial role in biogeochemical cycling and provide a variety of ecosystem services at multiple scales. Considerable progress has been made in understanding the dynamics of tropical and temperate deforestation and land-use and cover change. However, less attention has been dedicated to understanding the social and biophysical conditions under which reforestation occurs. Recent research documents the experiences of many countries that have undergone transitions from a period of high deforestation to a period of declining deforestation or even net reforestation. However, these transitions take place across a range of temporal and spatial scales. Here, we review global forest-cover trends and social processes affecting forest cover and then focus on a comparison of reforestation in

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the states of São Paulo, Brazil, and Indiana, United States. Both states have undergone extensive deforestation but now show forest restoration alongside continuing deforestation. Our focus on forest change at the state level permits a detailed examination of deforestation and reforestation dynamics and of the diverse social factors that underlie these changes. Among these factors, human values and attitudes appear most important.

4.1 Introduction

Forest systems play a crucial role in biogeochemical cycling and provide a variety of ecosystem services at multiple scales. Forested areas reduce soil erosion, are important sources of fiber and wood that are used in many economic activities, and directly support livelihoods by providing non-timber ecosystem services. Since the 1990s, considerable progress has been made in understanding the dynamics of tropical and temperate deforestation and land-use and cover change, leading to a rich understanding of the drivers of deforestation (e.g., Foster and Aber 2004, Geist and Lambin 2002, Gutman et al. 2004, Moran and Ostrom 2005, Rudel 2005). However, less attention has been dedicated to understanding the social and biophysical conditions under which reforestation occurs (e.g., Rudel and Fu 1996, Tucker and Southworth 2005). Researchers now recognize that regrowth can occur simultaneously with deforestation, but the relationship between the two is not well understood. Secondary forest regrowth and plantations can sequester significant amounts of carbon that offset at least part of the emissions from clearcutting and selective logging of forests (Nilsson and Schopfhauser 1995). The social and economic conditions under which a region transitions from deforestation to reforestation remain uncertain despite the attention paid to this topic for more than two decades (e.g., Walker 1993). The drivers of reforestation are demonstrably different from those that result in deforestation. The transition from one phase to the other represents the net impacts of a complex set of processes with connections to environmental policy and management at multiple levels of governance (Moran 2010). Such processes are affected by local, regional, and federal actors, as well as by larger-scale drivers. Interactions among actors at different levels (local, regional, federal) can produce shifts in decisionmaking that lead to a net increase in forest cover.

In several recent publications, Rudel et al. (e.g., 2005) have proposed that many countries seem to undergo a transition from a period of high deforestation to one of declining deforestation or even reforestation. These observations led to the formulation of forest transition theory (Mather 1992, Rudel 1998, Rudel et al. 2005, Walker 1993). This transition has been found in many cases, but the turnaround from deforestation to reforestation takes place across a range of temporal and spatial scales. Such a transition took place in northern Europe between 1850 and 1980, but does not appear to have happened in southern Europe. We are only beginning to understand the dynamics of social and environmental systems and the feedbacks that come into play at different stages of a forest transition or at different temporal and spatial scales. In some places, this dynamic seems to be associated with the creation of nonfarm employment that pulls farmers off the land (Mather 1992, Polanyi 1944), thereby inducing spontaneous recovery of forests in the abandoned fields. In other places, a scarcity of forest products has prompted restoration efforts by both governments and private landowners (Foster and Rosenzweig 2003), but other scenarios may also drive a transition.

The transition has important environmental consequences for carbon sequestration and biodiversity conservation (Foley et al. 2005). During the 1990s, 38 % of the world's countries experienced increases in forest cover, but the transition began at different points in their deforestation trajectories. Some countries entered the reforestation phase with 40 % of the original forest cover remaining, whereas others began at a forest cover near 0 %. The question of when a transition takes place has huge implications for the biodiversity of regenerating forests, among other factors. Rudel (2005) notes that the northern European transition in the twentieth century had different dynamics from the experience in Asia in the past 15 years, partly due to different governance approaches (e.g., command-and-control versus bottom-up). In parts of Asia, governments have responded strongly to a scarcity of forest products and increased flooding, resulting in aggressive reforestation campaigns. In China, this effort was centrally organized (Fang and Wang 2001, Zhang et al. 2000), whereas in India, village committees have increased the forest cover in a decentralized fashion (Foster and Rosenzweig 2003, Singh 2002).

Understanding the transition from deforestation to reforestation has major implications for mitigation of climate change, biodiversity conservation, stabilization of soils and of water supplies, and the availability of socially valuable recreation areas. The process affects both developed and developing countries. In this chapter, we briefly review the global trends and social processes that are affecting forest cover. We then focus on a comparison of reforestation in the states of São Paulo, Brazil, and Indiana, United States. Rudel et al. (2005) examined these processes at a national scale using data from the United Nations Food and Agriculture Organization, and it is mostly at this level that forest transition theory has been proposed and tested. Although national and global datasets offer the benefit of a broader spatial scale, they are often marred by inconsistencies in the quality of the data and by different definitions of "forest". To better understand the dynamics of changes in forest cover, we focused at a state level using two states for which high-quality data is available. This approach permits a more detailed examination of the forest-cover dynamics and of the diversity of social factors that underlie forest-cover change. Both states have undergone devastating deforestation but now show forest recovery alongside continuing deforestation.

4.2 Institutional dimensions of forest-cover change

The institutional dimensions of forest-cover change have received increasing attention in recent decades. Accumulating research indicates that institutional arrangements can permit forest destruction or promote forest conservation. Here, we define "institutions" as human-designed constraints on behavior (McGinnis 2011).

Institutions indicate what may, must, or must not be done in a given context and encompass formal and informal rules, norms, and practices (Ostrom et al. 2002). Forest management institutions exist across many levels, from local choices for forest use to municipal, regional, and national regulations and programs. Studies of institutional arrangements have revealed certain principles and features associated with successful (sustainable) forest management and conservation, but efforts to impose standardized, "one-size-fits-all" institutional arrangements have led to many failures (Ostrom et al. 2007). In many cases, successful institutional arrangements appear to evolve in situ and adapt continuously to their specific historical, political, economic, sociocultural, and environmental contexts.

4.2.1 Local rules and contexts

People who live in and around forests often depend on forest resources for their livelihoods, and this dependence can provide incentives to use the resources sustainably. Researchers have identified numerous cases in which local groups have crafted forest management institutions that foster sustainability (Banana and Gombya-Ssembajjwe 2000, Berkes and Folke 1998, Chhatre and Agrawal 2008, Gibson et al. 2000, McCay and Acheson 1987, Ostrom 1990). Contexts associated with effective local forest management include well-defined and secure tenure rights (including communal ownership), trust and shared understanding among the people who use and manage the resource, effective monitoring and enforcement, low-cost conflict mediation, the right to create and modify at least some of the rules, and recognition of the right to self-organize, among others (Agrawal 2002, Cox et al. 2010, Dietz et al. 2003, Ostrom 1990). Perhaps most important is the finding that monitoring and enforcement are strongly correlated with forest conservation under a wide range of contexts (Gibson et al. 2005, Hayes 2006, Tucker 2010, Van Laerhoven 2010). Moreover, the ratio of group size to forest size appears to matter, at least in some contexts. At low ratios, group members find it difficult to perform adequate monitoring and maintenance, whereas higher ratios can create coordination problems (Nagendra 2007, Ostrom 2005).

Local forest management regimes present diverse rules and practices and provide evidence that community-based institutions have adaptive advantages within specific circumstances (Agrawal 2007, Gibson et al. 2000, Van Laerhoven 2010). In contrast to top-down, "one-size-fits-all" programs, local regimes may permit more flexible, locally appropriate adaptations to transformative pressures. Some local arrangements prove unsuccessful, however, and changing circumstances during the past century have transformed or eliminated many community-based management regimes. Challenges that may undermine local regimes include market demands for export crops and forest products, political strife, privatization of property rights, the capture of rights by an elite, and power struggles, as well as economic and climatic shocks (e.g., Godoy et al. 2005, Henrich 1997, Schweik et al. 2003, Verhoeven 2011). Higher-level government interventions, typically imposed without regard for local institutions, also tend to undermine local rules and exacerbate deforestation (Jodha 1992, McKean and Ostrom 1995). In Brazil, deforestation appears to be significantly correlated with highway construction, cattle ranching, agricultural expansion, and programs that encourage immigration (Laurance et al. 2002, Moran 1992). Yet despite continuing deforestation in the Amazon basin, studies also reveal the presence of forest regrowth (Moran et al. 1996, 2000).

4.2.2 National regulations, programs, and top-down policies

National laws and regulations tend to encourage top-down government involvement in nonindustrial forest management. Centralized government programs, including forest concessions, settlement programs, and land grants, have become associated with extensive deforestation (Ascher 1999, Gill et al. 2009, Malingreau and Tucker 1988, Repetto and Gillis 1988). Typically, centralized approaches eliminate or severely limit local participation in forest management, which can compromise the livelihoods of forest-dependent populations and exacerbate deforestation. Forest policies and laws designed by central governments lack a nuanced understanding of local social and ecological circumstances that shape outcomes and, if the implementation process does not appropriately consider these factors, can undermine local practices and rules that would otherwise foster sustainability (Agrawal and Chhatre 2007, Cabarle et al. 1997). In countries with high institutional capacity and policy experience (e.g., European Union policies), national policies are designed with sufficient flexibility and "wiggle room" that they offer opportunities for local variability and adaptability during their implementation (Pelli et al. 2009, Winkel and Sotirov 2011).

During the late twentieth century, the inability of top-down national policies and programs to mitigate deforestation contributed to shifts in the policy tools used to encourage conservation. In the 1980s and 1990s, community-based and co-management efforts became popular, such as joint forest management in India (Behera 2009, Jha 2010, Murali et al. 2006). Some national governments (e.g., Bolivia, Guatemala, Peru, Tanzania) adopted decentralization initiatives as theoretical advances, and empirical examples indicated the potential advantages of devolving power over forests to the local level (Andersson et al. 2006, 2012; Persha and Blomley 2009). Although a desire to devolve costs to local levels evidently motivated decentralization of forest management, the rhetoric of decentralization emphasized the potential for local income generation, economic development, democratization, and increased social equity. The social and ecological outcomes have varied greatly (Larson and Soto 2008). Unilateral decentralization programs to promote forest conservation generally fail to account for the diversity of local and regional contexts. In some cases, decentralization has done little to affect genuine devolution of power from national to local governments. Even if decentralization programs do devolve power, they may be implemented ineffectively, sabotaged by special interest groups, or undermined by ineffective local politicians (Andersson et al. 2006, Ribot 1999). Nevertheless, studies of decentralization outcomes have supplied further evidence that sustainable forest management, forest regrowth, and better forest conditions are associated with monitoring and enforcement, community-based management, secure tenure, local autonomy to make and change rules, and limited state interference (Agrawal and Chhatre 2007, Persha and Blomley 2009).

4.2.3 Government programs and incentives for forest conservation and reforestation

Beyond efforts to create policies that decentralize power and allow co-management, many governments have attempted to counter forest loss by direct efforts to protect forests and to expand forest cover through reforestation. Protected areas are one of the most popular tools to conserve endangered forests. In highly developed countries, National Forest and National Park systems typically became established in unpopulated areas, and forest protection became associated with the prohibition of harvesting. This model spread around the world, but encountered resistance in the less-developed countries, where indigenous and traditional populations inhabit forests and depend on them for sustenance.

Today, protected areas have a mixed record, with troubling failures and shortcomings (Brandon et al. 1998, Curran et al. 2004, Liu et al. 2001, McKibben 2006, Terborgh 1999) that contrast with examples of successful protection (Bruner et al. 2001, Hilborn et al. 2006). Where protected areas are merely "paper parks" (i.e., they exist only on paper, with no or little management on the ground), they have been plagued by implementation and enforcement problems. In some cases, national forest laws and enforcement mechanisms have been inadequate to mitigate deforestation or have created perverse incentives that exacerbated development and land-cover change. For example, ecological degradation and deforestation rates increased after the creation of Mexico's Monarch Butterfly Biosphere Reserve (Brower et al. 2002) and China's Wolong Giant Panda Reserve (Liu et al. 2001). Degradation in the Monarch Butterfly Reserve had multiple causes; the reserve undermined community institutions by occupying communally owned land, local populations resisted the loss of their traditional use rights, and the government failed to establish adequate monitoring and enforcement, giving illegal loggers ample leeway to operate (Tucker 2004). Thus, efforts to remove forest peoples or prevent local people from using forest resources can backfire when they damage preexisting institutions and enforcement mechanisms and when new enforcement mechanisms are nonexistent or ineffective (Schwartzman et al. 2000, Tucker 2004).

In less-developed nations, reforestation and forest conservation have been well documented in parts of Brazil, Nepal, India, and Mexico (Bray et al. 2005, Ghate 2004, Nagendra et al. 2008). A number of highly developed countries, including France, Switzerland, Germany, and the United States (Davis and Jacobs 2005, Zanchi et al. 2007), are experiencing reforestation trends. Although the reasons for reforestation differ, even within a given region, global processes and changes may lead to attitudinal and behavioral changes toward forest management (Agrawal 2005).

4.2.4 International policies and programs

Global concern for forest change has spurred recent international efforts to encourage collaborative arrangements and top-down institutional frameworks for forest conservation. The United Nations' 2008 initiative on Reducing Emissions from Deforestation and Forest Degradation (REDD; http://www.un-redd.org/) and the subsequent 2010 revision to include conservation and sustainable management (REDD+) have sought to engage local forest owners and communities in developing nations to reforest and protect their forests. (Hereafter, we will refer to both programs as "REDD".) Millions of dollars have been allocated, mainly by highly developed nations, to implement REDD programs in less-developed nations (Boucher et al. 2008). As in previous efforts (e.g., Clean Development Mechanism, Global Environmental Facility), the greatest burden for adopting the new regulations and changing behavior is placed upon people living in and around the world's remaining forests, who are often poor, underprivileged, and dependent on forest resources for their survival (Blom et al. 2010, Thompson et al. 2011, Young 2010). Meanwhile, people residing in highly urbanized and industrialized countries, who consume the vast majority of global energy and natural resources, bear little responsibility for changing their behavior (Ghazoul et al. 2010). Previous top-down international programs have at times exacerbated inequity and poverty, even when the stated intention was to mitigate inequity, and REDD appears to continue this trend of overlooking local priorities and socioeconomic concerns (Rosendal and Andresen 2011). Therefore, REDD projects pose a number of risks, as well as opportunities to learn from past mistakes. Lessons learned from integrated conservation and development projects and early evidence from REDD projects indicate that success is more likely when local populations are active participants and beneficiaries (Blom et al. 2010, Oestreicher et al. 2009). In many ways, REDD constitutes an emergent system of environmental governance for which the ramifications and risks have yet to be recognized (Thompson et al. 2011).

4.3 Incentives, motivations, and household-level forest management

The globalization of markets and ideas, modernization of economies, urbanization, and industrialization affect forest cover in myriad ways. It is therefore important to consider incentives and motivations for enhancing forest conservation and increasing forest cover. Although globalization is often seen as a driver of environmental degradation, it can benefit forest cover through flows of ideas, labor, capital, and commodities (Hecht et al. 2006). For example, globalization can improve forest cover through worldwide concern over the future of tropical forests and the consequent spread of conservationist ideas, migration of people from poorer countries to economically advantaged countries, and expansion of ecotourism opportunities (Lambin and Meyfroidt 2010).

Global diffusion of environmental conservation ideas may affect individual and collective behavior toward forests (Lambin and Meyfroidt 2010). These ideas can also influence governments—public policies, creation of protected areas, and incentive and enforcement programs—and companies to develop an eco-friendly image. With increasingly urbanized populations, the perception of the value of forests as sources of ecosystem services, including esthetic and recreational values, has increased relative to the perception of forests as sources of timber and farmland, leading to changes in environmental attitudes and policies (Mather 1992). The desire to use rural land for second homes, recreation, tourism, or retirement makes forests seem more attractive.

Urbanization and industrialization also relate to reforestation through the creation of off-farm jobs and a decline in rural labor opportunities (Rudel et al. 2005). The undervaluation of rural work and livelihoods, along with the cultural and economic attractiveness of urban life, stimulate rural out-migration and land abandonment (Aide and Grau 2004, Rudel 2002). Simultaneously, adoption of more productive technologies concentrates farm production in more suitable areas. Land that is less suitable for agriculture is abandoned, allowing forests to regenerate through secondary succession. However, succession may not occur if agricultural production is integrated with regional and global markets; increasing agricultural productivity may prevent farmland abandonment if the demand for agricultural products remains high and farmers can export their products to other regions. Depending on market demand, capital availability and institutional arrangements, agricultural intensification may even lead to clearing of forested areas to increase production and profits (Lambin and Meyfroidt 2010).

Modernization of economies, openness to international markets, consumers, international non-governmental organizations, and local organizations can pressure companies and governments to adopt more environmentally sound practices. Consumer demand for green-labeled products may promote sustainable forestry practices worldwide, thus encouraging conservation and reforestation. In Brazil, for example, companies interested in exporting cellulose pulp from plantations have been pushed by external markets and by local organizations to comply with environmental legislation and adopt management practices that promote conservation of native forests (Farinaci 2012). Green labeling has been used to inform consumers around the world about a company's socio-environmental practices and can remarkably influence individual behavior (Moran 2010).

However, effective monitoring of certification labels is difficult and controversial. The current proliferation of ecolabels—with more than 300 in existence (Ecolabel

Index 2013)—makes it difficult to verify which certifications adhere to rigorous environmental standards and third-party monitoring. Corporations can invent their own private ecolabels as marketing schemes while avoiding third-party oversight (Forest Ethics 2010, Mutersbaugh 2005). More generally, monitoring remains a challenge across all levels of the commodity chain, from communities that harvest certified lumber through each link along the commodity chain in which non-certified lumber might be mixed with certified lumber. Several major organizations offer chain-of-custody certification; two of the largest (Purbawiyatna and Simula 2008) are the Forest Stewardship Council (https://ic.fsc.org/) and the Programme for Endorsement of Forest Certification (http://www.pefc.org/). However, recent efforts by certification agents and proponents to make certified goods mainstream and encourage firms to sell them have raised concerns about the vulnerability of certifications to corporate pressure (Mutersbaugh et al. 2005).

Although economic globalization can benefit forest recovery through consumer pressure and environmental discourse, it can also shift deforestation from one region to another (Lambin and Meyfroidt 2010, Meyfroidt and Lambin 2009). For instance, Mansfield et al. (2010) claimed that forest recovery catalyzed by economic growth reflects the ability of wealthy regions or countries to import forest and agricultural products and export environmental consequences. In a comparison of two biomes in Brazil, Walker (2012) concluded that forest recovery in the Atlantic Forest, where the most urbanized, populated, and industrialized Brazilian states are located, may be occurring at the expense of deforestation in the Amazon.

Globalization can benefit forest recovery through subtle mechanisms, such as changing individual attitudes, values, and choices. The dynamics of local forest management are important because they reflect the decisions of diverse and numerous forest owners. More than half of all forests in the United States are privately owned and managed by individuals, families, tribes, or the forest industry (Butler 2008, Smith et al. 2009). Management decisions are shaped by socioeconomic and ownership characteristics, market signals, policy programs, and biophysical conditions (Beach et al. 2005, Butler 2008). The practices of forest owners typically take place within defined, privately owned parcels, with limited consideration of landscape- or watershed-based impacts.

Harvests on small, family-owned lands in the United States, though episodic, are largely driven by market prices for timber, family financial needs, or the forest's health (Davis et al. 2010). Harvesting decisions, like other management choices, are among the multiple objectives landowners have for their land. These preferences often blend financial gain with an interest in the forest's non-commodity and amenity features (Best and Wayburn 2001, Knoot et al. 2010, Koontz 2001). The motivations for family ownership often focus on esthetic enjoyment, recreation, privacy, and creating a legacy for future generations (Butler 2008, Davis et al. 2010). In brief, household decisions reflect a diversity of values, attitudes, and land-use motivations (Alig 2007, Janota and Broussard 2008, Karppinen 1998).

The demographic and ownership characteristics of landowners also influence private forest management (Ross-Davis et al. 2005): age, income, education, the size of the landholding, length of property ownership, and residence location affect participation in government assistance programs, adoption of best management practices, and forest stewardship (Elmendorf 2003; Frimpong et al. 2006; Kilgore et al. 2007, 2008; Kindstrand et al. 2008; Koontz 2001). Recent private ownership changes show a growing number of nontraditional owners (e.g., younger, non-white, ex-urbanites), smaller parcels, and inadequate coordination among landowners (Best 2004, Butler 2008). Many researchers observe that landscape-wide benefits from forests may be lost as a result of increasing parcelization (i.e., division of forest tracts into multiple, smaller parcels), which typically leads to forest fragmentation and disruptions of ecological functions (Butler and Leatherberry 2004, Rickenbach et al. 2011, Vokoun et al. 2010).

Researchers further note that the sustainability of private forests demands cooperative management at multiple scales and attention to the varying spatial and temporal scales at which forests provide goods and services (Fischer and Ruseva 2010, Goldman et al. 2007, Rickenbach et al. 2011, Ruseva and Fischer 2013). Changes in forest landscapes are a function of the actions of a heterogeneous group of owners, whose individual decisions are seldom coordinated with those of others and rarely reflect the nature of forests as a public good that provides services such as clean water, air, and other amenities (MEA 2005, Ruseva and Fischer 2013). Rickenbach et al. (2011) note that "from a landscape perspective, small forest landholdings are managed in a haphazard ownership-centric way that often lacks any connection to multiscalar ecological principles". It is therefore important to closely examine the drivers and motivations that can potentially maintain forests and support forest recovery.

4.4 Findings from a household-level analysis: two reforestation case studies

Our research in São Paulo (Brazil) and Indiana (USA) aimed primarily to investigate the factors that motivated private landowners to plant trees, allow forests to recover, or conserve forest on their land, including interactions with government programs and social trends. Our work involved household surveys, interviews, and a time-series analysis of land-cover change. We paid particular attention to the institutional, socioeconomic, biophysical, and legal factors that potentially influenced management decisions. Although land use is affected by decisions at many societal levels, landscape change processes often involve individual decisions, which are influenced by social and biophysical factors and by subjective values (Moran 2010, VanWey et al. 2005). We therefore analyzed motivations and land-use preferences associated with decisions by rural landowners to protect or increase forest cover on their land. We explicitly chose to examine reforestation in the contrasting contexts of a developed and a developing country to explore whether similar or different factors influenced individual decisions during forest transitions.

Scholars have pointed out that developing countries, with Brazil as a prominent example, have not yet made the transition to increasing forest cover and that political elites continue to prevent this transition (Rudel 2005). However, research that aggregates data at a national scale, particularly for large countries such as Brazil, can miss the dynamics of change that take place at subnational scales. In fact, our initial examination of the trajectories of forest cover in São Paulo suggested similar trajectories to those in the United States—rapid deforestation then slower deforestation accompanied by gradual reforestation (Farinaci and Batistella 2012). Economic development, urbanization, and the transition in São Paulo lag a few decades behind those in Indiana, but the transition is taking place under very different political regimes, land tenure systems, and cultural and economic histories.

4.4.1 Description of the case studies

In São Paulo, we studied six municipalities (Campinas, Jundiaí, Monteiro Lobato, São José dos Campos, São Luiz do Paraitinga, and Ubatuba; Fig. 4.1) in the eastern part of the state, which has a humid tropical climate with annual average temperatures of 20 to 22 °C. The state is highly urbanized, with 96 % of the population living in urban areas, and is responsible for more than 30 % of Brazil's GDP (IBGE 2011, SEADE 2011).

In São Paulo, settlement was largely led by slave owners who operated coffee estates and expanded from the state of Minas Gerais to São Paulo (and then Paraná), a process that resulted in deforestation of half of the state during the nineteenth century. Through subsequent economic cycles involving cotton, sugar cane, coffee, and oranges, deforestation continued well into the 1960s and 1970s (Dean 1995). At the beginning of the nineteenth century, São Paulo retained 81.8 % of its forest cover, but by 1973, only 8.3 % of the forest cover remained, mostly on steeper terrain (Victor et al. 2005); since then, it has increased to about 17 % (Instituto Florestal 2010) (Fig. 4.2). Today, nearly one-third of the counties in São Paulo are experiencing some forest regrowth for various reasons (Ehlers 2007). For example, the state government has established conservation units, and reforestation is encouraged by state fiscal incentives (Hogan et al. 2000).

The dominant natural vegetation in São Paulo is the Atlantic Forest, which is both a high-priority area for biodiversity conservation (Joly et al. 1999, Myers et al. 2000) and the most urbanized and densely populated area of Brazil (Hogan 2001). In eastern São Paulo, along the Serra do Mar mountain chain, lie the most significant remnants of the Atlantic Forest, which form a mosaic of legally protected areas (Ribeiro et al. 2009). Forest inventories conducted before the 1990s reported net decreases in São Paulo's native forests. However, more recent assessments indicate increases, especially in the eastern portion of the state. Native forest in São Paulo covers an estimated 4 343 718 ha, corresponding to 17 % of the state's area (Instituto Florestal 2010). Plantations of exotic *Eucalyptus* spp. and *Pinus* spp. monocultures have increased from 886 393 ha in 2001 to 1 140 113 ha in 2006, a 29 % increase (Xavier and Leite 2008).

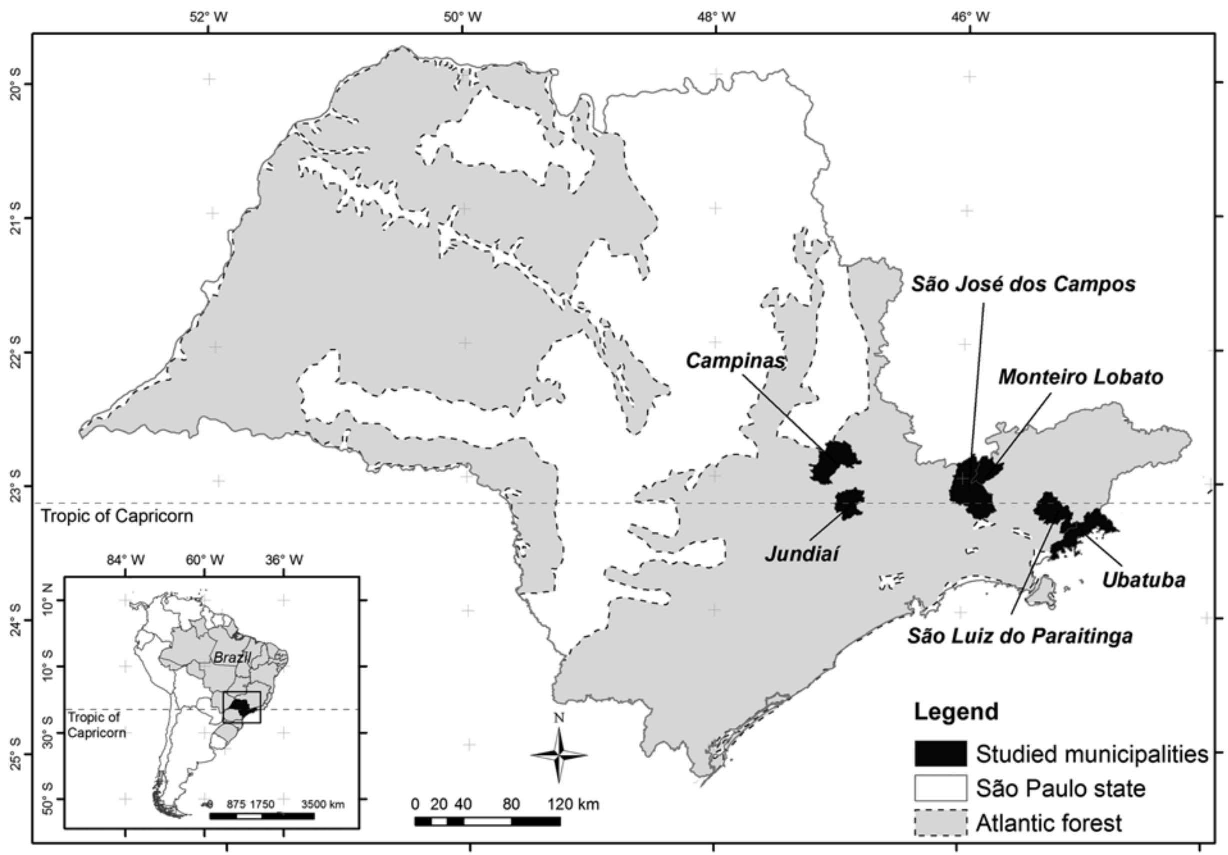

Figure 4.1 Locations of the six municipalities studied in the state of São Paulo and legal limits of the Atlantic Forest. (Map courtesy of Allan Yu I. de Mello, based on data from Instituto Brasileiro de Geografia e Estatística and SOS Mata Atlântica/Instituto Nacional de Pesquisas Espaciais)

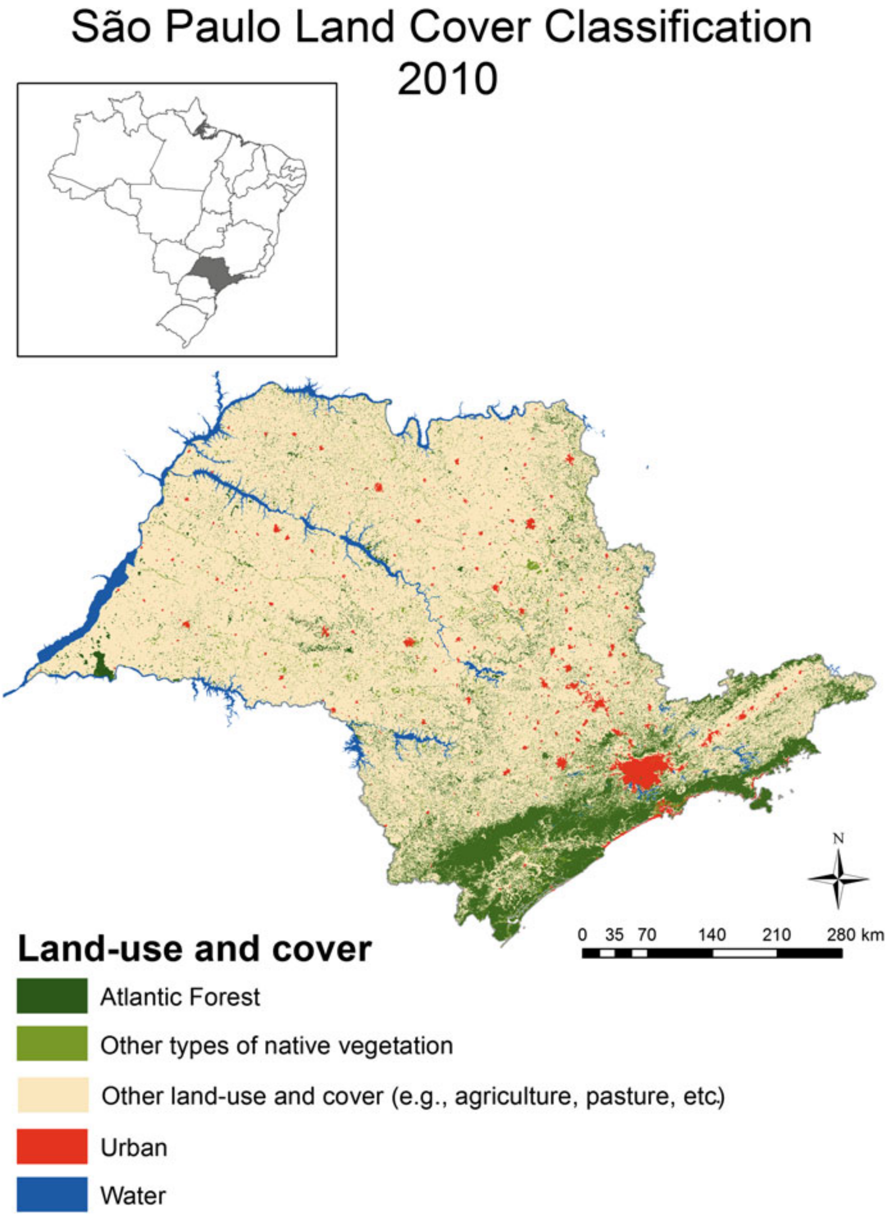

Figure 4.2 Land-use and cover classification in the state of São Paulo (2010). (Map courtesy of Allan Yu I. de Mello, based on data from Instituto Florestal and Secretaria do Meio Ambiente do Estado de São Paulo)

Our study also included six counties in south-central Indiana: Monroe, Morgan, Decatur, Jennings, Scott, and Washington (Fig. 4.3). South-central Indiana is characterized by a mixture of low hills, forest, pasture, and agricultural crop production (mainly corn and soybeans). The mix of small-scale forests and agricultural land uses offers a range of rural amenities to a growing group of residential landowners, effectively blending the rural with the urban (Kauneckis and York 2009, Koontz 2001). In this regard, south-central Indiana is similar to other regions in the American Midwest that are experiencing residential expansion, declining agricultural land use, and peri-urban reforestation (Deller et al. 2001, Erickson et al. 2002, Kauneckis and York 2009).

Indiana provides a representative example of the forest transition in the United States. European homesteaders began to settle the region in 1810 and quickly cleared the mostly forested landscape (Madison 1986). In the late nineteenth century, Indiana had among the largest timber harvests in the United States (Parker 1997, Streightoff and Streightoff 1916). By the end of the nineteenth century, the old-growth forests that had covered 92 % of the state prior to European settlement were almost entirely eliminated and replaced by farmland (Parker 1997). Beginning with the Great Depression and the industrial development of Chicago in the early twentieth century, rural out-migration and farm failures led to farmland abandonment and forest regrowth, so that today forest covers 20 % of the state, with the largest forested areas on steeper terrain (Fig. 4.4).

Recent trends show a steady increase in Indiana's forested area (Gallion and Woodall 2010), from 6 % of the state's area in the early 1900s to 20 % today, with most regrowth occurring in small parcels of land owned by approximately 190 000 private landowners (IDNR 2008, Woodall et al. 2005). Private individuals own 86 % of all forest in Indiana, equivalent to about 1.4 million ha (IDNR 2008). The majority own parcels smaller than 40 ha (Gallion and Woodall 2010). The main income sources for most landowners include nonfarm employment, agricultural work, and forestry activities (Evans and Kelley 2004, Koontz 2001).

4.4.2 Analytical approach

Our findings draw on survey responses, landowner interviews, and satellite imagery for São Paulo and Indiana. Household survey instruments were pretested prior to collection of field data. Where possible, survey responses were integrated with land-use and cover change data to produce household and subregional characterizations of forest-change trajectories. In São Paulo, maps showing property boundaries did not exist, nor could they be produced during our research. This limited our ability to integrate the survey and satellite data for São Paulo. Slight variations in the data collection approaches constrain a complete comparison between São Paulo and

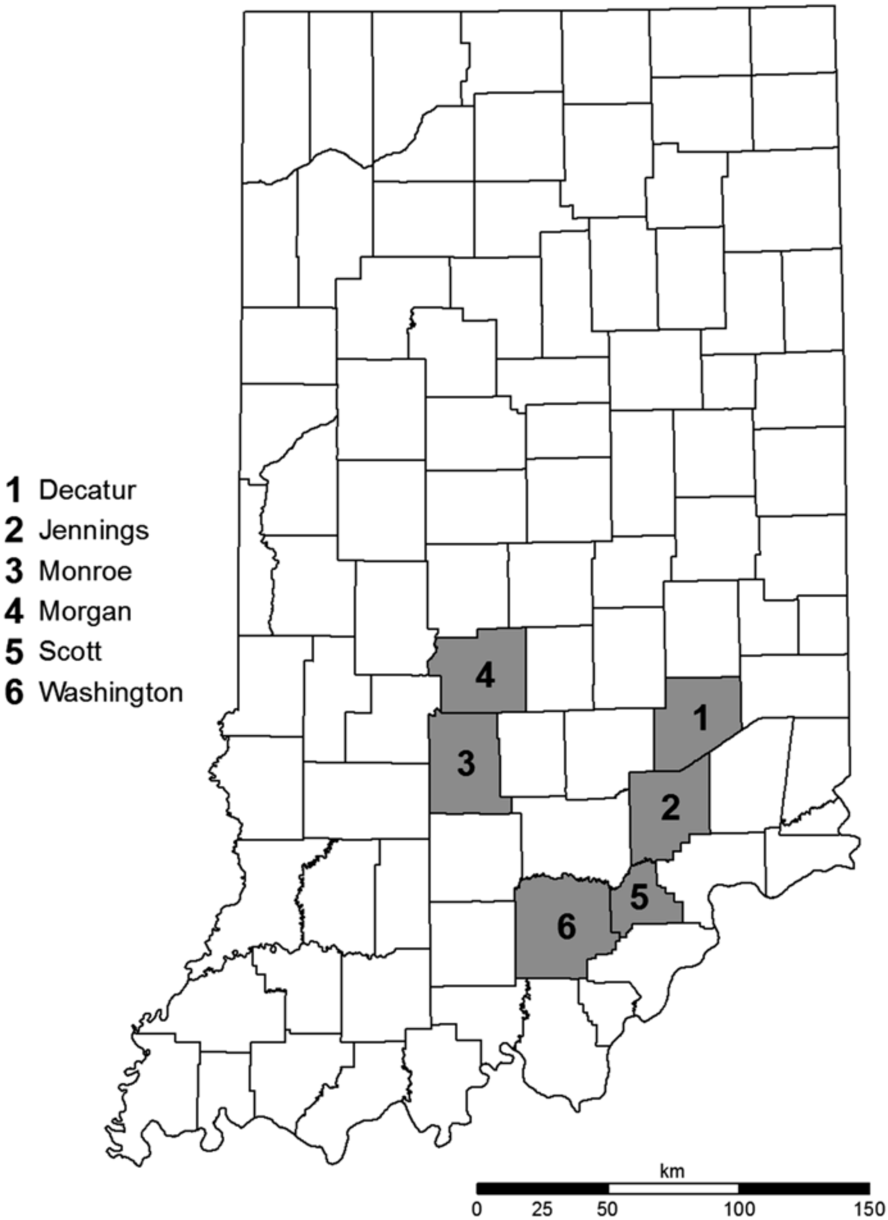

Figure 4.3 The study area in south-central Indiana, showing the six surveyed counties. (Map courtesy of the Center for the Study of Institutions, Population, and Environmental Change, 2009)

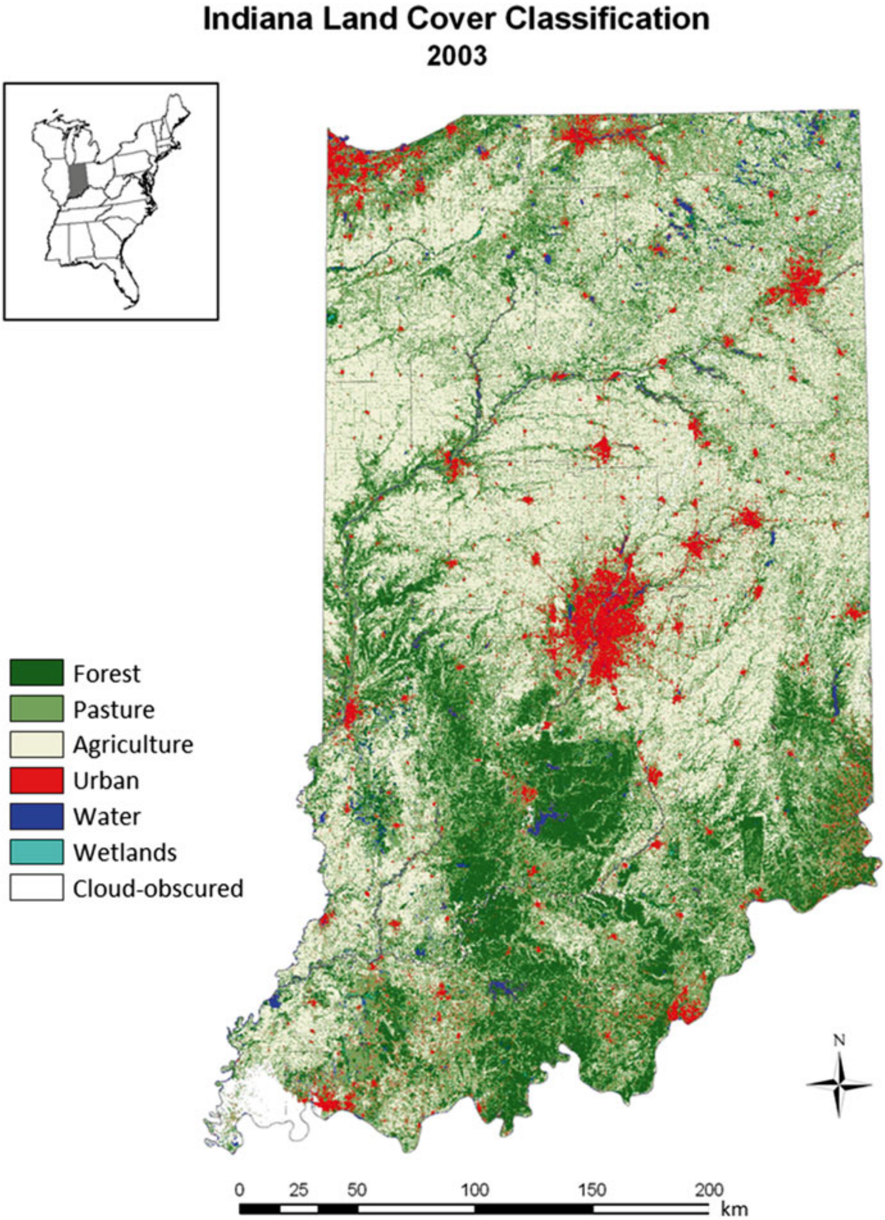

Figure 4.4 Land-use and cover classification in Indiana (2003). (*Source*: U.S. Department of Agriculture, National Agricultural Statistics Service)

Indiana, as well. Nevertheless, we compared the motivations for reforestation in the past 5 years using factor analysis for 15 identical survey items that measured stated motivations. We found three common motivational components or drivers (see Sect. 5).

4.4.3 Findings from São Paulo

We analyzed 537 structured interviews (household surveys) from nonindustrial rural properties in six municipalities in São Paulo. We focused on reforestation between 2003 and 2008 reported by landowners. We used the cluster sampling technique (Stuart 1962) to obtain this data. We overlaid a map with the geographic locations (points) of the rural properties in each municipality, provided by the Coordenadoria de Assistência Técnica Integral of São Paulo, on satellite images and road network maps. As our main goal was to understand the motivations that lead landowners to increase forest cover, this enabled identification of clusters of properties located near forested areas, which had a greater probability of revealing forest increases. These groupings were randomized, and a team of four interviewers explored the roads and visited all the properties until at least 100 interviews had been conducted for plots larger than 2 ha in each municipality. We removed 63 surveys from the sample used for the present analysis due to missing values for specific variables, leaving 537 valid surveys for analysis.

In addition to the household survey, we conducted semi-structured interviews with landowners and rural inhabitants ($n = 24$) and with environmentalists, government managers, and tourism entrepreneurs ($n = 15$) in São Luiz do Paraitinga. These interviews were recorded and transcribed and were then analyzed using the qualitative procedures of coding and categorization; the consistency of the information was checked via triangulation (Creswell 2003).

4.4.3.1 Land-use and management decisions

Of the total area of the properties in our study in 2008, approximately 47 % of the land was forested (39 % with native species and 8 % with exotic species), 42 % was pasture, 5 % was cropland, and 2 % was fallow. Landowners reported that radio, newspaper, and television were the most important sources of information for land management decisions (used by 32 % of landowners), followed by friends and family (24 %), printed information (21 %), Internet resources (18 %), contracted professionals (15 %), neighbors (14 %), and government professionals (6 %). Family incomes included significant contributions from off-farm jobs (50 %), land production (35 %), and social security payments (29 %). The main professional occupation for about 35 % of the landowners was related to land production (e.g., cattle breeder, farmer), 15 % were retired, and the remaining half declared a diverse set of occupations. About 42 % of landowners had the equivalent of a college degree or higher, but 32 % had no or few years of formal education.

4.4.3.2 Reforestation in São Paulo

Classification of Landsat Thematic Mapper (TM) and ETM+ imagery indicated that between 1988 and 2007, all of the studied municipalities except Ubatuba had a net

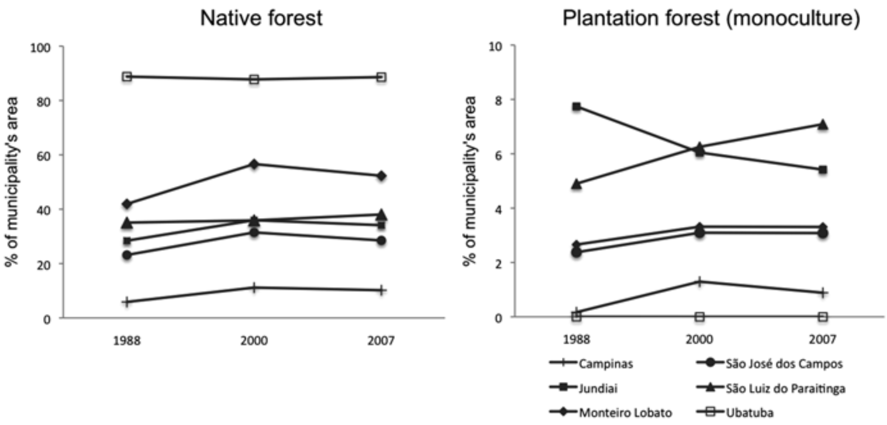

Figure 4.5 Forest cover trajectories in six municipalities in eastern São Paulo

increase in native forest cover, and all except Jundiaí (a decrease) and Ubatuba (no change) showed a net increase in monoculture forest cover (Fig. 4.5).

Of the landowners, 60 % indicated that the forest cover on their properties increased during the preceding 5 years. Regeneration through secondary succession was mentioned by 78 % of landowners, and 34 % reported planting trees. Pasture (48 %) and riparian areas (50 %) were the most common types of land converted into forest. On the one hand, this can be explained as forest regrowth in abandoned land; on the other, it represents compliance with Brazilian legislation that mandates protection of rivers and springs by vegetation buffers.

When asked why forest cover increased in their land, most landowners reported conservation and esthetic values as important motivations. Economic incentives, wood production, and professional advice were less important or unimportant. About 41 % of landowners expressed plans to reforest their land in the near future, motivated mostly by environmental conservation, esthetic values, and desired improvements in water quality. These results indicate the penetration of environmental discourse into rural zones of São Paulo, concomitant with declining land use for grazing or agricultural purposes. The property owners who increased forest cover tended to have a higher degree of formal education and to be employed in activities unrelated to land production. However, we did not detect a negative correlation between farm-based income and past reforestation, which suggests that reforestation might be compatible with productive land uses and with an economic dependence on the land. Due to a lack of information on property boundaries, we could not determine the extent of the forest increase for each property. Thus, our analyses focused on the presence or absence of reforestation as the primary outcome. Farinaci (2012) provides a detailed description of our statistical analyses.

Our results indicate that past reforestation and the intention to reforest in the near future were positively related to property size. Larger properties had higher percentages of native forest than smaller properties and were more likely to be reforested. A similar relationship was previously found in the Amazon, where deforestation intensity decreased with increasing property size (e.g., D'Antona et al. 2006, Fearnside 2005, Michalski et al. 2010). This can be explained by the higher land-use intensity on smaller farms, higher costs of maintaining cleared land on larger farms, ecological processes related to forest regeneration near existing forest areas, or a combination of these factors.

Our analysis of data from the semi-structured interviews provided a more detailed understanding of the factors that influence land-use and cover change in São Luiz do Paraitinga. Landowners generally perceived the forest area to be increasing in at least some parts of the municipality and cited a diverse set of interrelated motivating factors. The decline of dairy farming was probably the most important process leading to land abandonment and forest recovery. Modernization of the dairy farming industry, competition from other regions, and introduction of exotic grasses led to a loss of rural jobs and declining profits. Therefore, several landowners sold their land or reduced their activities to subsistence levels. In addition, declining soil fertility, a lack of investments to restore fertility, and steep slopes restricted the range of alternatives and productive land uses. Concomitantly, increases in the number of people willing to purchase land for leisure or long-term investment, legal restrictions on timber harvests, fire monitoring, and proximity to the Serra do Mar State Park were other important factors in the increased forest area in São Luiz do Paraitinga. Finally, interviewees frequently related the increased forest area to the importance of nature conservation and rural amenities (e.g., water and air quality, scenic beauty, wildlife), a finding that echoes our household survey results.

4.4.4 Findings from Indiana

Our results from Indiana are based on survey responses from 1939 nonindustrial private forest owners (with a 28.8 % survey response rate). We constructed two samples of the landowners: one was a random sample drawn from all parcels in each of the six counties, and the second was drawn from landowners on whose land reforestation was evident from satellite imagery. We used the latter sample to ensure that we had a sufficient number of responses to characterize the attributes and preferences of landowners who had reforested their properties through tree planting or abandonment of agricultural land. We also conducted follow-up interviews with a subset of the landowners ($n = 42$). All landowners were non-urban residents who owned more than 2 ha of land. The survey responses were integrated with land-use and cover change data derived from Landsat TM satellite imagery (Evans et al. 2001, 2010; Sweeney and Evans 2012) to produce household and subregional characterizations of land-use and cover change trajectories at different spatial scales. Here, we primarily focus on the subregional rates of change to describe the social and biophysical conditions related to reforestation within the study area.

4.4.4.1 Land-use and management decisions

Our research found a diversity of land-use preferences among Indiana landowners, which is characteristic of forest owners in developed countries (Butler 2008, Hujala et al. 2009, Karppinen 1998). At the local level, multiple forest values, ownership characteristics, and biophysical constraints are associated with forest-cover change. The landowners in our study had a mixed land-use portfolio: on average, 39 % of the land was forested, 36 % was cropland, 15 % was mowed or used for hay production, and about 10 % was grazed. Roughly 40 % of landowners reported that they or their family had owned the land for more than 30 years. Length of ownership is an important measure of knowledge and forest management experience, emotional attachment to a place, and the desire to leave a legacy for future generations (Nagubadi et al. 1996, Rickenbach and Kittredge 2009). This may be changing, however, since nearly two-thirds of Indiana landowners did not expect their children to live on the land as adults—a finding that suggests a diminishing perception of land as a family legacy. Off-farm employment was a significant income contribution for more than half of the landowners. One-third had some college or technical degree training, and another third were retired.

Over two-thirds of landowners identified family and friends, along with printed information, as very or somewhat useful sources of information for making management choices. About 50 % of landowners pointed to neighbors as useful sources of information, and less than half mentioned private consultants, government professionals, and the Internet. Information exchange between landowners and natural resource management professionals was limited. Most (>50 %) of the landowners were unfamiliar with existing federal and state forest assistance programs, for example. Over 89 % had never participated in such programs, but a sizable proportion (35 to 43 %) had heard of the federal Conservation Reserve Program and the state's Classified Forest Program—two of the major government programs that promote forest conservation on private land.

Past management behavior, such as the decision to harvest timber, was an important indicator of land-use preferences. For example, 24 % had cut trees in the past 5 years, and of those, 54 % harvested timber for commercial sale and 56 % cut firewood for personal use. Financial motivations were rarely a driving factor—a finding consistent with prior research (Koontz 2001). For most landowners, the decision to harvest was based on a longer time horizon and the desire to improve the forest's health. Most landowners had cut trees to remove mature trees and improve the quality of the remaining trees, improve wildlife habitat, supply wood for their own use, or achieve objectives in a management plan. Professional foresters assisted in 13 % of these harvests.

In addition, changes in neighboring land were important contextual factors that affected forest land use. Residential development on nearby land was reported by more than half of the landowners. Timber harvesting was the second most commonly observed change in surrounding land and was positively correlated with landowner intentions to plant trees. This finding suggests the importance of the social context and influences of adjacent land-use practices on household decisionmaking (Knoot and Rickenbach 2011, Korhonen et al. 2012, Rickenbach et al. 2011).

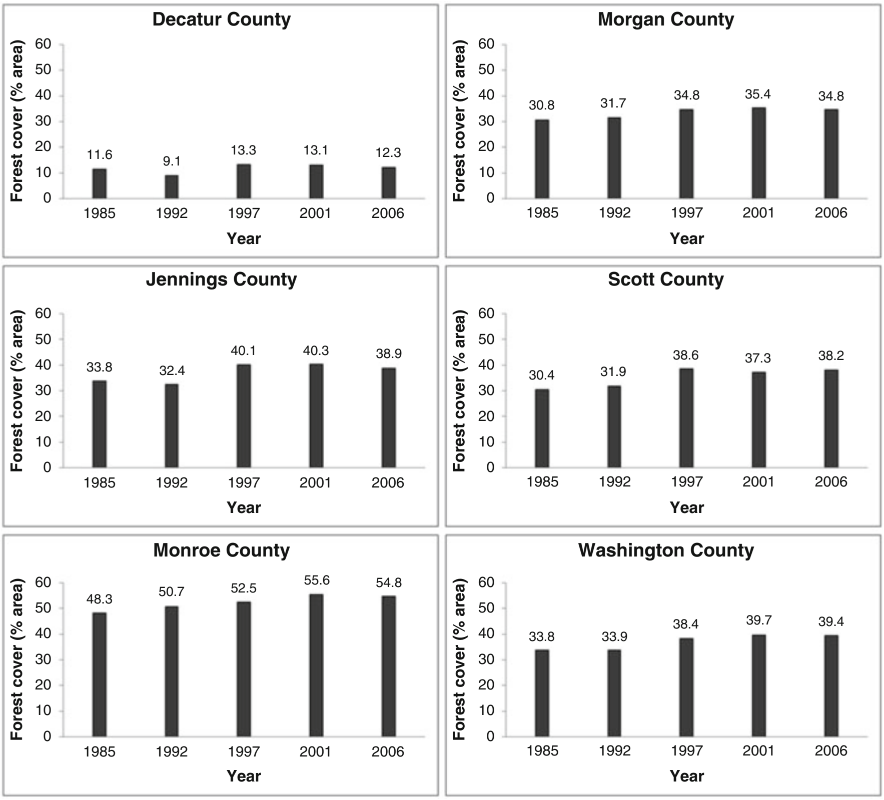

Figure 4.6 Forest cover trajectories for six counties in south-central Indiana. (Courtesy of the Center for the Study of Institutions, Population, and Environmental Change, 2009)

4.4.4.2 Reforestation in Indiana

Similar to the case in São Paulo, we examined reforestation on private land in south-central Indiana. Analysis of cover changes using Landsat TM data revealed evidence for a modest forest-cover increase. The observed increase in forest cover area varied between 0.7 % points for Decatur County and 7.8 % points for Scott County from 1985 to 2006 (Fig. 4.6). In addition, among the 20 % of landowners who reported an increase in the forest area in the past 5 years, 63 % indicated an increase of less than 0.8 ha. Pasture was the most common type of land converted into forest. Forest regrowth mostly resulted from natural succession—i.e., leaving the land alone and letting it return to forest. A large percentage of landowners (44 %) also planted trees themselves or contracted with someone to plant trees.

Landowners were asked about their reasons for reforestation. As in São Paulo, most Indiana landowners were concerned with nature conservation and esthetics. Economic incentives, land protection, and professional advice were also important decision factors. However, we observed differences in motivations based on the size of the forest area increase and length of property ownership. Individuals who had owned their land for less than 10 years expressed the highest level of desire for forest conservation. Economic incentives and land protection were important among those with relatively short ownerships (5 to 10 years) and a more substantial increase in forest area (by 2.4 to 4 ha). We also found that past reforestation was significantly related to income derived from farming, timber harvesting, and land leasing, but that there was little connection between off-farm income and reported reforestation.

Most Indiana landowners (80 %) preferred their land to remain the same in the future, and about 16 % expressed an interest in having more forest or a mix of forest and open space. Close to 23 % of landowners intended to plant trees in the next 5 years. This represented the most common choice among a range of future land uses, such as timber harvesting, selling the land, and residential development. Most Indiana landowners are considering tree planting due to extreme weather (e.g., tornados, storms) or flooding problems, and as a result of available incentives. Government incentives, such as free seedlings, technical assistance, and direct payments, were important drivers of future land-use preferences.

We found that free seedlings and direct payments were positively associated with an intention to plant trees in the next 5 years. Expected timber price increases were particularly important for owners of larger properties. In addition, the owner's time horizon—a reflection of their age—underlay many of their intentions and preferences. For instance, we found that older owners were less likely to reforest their land. Other barriers included biophysical constraints, uncertainties related to nearby infrastructure projects, and perceptions of a lack of control over land management. In short, a combination of diverse landowner values, land-use preferences, and biophysical constraints affected local-level reforestation dynamics in south-central Indiana. The land's biophysical attributes were important factors, particularly on properties where the land was unsuitable for cultivation, on steep slopes, and in low-lying areas.

4.4.5 Comparison of motivations for reforestation in Indiana and São Paulo

Differences in the research methods limit the scope of comparisons between the two study areas. Nevertheless, the objectives and survey questions overlapped substantially, allowing an identification and comparison of trends. We used a merged dataset of the survey responses from Indiana and São Paulo in this part of the analysis. A factor analysis focused on 15 questions that were identical in the two studies about motivations for a forest cover increase to identify commonalities and uncover the main motivational drivers. The importance of each motivational factor was calculated as the mean response. Motivation-related questions (e.g., "I felt the land should be put into timber production", "To enhance the scenic beauty of the land", or "Tax benefits were available") were measured on a three-point ordinal scale (very important, somewhat important, not important for the forest area increase).

Although there were some differences in the two case studies, we found substantial similarities in the landowners' stated motivations to reforest. Most notable was the overwhelming role of conservation goals among landowners in both Indiana and São Paulo. Our analysis revealed three main drivers of reforestation. The first and most important one was a *conservation ethic*, which reflected the desire to enhance scenic beauty, conserve nature, protect forests for future generations, and provide food and habitat for wildlife. The second important motivation was an *economic incentive*, including government cost-sharing, low-cost seedlings, tax incentives, and timber sales. The third motivational factor was *to protect the land*, in particular to improve water quality and provide a windbreak.

A conservation ethic was consistently strong and important in both Indiana and São Paulo. These motivations were slightly more pronounced in São Paulo, where 93 % of the landowners cited nature conservation as very important in their reforestation decisions, whereas in Indiana, 81 % identified nature conservation as important or very important.

Initially, we expected that the different legal, political, and economic contexts of São Paulo and Indiana would result in contrasting incentives for reforestation. For instance, tax breaks and economic incentives for forest conservation are more accessible in Indiana. Although landowner responses to economic incentives differed, economic incentives were secondary to conservation motivations in both Indiana and São Paulo. In Indiana, landowners often benefitted from tax breaks, but in São Paulo, few landowners stated that tax incentives were important factors in their decision to reforest. Some of these differences were attenuated when the length of property ownership was accounted for. Similarly, differences related to the desire for land protection disappeared when we controlled for the length of property ownership. The only exception was for owners with 11 to 30 years of ownership, among whom Indiana landowners showed a greater motivation than their Brazilian counterparts to use reforestation as a way to improve water quality or provide a windbreak (i.e., protect their land).

Initially, we hypothesized that the social context, in terms of the land-use practices on neighboring lands, would be an important influence on land-use choices. In both case studies, however, neighbors' activities had a minimal effect on preferences for future land use. Only 5 % of Indiana landowners and less than 1 % of the São Paulo landowners indicated that seeing neighbors plant trees had affected their decisions. In summary, the most important finding was the role of a conservation ethic as a key driver of household-level reforestation in both Indiana and São Paulo.

4.5 Concluding remarks

Our data and previous work revealed a number of factors that can be associated with increased forest cover. However, the complex causal linkages and interactions among these factors can vary among regions, making it difficult to generalize and predict future forest-cover trajectories.

Where social and economic incentives blend with favorable policies, the relationship between people and forests can lead to reforestation. Despite this, net reforestation cannot outweigh net deforestation on a global scale. Among other factors, the increasing demand for biofuels and meat and the increasing standard of living in some countries lead to both direct and indirect adverse impacts on forests. Moreover, human influences have nonlocal impacts, as in the case when net reforestation in one country is sustained by deforestation in another.

Domestic institutions and policy approaches are essential in determining the balance. Although institutional arrangements play critical roles in promoting forest conservation or permitting forest destruction, we found no clear connections between the effects of national- and state-level forest policies and changes in forest cover in the United States and Brazil. However, our focus on states within the two countries let us examine the social and environmental processes underlying changes in forest cover in greater detail, at regional and local levels. Among the drivers, human values and attitudes appear to be the key to forest conservation and reforestation. In both Indiana and São Paulo, nature conservation and esthetics were important motivations for increasing forest cover on private land. In addition, education was positively correlated with reforestation in both Indiana and São Paulo. This mirrors the generally positive correlation between education, as a measure of socioeconomic status, and concern for the natural environment. Nevertheless, it is unclear whether education itself is a driver of forest conservation and recovery or whether a higher level of education indicates a landholder whose lifestyle depends less on land production. Whereas dependence on nonfarm income was associated with reforestation in São Paulo, reforestation in Indiana was associated with income from farm-based activities.

Overall, our study illustrates the difficulty of creating generalizations that are suitable to all countries and regions. Important human drivers are the foundation of forest change, from high-level government policy to regional and local institutions, household livelihood strategies, and individual-level behaviors. To understand the influence of people on forests, researchers must understand human institutions at all these levels and account for the diversity of social and environmental factors that exist between regions and across scales.

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Chapter 5
Changes in the ecosystem services provided by forests and their economic valuation: a review

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Abstract In this chapter we discuss the trends in forest change and the associated drivers, the economic value of forests, the principles and challenges in evaluating the economic value of forests, and the role of valuation in informing decision-making. We address current major forest conservation initiatives at different scales and the mechanisms involved, whether supported by economic valuation or not. Today, 30 % of the world's forests are designated for productive functions, 24 % for multiple uses, 11.5 % for biodiversity conservation, 8.2 % for protective functions, and 3.7 % for social functions. The remaining 22.6 % are designated for other uses or remain unclassified. Global trends indicate that although the area of intensively managed forest continues to expand, the global extent of conservation and protective forests is also increasing as a result of political efforts to preserve and restore the ecological functions of forests. Forest management practices are potentially better supported by extended cost–benefit analyses that require an economic valuation of the whole array of benefits, whether market or non-market, provided by forests. Although we acknowledge other values and decision-making and support tools, the focus of the chapter is on the economic valuation approach. Our review in this chapter

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was guided by the goal of updating previous reviews of these topics. We have provided additional evidence that forests contribute to human well-being in many ways, and use the concept of ecosystem services as a building block to better understand, frame, and assess the economic benefits we derive from well-functioning forests.

5.1 Forest ecosystem services

Ecosystem services are broadly defined as the benefits that people obtain, directly or indirectly, from ecosystems (MEA 2005), and that contribute to human well-being when combined with other factors such as education, health care, and social equity.

The interdependency between ecological and social systems can be seen as a feedback loop: human well-being depends on the delivery of ecosystem services, but the capacity of ecosystems to deliver services depends on ecosystem conditions, which in turn are affected by society's choices about how to use ecosystem services and manage the ecosystems that provide them. These choices are greatly influenced by the level of human well-being and by the way society perceives and values ecosystem services.

Forests provide many ecosystem services, including supporting a large percentage of the world's biodiversity and contributing to human well-being at local (e.g., wood production), regional (e.g., groundwater recharge), and global (e.g., climate regulation) scales. The most easily understood and most quantifiable source of benefits derived from well-functioning forests pertains to the provision of goods and materials, even if their provision is not directly observed. Water, a basic and valuable good required for human existence, is a suitable first example. Though it is not obvious to the untrained eye, forests interact closely with and affect the hydrological cycle through evapotranspiration and their ability to increase infiltration into the soil by decreasing runoff; thus, forests are a key source of freshwater resources (Wang and Fu 2013). For example, about 80 % of the freshwater resources in the United States at the turn of the century originated from forests, which covered, at that time, about one-third of the country's surface area (USDA 1999). Human use and management of forest ecosystems can change the level of ecosystem services delivery and induce the production of one service to the detriment of others. This is the case for productive forests, which are planted and managed to produce timber, and the case for protective forests, which are planted or managed to prevent or reduce soil erosion.

Less quantifiable benefits that are often inadequately addressed include the benefits people obtain from forests through abstract concepts such as *esthetic*, *spiritual*, and *inspirational* values—which are called cultural services. Unlike timber production or soil erosion control, these benefits are not physically measurable. Instead, they take the form of *experiences* people can obtain from forests (Kareiva et al. 2011). Because they are intangible, communicating this category of benefits is more difficult, even when attempts are made to express the values in monetary terms. For forest ecosystems, a significant part of these benefits relates to recreational opportunities.

In southern Africa, for instance, trees play a crucial role in the cultural and spiritual lives of local communities (Sileshi et al. 2007), despite any hypothetical benefit they provide as tourist attractions. The inherent complexity of valuing people's experiences is well acknowledged in the literature (Boyd and Banzhaf 2007). We examine the methods for valuing ecosystem services more closely in Sect. 4.

5.2 Classification of ecosystem services and conceptual approaches

Exhaustively listing the whole array of benefits people obtain from ecosystems can be a challenging task, and some sort of labeling and operationalization of the concept was required at the beginning of efforts to conceptualize ecosystem services. Multiple classification systems for ecosystem services have evolved, and this variety has been justified by the premise that classification systems should focus on the purpose and context of the study (Costanza 2008). Notwithstanding, one of the most generally used classifications was described by the Millennium Ecosystem Assessment (MEA 2005), which distinguishes among four categories of services, which MEA defines as "the benefits that people derive from ecosystems": provisioning services (products obtained from ecosystems), regulating services (benefits obtained from the self-regulation of ecosystems), cultural services (non-material benefits obtained from ecosystems), and supporting services (services that are necessary for the production of other ecosystem services). Moreover, in this classification, biodiversity is understood as not only underpinning the ecosystem services but also as an ecosystem service itself; for example, medicinal plants are a provisioning service, whereas bird-watching is a cultural service.

Although this classification is still in use and it is generally accepted, it has some drawbacks. In particular, the consideration of supporting services as a separate category often leads to overlapping estimates and double-counting; this issue is a particular concern if economic valuation is to be undertaken, as we discuss later in the chapter. Another ecosystem service classification emerged from a more recent global initiative, *The Economics of Ecosystems and Biodiversity* (TEEB 2010). Although TEEB is similar to the MEA classification, TEEB considers the supporting services only as ecological processes, and introduces a new category called "habitat" services (TEEB 2010) to highlight the importance of ecosystems in providing habitat for migratory species and as gene-pool protectors. The TEEB classification and its approach differ from the MEA approach because TEEB explicitly aims to incorporate an economic analysis of changes in ecosystem services.

The Common International Classification of Ecosystem Services (Haines-Young and Potschin 2010) does not aim to replace existing classifications, but rather provides a framework that enables translation among different classifications and links to other classification systems that are used in economic and environmental accounting. Each classification has its own purposes, drawbacks, and advantages. The MEA and TEEB approaches are directed more at assessment and valuation of ecosystem services, whereas the CICES approach was conceived as a system compatible with the design of integrated environmental and economic accounting methods (Maes et al. 2013). Those who advocate the use of CICES have pointed out that this classification at least potentially helps to overcome the problem of double-counting. This topic has been widely addressed in the literature (e.g., Boyd and Banzhaf 2007, Fisher et al. 2009, Mace and Bateman 2011), with debate focusing on the need to distinguish between services and benefits when economic valuation or environmental accounting is the purpose of the study. In essence, and regardless of the specific nomenclature adopted by each author in explaining their rationale, valuation should only be applied to things that are directly consumed by beneficiaries given that the values of ecological processes are already embedded in that final output. Some argue (Bateman et al. 2010) that if other input capitals are used to generate a benefit, they should be subtracted from the estimated value of the benefit to provide the net benefit.

Perhaps more important than finding a sovereign and unifying classification system or approach, we argue in this chapter that a deep understanding of the ecological dynamics of forest ecosystems is necessary to generate powerful insights into details of the chain of benefit delivery, and can therefore help managers to identify the best management options based on a more fully informed economic valuation.

5.3 Global trends and drivers of forest ecosystem services

5.3.1 Past, current, and future trends for forest systems

Human activity has caused the loss of about 40 % of the planet's original forests since preagricultural times, starting ca. 8000 years ago (Shvidenko et al. 2005). Temperate regions, such as Europe and North America, were particularly affected, losing more than 50 % of their natural forest cover before the mid-twentieth century (Fig. 5.1; Kaplan et al. 2009, MEA 2005). In tropical regions, the loss of forest cover has been less severe, but became a pervasive trend during the last half-century and will probably continue during the twenty-first century (Fig. 5.1; FAO 2012, MEA 2005).

In temperate and boreal regions, laws and policies to protect forests and to reverse deforestation emerged as a response to the shortage of timber and fuelwood and to the degradation of the forest's protective functions (Farrell et al. 2000, Rudel et al. 2005). Moreover, rural abandonment due to economic growth, improvements in agricultural efficiency, and the replacement of wood by fossil fuels as a source of energy also decreased pressures on forests (FAO 2012, Kaplan et al. 2009). Reforestation initiatives in the twentieth century, but also a few centuries ago, and natural forest regeneration following land abandonment restored much of the forest cover and helped to halt forest decline in temperate and boreal regions (Hobbs and Cramer 2007, Keenleyside et al. 2010, Rudel et al. 2005).

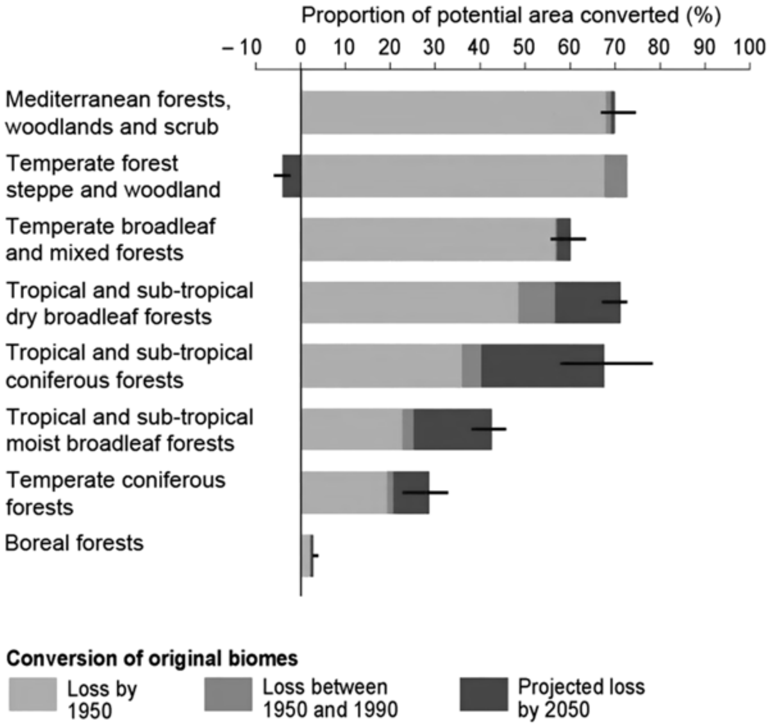

Figure 5.1 Past forest losses and projected future losses in the world's main forest systems. The proportion of the forest lost before 1950 was estimated based on the potential distribution of each forest system based on soil and climatic conditions. Projections of forest loss correspond to the average value of projections obtained for the four Millennium Ecosystem Assessment future scenarios; error bars indicate the range of values for the four future scenarios. Adapted from the Millennium Ecosystem Assessment (MEA 2005)

Globally, the overall extent of natural forests continues to decline, and the expansion of new forests does not compensate for the loss of natural ones (Butchart et al. 2010, FAO 2011). Moreover, the value lost with the degradation or deforestation of old-growth forests cannot be fully replaced by new forests because planted and regenerated forests differ from natural stands in many characteristics (Rey Benayas et al. 2009). First, restored forests do not support the same biotic communities as old-growth forests (Hobbs and Cramer 2007, Rey Benayas et al. 2009). Second, most new forests are located in temperate regions and cannot replace the biodiversity lost in highly diverse tropical regions; that is, the creation of forests in one region may not compensate for the destruction of forests in another region. Third, many planted forests are grown and managed for industrial purposes, so their contribution to biodiversity conservation and to the delivery of regulating and cultural services is modest or even negative (Kanowski 2003; Proença et al. 2010a, b). For instance, when continuous forest plantations replace traditional landscape mosaics, there is a loss of landscape heterogeneity and a decline, or even local

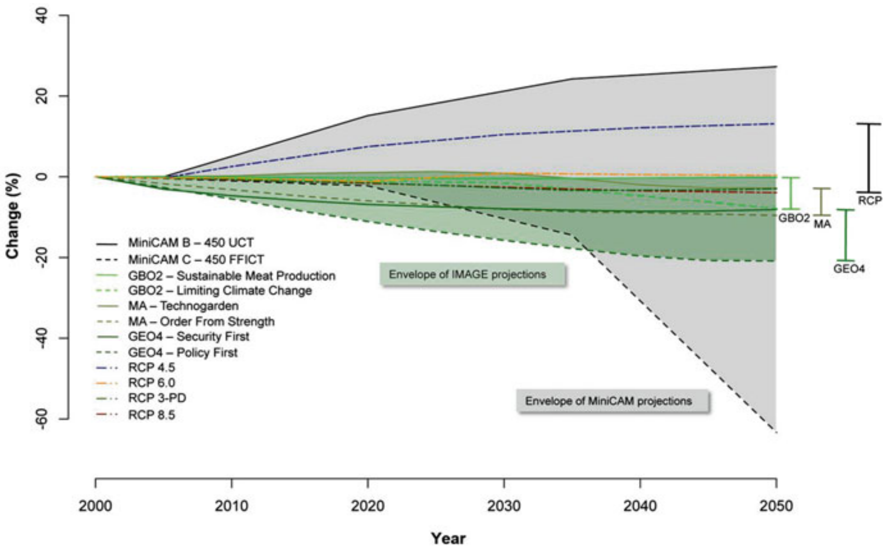

Figure 5.2 Projected changes in global forest cover until 2050 under various global scenarios: the Millennium Ecosystem Assessment (MEA) scenarios (Sala et al. 2005), the Global Biodiversity Outlook 2 scenarios (ten Brink et al. 2006), the Global Environmental Outlook 4 scenarios (UNEP 2007), the representative concentration pathway scenarios (Hurtt et al. 2009), and the MiniCAM scenarios (Wise et al. 2009). For each set of scenarios, we have only shown the two most contrasting results. The wider envelope for the MiniCAM projections, compared to the envelope of scenarios with the IMAGE model (IMAGE-team 2001), suggests that there are opportunities for action to reverse the global trend of forest decline, but also that wrong policy choices can exacerbate the loss of forest cover compared with the other scenario assessments. *Sources*: Leadley et al. (2010), Pereira et al. (2010)

extinction, of species associated with open habitats such as grasslands and meadows (Poyatos et al. 2003, Reino et al. 2009).

The global area of forest will probably continue to decline if countries fail to provide adequate incentives to halt deforestation. Global policy choices influence society's choices and may play a critical role in determining the selection of land-use options. A recent study by Wise et al. (2009) explored the effect of different carbon taxation policies on global land-use changes. The authors found that imposing a global carbon tax covering anthropogenic carbon emissions from all sectors, including emissions from land-use change, would promote the protection and expansion of forests, leading to an increase in forest cover (Fig. 5.2; the MiniCAM B scenario).

However, taxing only fossil fuel and industry emissions may prompt the expansion of biofuels and lead to a drastic loss of forest cover worldwide (Fig. 5.2; MiniCAM C scenario). Previous scenarios, which were also based on socioeconomic drivers, projected less drastic changes in forest cover (Fig. 5.2; MEA, GBO2, and GEO4 scenarios). The narrower range of variation in forest area projected by these scenarios is in part explained by compensatory mechanisms in the underlying socioeconomic scenarios, which lead to a convergence of the trend lines for changes in forest cover area. For instance, the option for biofuels in the "environmentally friendly" scenarios implies the replacement of forests by biofuel crops (Leadley et al. 2010, Pereira et al. 2010).

Forest use and deforestation in the northern hemisphere are historically associated not only with socioeconomic development but also with ecosystem degradation, a shortage of forest products, and environmental disasters, such as flooding, which later motivated forest restoration and sustainable forest management in these regions (FAO 2012). Today, tropical forests are the ones most exposed to deforestation and forest degradation. The unsustainable use of forest resources jeopardizes socioeconomic development and human well-being in these regions (Rodrigues et al. 2009), and its negative impacts may also be felt at larger spatial and temporal scales (Leadley et al. 2010). Halting unsustainable use trends requires action from local to global levels and the adoption of socioeconomic development pathways that will ensure the sustainable use of forests, including the management of both tangible and non-tangible services, and sustainable support for human welfare. The management of forests and forest ecosystem services, and particularly non-provisioning services, should be based on polycentric and diverse governance systems that ensure an equitable representation of users and that promote knowledge sharing, collective decision-making, and enforcement of management decisions (Ostrom 2009).

5.3.2 Global trends in the use of forest ecosystem services

From a utilitarian perspective, natural forests have by default a multifunctional nature in the sense that they can simultaneously deliver several benefits to people, including forest goods (e.g., fuelwood, medicinal herbs, bush meat), regulating services (e.g., climate regulation, soil protection), and cultural benefits (e.g., esthetic pleasure, sacred groves). Still, some forests are designated and managed for a particular function, such as industrial plantations and forests in protected areas. Today, 30 % of the world's forests are primarily designated for productive functions (production of wood, fiber, biomass, and non-wood forest products), 24 % for multiple uses (forests managed to deliver a range of benefits without the dominance of a particular function), 11.5 % for the conservation of biodiversity, 8.2 % for protective functions (conservation of ecosystem functions and processes underlying the delivery of regulating services), and 3.7 % for social functions (recreation, education, and conservation of cultural heritage) (FAO 2011). The remaining 22.6 % are designated for other uses or remain unclassified.

The annual rate of growth between 2000 and 2010 was particularly high in regions with a low proportion of conservation forests compared with the global average (e.g., a 3.3 % increase in East Asia) but also in Europe (3.9 %, excluding the Russian Federation) and in South America (4.8 %). Most regions have already set aside 10 to 20 % of their forest for biodiversity conservation purposes (FAO 2011). The region with the highest proportion of conservation forest is Central America

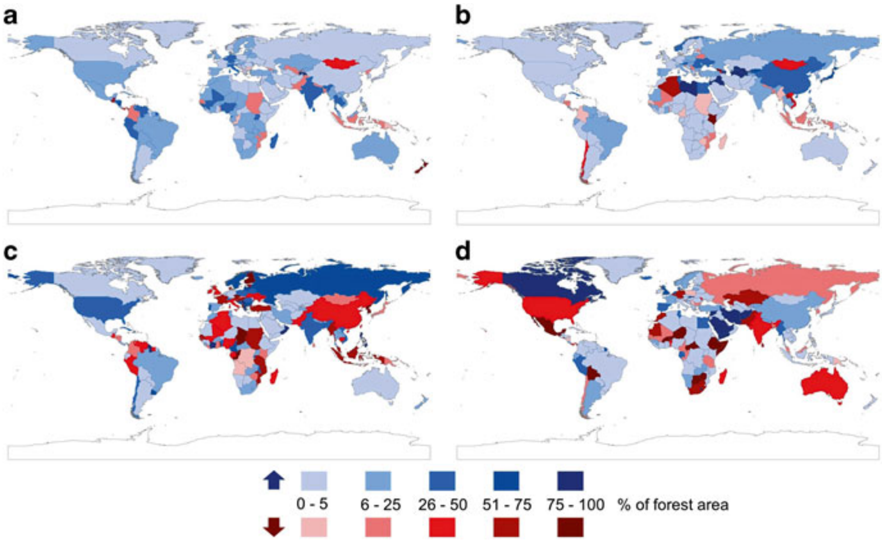

Figure 5.3 Proportions of the forest area designated for (**a**) biodiversity conservation, (**b**) water and soil protection, (**c**) production of forest products, and (**d**) multiple uses in 2010, and recent trends. Proportions are indicated by the color gradient, with darker tones indicating higher proportions. Trends are indicated by color, with *blue* indicating an increase in the designated area from 2000 to 2010 and *red* indicating a decrease. Note that the proportion does not indicate the extent (area) of forest in a country, and that the trends do not indicate the rate of change. *Source*: FAO (2011)

(47 %), whereas East Asia, West and Central Asia, and the Russian Federation have designated less than 10 % of their forest for conservation. Globally, 463×10^6 ha of forest have been primarily dedicated to biodiversity conservation (Fig. 5.3a).

Nevertheless, protective forests are now emerging as a tool to conserve ecosystem functioning and manage the delivery of regulating services. For instance, several Asian countries have reported a high proportion of protective forest (Fig. 5.3b). This is particularly the case for China, where a large area of forest has been planted with the main purpose of controlling desertification (Cao et al. 2011), and for several western Asian countries in arid zones, such as Turkmenistan and Uzbekistan, where water is a critical resource (FAO 2011). The same pattern is found for African countries in arid zones, such as Libya and Kenya.

Globally, the area of forest designated primarily for productive functions (Fig. 5.3c) has decreased at an annual rate of 0.2 % from 2000 to 2010, and currently covers 1200×10^6 ha (FAO 2011). This reduction in area is in part explained by the increase in the area of forest dedicated to intensive forestry but also because some areas previously classified as productive forests were reclassified as multiple-use forests (FAO 2011). Europe is the region that has reported the largest proportion of areas designated primarily for productive functions (52 % or 57 % excluding the Russian Federation), and North and South America have reported the lowest proportions

(14 %, but with heterogeneity within the region; for example, 1 % in Canada compared with 29 % in the United States) (FAO 2011). Despite the global decrease in the area designated for productive functions, the pattern is heterogeneous, showing a pattern of interspersed areas with increasing, decreasing, and stable trends.

In addition to areas specifically designated for productive purposes, many forest products are obtained illegally or informally from areas that are not classified as productive. This implies that the real area used for the extraction of forest products is much larger than the area that is formally designated as productive forest. Moreover, multiple-use forests (Fig. 5.3d) also encompass productive functions. Currently, the area designated for multiple uses totals 949×10^6 ha globally, and increased by 10×10^6 ha between 1990 and 2010 (FAO 2011). Global and regional trends are heterogeneous, reflecting different types of transitions, including shifts in the classification from productive to multiple use and vice versa but also shifts from undesignated to multiple use.

Social forests, which are primarily used for recreation, environmental education, and preservation of cultural heritage, are still infrequent, despite the widespread use of forests for outdoor activities and their cultural role as natural heritage sites. The social function of forests is usually associated with conservation forests or with multifunctional forests. Today, only 3.7 % of the world's forests are designated primarily for this purpose, but available data suggests that this proportion is increasing (FAO 2011). Brazil has the largest area of social forest, at 119×10^6 ha (i.e., more than 75 % of the global area of social forest), and this forest is designated for the protection of indigenous peoples and their culture.

The statistical data used in this section was reported by individual countries and gathered together for the Global Forest Resources Assessment 2010 (FAO 2011), which is the most comprehensive assessment to date. However, two main sources of uncertainty should be considered when comparing countries or regions. First, there are disparities among the reporting countries in terms of data availability, either due to real data gaps or due to differences in national forest inventory methodologies. Second, the criteria used to define forest categories and functions are subject to different interpretations by the reporting countries. Also note that the designation of a forest for a particular purpose does not imply the existence of sustainable management practices or even of a management plan for that forest.

5.3.3 Drivers of change and impacts on forest ecosystem services

The Millennium Ecosystem Assessment identified five main direct drivers of biodiversity and ecosystem change (MEA 2005): habitat change, climate change, invasive species, overexploitation, and pollution. The impacts of these drivers and their trends vary across the globe, and affect forest biomes differently. Direct drivers are often shaped by social demand for provisioning services, including both forest provisioning services and farmland services when agriculture replaces forest use.

For example, population growth can cause a higher demand for food and fiber and can therefore lead to production activities that cause deforestation. On the other hand, global policies for climate mitigation can encourage forest conservation, and technological advances can improve the efficiency of forestry and agricultural production, thereby lessening the pressure on natural forests.

Pollution became an important driver in the last century in many forms, and particularly in the forms of excessive nutrient loading in production systems and of industrial emissions (Shvidenko et al. 2005). Boreal forests have been particularly badly affected by air pollution from industrial sources during the last century, with reported events of significant tree damage and mortality (Shvidenko et al. 2005). When combined with climate change, pollution is expected to have a serious impact on the condition of these forests during the twenty-first century. Climate change not only will affect tree physiology and phenology but will also affect the fire regime by increasing the frequency and severity of wildfires as a consequence of drier and hotter summers (Soja et al. 2007, Stocks et al. 1998).

Habitat change and overexploitation were the main drivers of forest change in temperate regions during the last century. Today, the effect of these drivers is declining as new forests are planted and regenerate in abandoned fields and pastures. On the other hand, tropical forests have been particularly affected by land-use change during the last century, and the impact of this driver is expected to increase in the twenty-first century as forests are replaced by pasture and cropland (in part to respond to international demand for food) but also by infrastructure and urban areas. Overexploitation of forest goods is also expected to intensify due to population growth in these regions, as well as logging driven by international demand (Davidson et al. 2012, Lambin et al. 2003). Overall, the impacts of climate change will increase during the twenty-first century in all biomes (Leadley et al. 2010, MEA 2005). The impact of invasive species is also expected to increase due to global trade and travel, as well as the impact of pollution, in particular due to a significant intensification in the flow of reactive nitrogen into the environment (MEA 2005).

The effects of drivers are often synergistic. Changes caused by a driver or by a set of drivers may create the conditions for triggering, intensifying, or maintaining other drivers, rendering the control of their impacts difficult (Lambin et al. 2003, Leadley et al. 2010). In some situations, these interactions lead to regime shifts, with strong impacts on ecosystem structure and functioning. Although researchers can identify tipping-point changes and their potential risks, their dynamics are complex and difficult to predict (Leadley et al. 2010). Tipping points can be broadly defined as events that occur when an ecological threshold is passed, leading to shifts in ecosystem functioning that significantly affect biodiversity and ecosystem services. Tipping-point changes tend to be fast due to reinforcing feedbacks that amplify the effects of drivers or due to abrupt shifts when thresholds are crossed. They also tend to be difficult to reverse due to feedback loops that trap systems in undesirable stable states and long lag times between a driver's action and its impacts, which hamper policy decisions (Leadley et al. 2010, in press). The Amazonian, Mediterranean, and boreal forests present important examples of potential regime shifts in forest systems.

In the Amazon basin, forest conversion coupled with climatic changes may lead to a regime shift that will have impacts from local to global scales (Davidson et al. 2012, Nobre et al. 2010). Deforestation, logging, and forest fire are inducing regional climate changes, including less rainfall and increased frequency and severity of drought, which increase the susceptibility of forests to fire, thereby creating a feedback loop that sustains fire occurrence, promoting further forest damage and fragmentation. In addition, projections from climate models indicate that long-term global climate change will amplify drought in the Amazon region due to a combination of climate warming and less precipitation. At moderate to high rates of deforestation, the interaction between land-use change, fire, and climate change may lead to a feedback loop that will be difficult to control and that may cause extensive forest loss (Davidson et al. 2012, Vergara and Scholz 2011). Carbon release due to this deforestation will also contribute to global climate change and will aggravate climate-change impacts at the regional scale.

Consequences for ecosystem services and biodiversity will be severe. The Amazon is one of the world's largest carbon pools and carbon sinks, with the exception of dry years, when forests becomes a carbon source (Davidson et al. 2012, Phillips et al. 2009). The shift from a carbon sink to a carbon source will contribute to global warming and cause negative impacts at a global level. The Amazon is also a biodiversity center (Pereira et al. 2012), and loss of Amazonian forest will result in a severe loss of biodiversity at a global scale. In addition, there is the risk of losing species, many still unknown, that have medical and pharmacological value, and consequently a risk of losing the opportunity to find and develop new medications and vaccines. At local and regional scales, local communities will be affected by the loss of forest goods, including food, fiber, fuel, and medicinal plants; by the loss of regulating services, such as climate regulation, fire regulation, and flood regulation; and by the loss of cultural services, since the forest environment is a major component of the cultural heritage and way of life of local peoples.

In the Mediterranean region of southern Europe, land-use change, fire disturbance, and climate change are interacting to create conditions suitable for a shift in ecosystem composition (Proença and Pereira 2010). Rural abandonment is driving land-use change in marginal areas of farmland, through the regeneration and encroachment of natural vegetation and the expansion of fire-prone forest plantations, thus promoting fuel continuity in the landscape. The accumulation of biomass coupled with frequent (anthropogenic) fire ignition is causing a change in the fire regime, with more frequent and severe fires. This situation is further aggravated by climate change, in particular by hotter and drier summers. All these factors cause an increase in fire risk and promote the expansion of fire-prone communities, such as shrublands, which then create the conditions for the establishment of a feedback loop that inhibits the progression of natural succession towards regeneration of natural forests. Under some circumstances, alien invasive species gain competitive advantages in the burned areas, letting them replace native species and impoverishing natural communities (Keeley et al. 2003, 2005). This may eventually lead to a compositional shift that will be hard to reverse.

Boreal forests provide a third example of regime shifts. In this case, climate change is leading to a warming trend that is moving northward, creating an environment unsuitable for boreal species. On the one hand, these species may not be able to respond to this change because their natural rate of propagation is too slow for them to migrate north and also because tundra sites into which the boreal species will be forced to migrate might be unsuitable for their establishment and growth (Lloyd et al. 2011, Soja 2007). Chapter 2 of this book discusses these issues in more detail. But on the other hand, where tundra sites are suitable for boreal species, changes in soil albedo, the melting of snow cover, and retreating permafrost will allow tree establishment and forest invasion into the tundra (Soja 2007). The main mechanism underlying this change is an amplifier feedback loop driven by increasing summer temperatures (Fernandez-Manjarrés and Leadley 2010). Warmer temperatures lead to earlier melting of the snow cover, exposing soil with a lower albedo that traps more solar radiation over a larger period. Snow cover creates an insulating effect that is critical for the maintenance of permafrost; the loss of snow cover causes permafrost degradation and increases the warming effect, thereby promoting further snow melting. These changes will have impacts on the lives of local people, who will have to adapt to a changing landscape (e.g., travel routes may become unsafe due to decreasing ice stability), but will also have impacts at a global level due to the release of large quantities of carbon and methane stored in the permafrost, with consequences for climate change (Fernandez-Manjarrés and Leadley 2010, Schaefer et al. 2012). Moreover, some parts of the boreal forest are being increasingly affected by fires driven by climate change, which also causes the release of carbon stored in the trees and soil in addition to the loss of old-growth forest and other social and ecological losses (Soja et al. 2007).

As the demand for non-provisioning services (e.g., soil protection from erosion, water purification, recreation) increases, it counteracts the economic bias towards provisioning services. Direct drivers will be gradually affected by indirect drivers in response to this demand. This may include national to global policies but will also include changes in markets. In the past, and to a certain extent, still today, economic choices tended to disregard non-market services and promote the expansion and overexploitation of productive forests or the replacement of forest by more profitable land uses. The incorporation of the benefits delivered by non-provisioning services in economic choices is likely to reshape market demand and its effect on the direct drivers of change, thereby promoting forest conservation and restoration to preserve forest's regulating services and cultural value.

5.4 Economic valuation of forest ecosystem services

Forests can provide multiple benefits to society other than wood, with the whole array of benefits depending on the characteristics of the forest and the prevailing management strategies (Duncker et al. 2012). This understanding is a prominent feature of the current literature and is usually associated with the concept of multifunctional forests (e.g., Carvalho-Ribeiro et al. 2010, Gustafsson et al. 2012). One possible approach to capture the contribution of forest ecosystems to humans is through an improved understanding of the linkage between the functioning of the ecological system, which is perceived as a composite of processes and structures, and the functioning of the socioeconomic system. The crucial role that natural systems play in underpinning economic activity and human well-being is of growing concern (Bateman et al. 2010). Thus, economic valuation of ecosystems and their services has been receiving increasing attention in the literature.

Economic valuation is not the only approach to assigning a value to nature, nor is it necessarily the best approach; for examples of other forms of valuation, see Oksanen (1997), Martín López et al. (2012), and TEEB (2010). As Kareiva et al. (2011) pointed out, it is important to emphasize that an economic valuation does not replace or ignore the intrinsic value of nature, nor does it reduce the moral imperative to conserve nature. Following the logic of Martín-López et al. (2009) and Mace and Bateman (2011), we note the importance of combining economic and other valuation approaches to provide a more holistic picture of the value of forests. Nonetheless, in this chapter we will focus on the economic valuation approach. The primary role of economic analysis is to assist decision-making (Daily et al. 2000, Pearce et al. 1989, Tietenberg 1996). In the context of forest management, the high rate of deforestation we are facing globally—13×10^6 ha per year (FAO 2007)—and the rise of international concern about the consequences of deforestation together mean that economic valuation of forest ecosystem services has an important role to play.

Before jumping into the principles and methodological details of economic valuation, we will briefly illustrate how economic valuation of a forest ecosystem can restrain deforestation. As we noted earlier, forests provide many non-market goods, such as watershed protection. Landowners seek profit maximization, and in the absence of other mechanisms, they rely on existing markets to pursue this goal. Existing markets define their costs and revenues. Hence, even though we know that clearing the forest would increase problems such as downstream flooding and sedimentation, these costs do not accrue to the landowner who will decide whether to harvest the forest; thus, these costs are not factored into the landowner's decision. This is clearly a market failure from a larger perspective. Economic valuation can mitigate this problem if the analysis allows for an extended accounting of benefits and costs and, based on this more complete picture, fosters mechanisms such as subsidies, taxes, direct payments, and payments for ecosystem services that can prevent the market failure and reduce the likelihood of deforestation. For a concise review of market-based mechanisms, see Pagiola et al. (2002). These mechanisms aim to fully internalize the benefits and costs that do not accrue directly to landowners but rather that affect other groups in society. In Sect. 4.1, we further explain the occurrence of externalities and market failure from a conceptual point of view.

At this point, and before we begin discussing the principles of economic valuation, we want to emphasize that the value of forest ecosystem services reflects the different ways in which they satisfy human needs. This can be considered from the perspective of the total economic value (TEV) taxonomy (Pearce 1993).

This taxonomy defines the different sources of values that people may attach to the different services provided by a given ecosystem. Note that this taxonomy relies on whether ecosystem services satisfy human needs directly or indirectly. Economic value, then, is a measure of the degree of satisfaction provided by these services. The TEV approach and terminology are not uniform across the literature, but TEV generally includes the following value components: direct use, indirect use, option, and non-use. The first three categories are generally referred to together as use values, and the non-use values often aggregate values such as bequest and existence values.

Among the use values, direct use values include services that are used directly, and include provisioning services (e.g., forest goods) and cultural services (e.g., recreation opportunities). Indirect use values include services that are indirectly used, such as the benefits derived from regulating services (e.g., climate regulation). Non-use values are divided into bequest and existence values, and are almost entirely associated with cultural services. Bequest values represent the value that an individual assigns to an ecosystem or species due to its relevance to the well-being of future generations. Existence value, on the other hand, represents the value that an individual assigns to an ecosystem or species due to its personal relevance at the present time. In other words, it is the satisfaction this individual derives from knowing that a certain species or ecosystem exists.

Option values include all values (both use and non-use) that are expected to be enjoyed in the future (e.g., provision of genetic resources, maintenance of a gene pool for bioprospecting, cultural heritage). Note that the option and bequest values both reflect the importance that people give to maintaining or restoring ecosystems in order to ensure the delivery of ecosystem services in the future.

5.4.1 Principles of economic value estimation

The economic value of an ecosystem service refers to the contribution of a certain ecosystem functional dynamic to human well-being. Many ecosystem services are only obtained because of other capital inputs; for instance, agricultural production of food implies the use of machinery and labor together with the use of natural resources and ecosystem processes. Hence, as pointed out by Bateman et al. (2010), estimating the economic value of ecosystem services requires isolation of the ecosystem function's contribution before the value can be converted into a monetary metric. This suggests that it is also necessary to clarify how economic analysis differs from financial analysis: the former examines society as a whole, whereas the latter focuses on particular groups within society. Hence, when estimating the economic value of an ecosystem service, we must account for the costs (private and external) of producing the service and for the benefits (private and external) generated by it. Here, "external" refers to externalities, whether benefits or costs, that are generated as unintended by-products of an economic activity that do not accrue to the parties involved in the activity, and for which no compensation is provided.

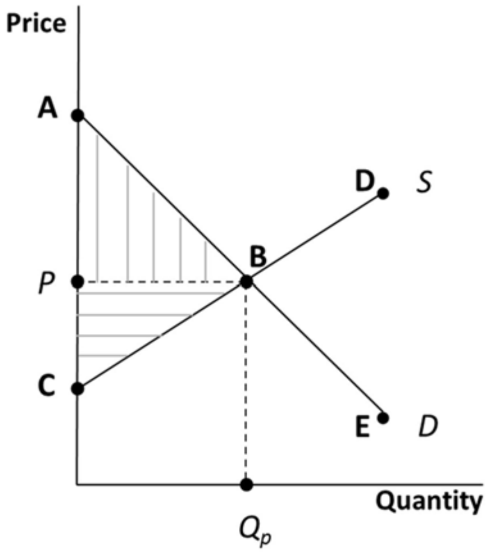

Figure 5.4 Social surplus [*ABC*] for a forest good such as timber under perfect market conditions. *D* demand curve, *S* supply curve, *P* price where $D = S$ at point *B*, Q_p the quantity at price *P*

Depending on its impact on a third party, an externality may be positive (e.g., the creation of a forest landscape) or negative (e.g., the creation of fragmentation).

We first approach the economic foundation of ecosystem service valuation by considering a well-defined market in which ecosystem services can be traded and in which there are no external costs or benefits. There are two building blocks in the process of estimating economic value: consumer and producer surpluses, with the social surplus equaling the sum of these two surpluses. See Mankiw (2008) for a discussion of this topic. These measures are illustrated in Figure 5.1 for the case of an ecosystem service for which there is a market, such as timber (a forest provisioning service), based on the assumption of a perfectly competitive market. The timber market is in equilibrium when demand (*D*) equals supply (*S*) at price *P*. The demand curve shows consumer marginal willingness to pay (WTP), which represents the consumer's WTP for each additional unit of a product. The supply curve shows the marginal costs of harvesting timber, which represents the producer's marginal willingness to accept (WTA) a given price for their product.

Figure 5.4 tells us that buyers who value the good more than the price (represented by the line segment *AB*) choose to buy the good and receive a surplus of benefit: the area of the triangle *ABP* defines the magnitude of the consumer surplus. This represents the amount a buyer is willing to pay for a good, minus the amount the buyer actually pays for it, or, in different words, the benefit that buyers receive from participating in the market. Buyers who value the good less than the price (represented by the line segment *BE*) choose not to buy the good or receive its benefits. Symmetrically, on the production side, those sellers whose costs are less than the price (represented by the line segment *CB*) choose to produce (in this case, to harvest) and then sell the good (wood). Sellers receive a surplus given by the area of the triangle *PCB*; this represents the amount a seller is paid, minus the cost of production. The producer's surplus measures the benefit sellers receive from participating in the market. Sellers whose costs are greater than the price (represented by the line segment *BD*) do not sell the good or receive benefits from the sale. The social benefit (i.e., the overall surplus) in this case equals the private benefit, which equals the sum of the consumer and producer surpluses (i.e., the area of triangle *ABC*). The social surplus is of interest in economic analysis because it concerns the net benefits that society as a whole derives from the good. Mathematically, the total or social surplus can be expressed as follows:

$$\text{Social surplus} = \begin{bmatrix} (\text{value to buyers} - \text{amount paid by buyers}) \\ +(\text{amount received by sellers} - \text{costs beared by sellers}) \end{bmatrix}$$

Thus far, we have analyzed a situation in which the social benefit equals the private benefits. However, when there are external costs and benefits, the private surpluses do not equal the social surpluses. Let's again consider a timber market, but in the presence of external benefits, based on the example provided by Hanley and Barbier (2009). Consider a sustainable timber harvester, with sustainability here defined as a state of non-declining well-being, as defined by Tietenberg (1996; pp. 33–34). This harvester manages their land in a wildlife-friendly manner, thereby improving the ecological quality of their woods and overall forest health by (among other things) creating many habitats for birds and butterflies. They also harvest timber for sale. The market rewards them for their timber production, since they can sell the timber to interested buyers. But the market is unlikely to reward them for their "production" of wildlife habitats, even though these habitats might be valued by society. Although this is not the forum for further discussion of this topic, these types of services fall into the category of public goods (non-excludable and non-rival in consumption). See Boardman et al. (2001) for further explanation. In Figure 5.5, D_m is the market demand curve for timber, *S* is the supply curve for

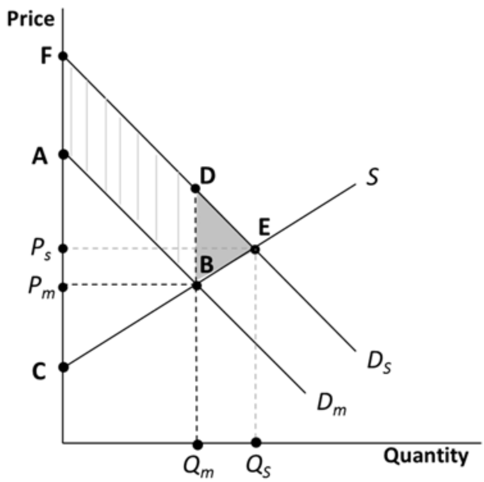

Figure 5.5 Social surplus (*ABDF*) for a good (e.g., timber) in the presence of a positive external effect (e.g., provision of wildlife habitat). *S* supply curve, D_m market demand for timber, D_s society's demand for timber plus its external benefits, Q_m the quantity at price P_m, Q_s the quantity at price P_s. *Grey area* (*BDE*) represents welfare lost under perfect market conditions (e.g., without government intervention)

timber, and D_s is society's demand for timber plus its external benefits (in this example, "production" of wildlife habitats). The market reaches equilibrium at point *B*. At this point, both timber consumers and the rest of society receive a benefit (an external benefit whose magnitude equals the vertical distance between D_m and D_s) which is represented by the line segment *BD*. The social surplus obtained when quantity Q_m is produced therefore equals the area of the polygon *ABDF*. Notwithstanding, the social optimum would be reached at point *E*, where the marginal social cost (which, as no negative externality is being considered, is the same as the marginal private cost) is equal to the marginal social benefit. Under perfect market conditions (e.g., without government intervention), quantity Q_s will not be supplied because harvesters are not rewarded for producing such quantity. The area of triangle *BDE* (grey shade) represents the welfare loss under these market conditions. Many ecosystem services are externalities, in the sense that the benefit or cost they represent to society is generated as a consequence of standard ecosystem management but it is not intentionally produced, and it does not accrue benefits or costs to the producer. This means that, for instance, the value of timber does not reflect the array of benefits that may be jointly provided by forests to society as a whole. Often, in the literature, these externalities are referred to as non-market ecosystem services. As we discuss in the next section, several methods have been developed to estimate the value of such ecosystem services.

5.4.2 Economic valuation methods

Our purpose is not to fully review all the available valuation methods, but rather to provide a concise overview of such methods while illustrating the objective and context of their application. In the previous section, we focused on consumer and producer surpluses as the measures of interest and explained how these measures relate to WTP and WTA. Bearing in mind that these are the measures of interest, we should also note that the focus of economic valuation is to estimate such measures for a well-defined change. This implies estimating the changes in the consumer and producer surpluses and the change in their sum (Freeman 2003) by considering changes in the welfare of both consumers and producers.

The methods used to value ecosystem services can be grouped into three main categories: direct market valuation approaches, revealed preferences approaches, and stated preferences approaches. Direct market valuation approaches rely on the use of data that can be readily obtained from existing markets (such as prices, demanded quantities, and production costs), and include three main approaches: approaches based on market prices, costs, and production functions. Market price approaches rely on the use of market prices as a proxy for value. Although this appears to be the most straightforward approach, there are several aspects that should be emphasized about its application. Under the general case of perfect competitive markets, prices are defined by the interaction of supply and demand; as a result, prices are acceptable or starting point approximations of the marginal value.

If this holds true, and the change being analyzed is sufficiently small that prices remain constant, application of the method is straightforward: we just multiply the change in the number of units (for instance, the increase or decrease of the available m^3 of water) by the associated marginal price. When the changes are large enough to change prices, then the changes in consumer and producer surplus must be estimated. Even if prices can be taken as a proxy for the marginal value, price distortions created by subsidies and taxes should be taken into account; the cost of making the good available should be subtracted from the price in some cases, since labor and transportation costs involved in making the benefit available represent opportunity costs that could be transferred to generate alternative goods and values; in addition, prices generated by supply and demand reflect scarcity, not value, as is often illustrated using the relative prices of water and diamonds (a paradox originally posed by Adam Smith), since water is vital to support life (unlike diamonds) but because it is generally abundant, it is cheaper than diamonds.

Cost-based approaches include the avoided cost, replacement cost, and mitigation or restoration cost methods, and are used to estimate the costs that would be incurred to artificially provide the benefit instead of using ecosystem services. In the context of forest ecosystem services valuation, the avoided cost method could be used (for example) to estimate the value of flooding protection provided by a forest based on the costs of building protection infrastructures to generate the same benefit; for other applications of the avoided cost method, see Nowak et al. (2006) and van Kooten (2007). The replacement cost method could be applied (for example) to estimate the value of soil protection based on the costs to restore the storage capacity of downstream dams after siltation of the reservoir. For other applications of this method, see Chopra and Kumar (2004) and Rodríguez et al. (2007). The restoration costs may, for instance, be useful in determining the value of water purification or infiltration based on the investments made to reverse degradation of the service. For other examples of the restoration cost approach, see Birch et al. (2010).

The last of the approaches based on market valuation is the production function method, which Barbier (2007) referred to as "valuing the environment as input". Behind the method's application is the idea that several ecosystem services (e.g., regulation services, biodiversity) enhance the production of market goods. Hence, if changes in these services affect the marketed product, then the effects of these changes will be visible through the price system. For instance, if the purification capacity for water decreases and this generates additional costs for the producers of bottled water, then the price of the water would increase. An example of this method in the context of forest ecosystem service valuation is provided by Nahuelhual et al. (2006).

In the revealed preferences approach, the main methods are the travel cost and hedonic pricing methods. These methods use consumption behavior in markets that are related to the non-market goods and that therefore serve as proxies for those goods. The travel cost method is the most commonly used method, and has been widely applied to infer the value of forests for recreation (e.g., Badola et al. 2010, Bowker et al. 2007). This method uses visitation rates and the distance traveled to infer the demand for such a benefit. The observed variation in visitation rates and travel costs (used as a proxy of price) describes the changes in demand for the site, and the demand function allows researchers to determine the consumer surplus. The hedonic pricing method uses the differences in the price of a benefit that reflect its inherent properties to infer the value of non-market attributes of ecosystems. A recent application of the method was provided by Sander and Haight (2012).

In the stated preferences approach, contingent valuation is the most well-known method. This method involves directly asking a representative sample of a population to define their WTP and their WTA for a well-defined change in the provision of a certain ecosystem service (for instance, a change in water quality). Researchers can then use compensating variation or equivalent variation to estimate the economic value. Both are exact welfare measures, and may not be identical to the consumer surplus for market goods. Instead, these measures estimate the change in income that is needed to maintain a certain level of utility (welfare, satisfaction). Note that along an ordinary demand curve, utility is not constant if income is kept constant. For further explanation of these measures, see Freeman (2003) and Zerbe and Bellas (2006). Choice modeling is a questionnaire-based method that gained relevance with practitioners of economic valuation of ecosystem services. The method consists of presenting individuals with two or more alternatives defined by a set of attributes regarding the ecosystem services under valuation, and it is designed to elicit the WTP for having that alternative. The levels of the set of attributes vary among the alternative sets that individuals must choose among or rank. Both methods have been applied to estimate the value of several forest ecosystem services. For examples of contingent valuation applications, see Sattout et al. (2007) and Barrio and Loureiro (2010); for examples of choice modeling, see Rolfe et al. (2000) and Brey et al. (2007). Individual-based questionnaires aggregated to represent a socially relevant unit (e.g., a community) might be appropriate when the services being valued are purely enjoyed on an individual level (e.g., valuing forests for timber), but have limited applicability in the cases of more communal services. For example, the value of forests to a community whose social system is intimately dependent on them is more than the sum of the independent personal values (Farber et al. 2002). Hence, another stated preferences method, group valuation, is gaining relevance. Although a stated preferences method, its focus is not on valuing individual preferences but rather on collecting social preferences. Wilson and Howarth (2002) and Chan et al. (2012) provide a detailed discussion of this method.

5.4.3 Challenges in estimating economic value

Estimating the economic value of ecosystem services faces several challenges, and regardless of the objective of the economic valuation, whether to inform macroeconomic policies or to evaluate programs (Bateman et al. 2010), estimation of flows of ecosystem services is often necessary. A flow estimation is usually an estimate of money per unit area obtained for a certain period, usually on an annual basis. Although flow estimations provide valuable information, they are not, per se, relevant to inform land-use decisions because few interventions would result in an entire loss of the flows of ecosystem services. Instead, management often results in incremental small changes. What is needed is an understanding of how land-use changes would affect societal well-being, so the focus of the economic valuation is on valuing the incremental or marginal changes in the flows of services. This is often done by means of scenario analysis, in which researchers compare the consequences of two or more scenarios. Valuing such changes implies a deep understanding of the ecological dynamics of the system and how the system responds to perturbation.

Though the economic valuation approach is remarkably valuable because of its ability to provide more objective comparisons of alternatives, it has not yet overcome significant challenges to tackling such complexity (Robertson 2011). This problem has been pointed out by several authors under the headings of uncertainty, ecological thresholds, and irreversibility (Morse-Jones et al. 2011), and in the contexts of weak or strong sustainability (Olschewski and Klein, 2011). Because the valuation focuses on estimates of marginal changes, caution is needed with the valuation itself because the marginal value may not be constant. This is clearly illustrated by the example provided by Bateman et al. (2010), who examined the recreational value of an urban green space (a park). They found that increasing the area of this space altered the recreational marginal value, with the first increases in area being highly valued, but subsequent increases becoming less valued.

There are other problems related to the assumption of a constant marginal value that suggest a need for caution. Ecosystems and their services are not spatially homogeneous and thus may not provide the same flow throughout the system's spatial extent (Fisher et al. 2009). Moreover, even when ecosystems provide the same flow of services from different areas, the marginal values of these flows may not be the same. We can illustrate this again using the value of a green space for recreation. An urban forest area of a given size may have a higher recreation value when it is near an urban area than when it lies in a region that is not accessible to urban residents. The issue of spatial variability of ecosystems and ecosystem services suggests the need to perform economic valuation on a spatially explicit basis. In addition, the effect of scale is a challenging topic that has not been fully tackled. This affects the discussion of ecosystem services valuation because the scale at which benefits might be provided ranges from local to global. For example, a forest might provide recreational opportunities (local), downstream flood prevention (regional), and climate regulation (global) when considered from the supply side. This variability also holds for the demand side. For instance, endemic species may have beneficiaries very far from the location of their occurrence (as in the case of residents of developed nations placing value on endangered species in developing nations). The issue of scale has been extensively debated (EEA 2010, Hein et al. 2006), and the advantages of spatial analysis in tackling the issue are making scale-explicit analyses increasingly relevant. Failing to properly address the issue of scale may complicate or bias the design of ecosystem services payments, which is an emerging mechanism to ensure the provision of non-market ecosystem services.

5.4.4 Economic estimates of forest ecosystem services around the world

As we have stressed in the previous sections of this chapter, awareness of the importance of forest ecosystems and the vulnerability of their valuable services is increasing around the world. This awareness is ultimately at the base of recent economic valuation efforts that targeted forest ecosystem services. In this section, we provide an overview of recent studies in which forest ecosystems from around the world were monetarily valued.

Previously, we briefly introduced the concept of TEV as a commonly used taxonomy to determine the aggregate economic value of all the benefits people obtain from ecosystems. Several researchers have attempted to obtain the TEV of forests (e.g., Adger et al. 1994, Merlo and Croitoru 2005, Pearce 2001, Thompson et al. 2011) and Ferraro et al. (2012) provide a review of this topic.

Targeting specific areas for the estimation of TEV is a common practice. One recent attempt to assess TEV focused on the Hoge Veluwe Park (Hein 2011), a protected area in the Netherlands. Hein estimated the economic value of ecosystem services provided by the park's more than 5000 ha of pine and deciduous forests to be around 10.7 million € per year, of which 2.1 million € per year were due to air pollution removal and 1.9 million € per year were due to groundwater infiltration, for example. By combining land-cover mapping with benefit-transfer calculations, Vorra and Barg (2008) estimated the aggregate economic value of ecosystem services provided by the Canadian UNESCO World Heritage Site of Pimachiowin Aki to be approximately C$130 million per year, mostly due to its provision of pure water and fish. Ingraham and Foster (2008) used a similar approach, but their goal was even more ambitious: to determine the aggregate economic value of the ecosystem services provided by a large network of protected forests (the U.S. National Wildlife Refuge System) in the 48 contiguous states, which turned out to be around $26.9 billion per year. Although this was a first approximation, their research highlighted the need for further and more rigorous examination of the value of ecosystem services to assist management and policy decision-making, and their results emphasized that the TEV of protected areas exceeds that of its pure recreational value.

Bolder initiatives to assess the aggregate monetary value of global forest ecosystem services have also taken place in recent years. By using ad hoc value-transfer protocols, Chiabai et al. (2009) estimated the economic value of a comprehensive set of ecosystem services (timber and non-timber forest products, carbon storage, and recreation and tourism) for all forest biomes around the world. Benefit (or value)-transfer uses economic information captured at one place and time to make inferences about the economic value of environmental goods and services at another place and time, as discussed by Wilson and Hoehn (2006). An interesting aspect of this research is that it also presents potential estimates of total economic losses by the year 2050 due to policy inaction. They identified major economic losses of 78 billion €, mostly due to the loss of the forests' provisioning and carbon sequestration services. Their results underlined current production scenarios; for example, they estimated the marginal value of provisioning services from tropical forests in Africa at around US$1800 per ha per year as a result of the high production value of fuelwood in Africa.

Even though valuing nature as a whole through the TEV taxonomy might be pertinent for current conservation agendas, valuation studies are most commonly performed on a case study basis, with a particular ecosystem service or a particular subset of services being targeted. Focusing on a particular ecosystem service might help to solve specific challenges and support policy development at a relatively small scale, as it is a far simpler task than estimating TEV and may provide more practical outputs. For example, to provide insights into the protection of woodlands as a climate-change mitigation measure, Brainard et al. (2009) performed a cost–benefit analysis to assess the value of the carbon sequestration services provided by the woodlands of Great Britain. Their results, which depended strongly on the discount rate and the social value of sequestered carbon being considered, ranged from US$82 to US$853 million annually. The variation of estimations due to the application of different discount rates is not uncommon, and imposes an additional challenge for the valuation of nature (Freeman and Groom 2013). In fact, the choice of the discount rate to be applied in environmental appraisal is not exempt from problems and criticisms, and there is no single discount rate to be applied, so sensitivity analysis may be generally performed for a range of discount rates (Boardman et al. 2001).

Appraising the value of the carbon sequestration services of forests as part of the tactics for mitigating human-origin CO_2 emissions has been receiving increasing attention in response to growing awareness of climate change. In Brazil, Guitart and Rodriguez (2010) have assessed the value of potential carbon sequestration services provided by two commercial eucalyptus plantations. Based on their results, they suggested a minimum annuity of US$18.8 per ton of stored carbon to be paid to the owner of the forests in order to stimulate and justify the adoption of silvicultural regimes that would increase carbon sequestration.

Though globally relevant nowadays, the ability to sequester carbon is not the only valuable service that forests can provide. Olschewski et al. (2012) determined the WTP for the avalanche protection services that the forests of the Swiss Alps provide to the population of the Swiss municipality of Andermatt, in Canton Uri. Their results ranged from US$20 per household (a one-time payment due to avoidance costs) to more than US$300 per household (a one-time payment due to risk reduction). In Africa, the risk of snow avalanches is obviously not a primary concern. Instead, Schaafsma et al. (2012) estimated the total flow of benefits from charcoal production in the tropical forests of southern Kenya to be worth around US$14 million per year, as charcoal provides an important source of income to local households and supplies around 11 % of the charcoal used in the major cities of Kenya and Tanzania. For timber forest products, Ojea et al. (2012) estimated the potential value of wood provision in sustainably harvested Mediterranean forests in Spain at around 500 € per ha per year based on the sustainable provision of timber over 30 years at a discount rate of 2 %. Their results indicated that, in some cases, non-sustainable forests provided higher returns over shorter time spans, but that sustainable harvesting provided the highest overall returns over long terms. Regardless of the ecosystem service targeted, studies at a regional level bring a very policy-oriented perspective into the valuation exercise.

5.5 Initiatives and policy responses

Today, most large-scale initiatives and agreements regarding forest management revolve around the concept of sustainable forest management to prevent forest degradation and to promote the development of multifunctional forest systems.

The REDD+ mechanism (*Reducing emissions from deforestation and forest degradation in developing countries and the role of conservation, sustainable management of forests and enhancement of forest carbon stocks in developing countries*) is currently the most promising tool designed to support the conservation of forests, with a particular emphasis on carbon-regulating services (FAO 2012, http://www.un-redd.org/). The mechanism was established under the United Nations Framework Convention on Climate Change (http://unfccc.int/2860.php) and included in the global climate-change agenda in 2007 that was defined at the climate summit in Bali (Angelsen and Rudel 2013). The mechanism has been implemented through several initiatives, such as the UN-REDD program (http://www.un-redd.org/) or the Forest Carbon Partnership Facility hosted by the World Bank (http://www.forestcarbonpartnership.org/). REDD+ is a financial mechanism designed to reduce carbon emissions caused by forest losses and degradation in developing countries while at the same time creating conditions for sustainable forest management and promoting sustainable development programs in the participating countries (Angelsen et al. 2012, IUCN 2009). In brief, the underpinning idea is to implement payment for ecosystem services (PES) schemes in which the international community pays forest users in developing countries to adopt policies and programs aimed at conserving forests, improving forest stocks, and reducing forest degradation (Angelsen et al. 2012).

Despite the overall support for this mechanism, and its ongoing application in several countries, it has also been criticized based on several important issues. These include the need to guarantee sustainable sources of funding, the difficulty in monitoring the outcomes of implemented projects, and more importantly, the lack of a clear understanding and a legal framework for land tenure and carbon rights in many countries, which can be a barrier to the implementation of PES schemes, particularly if this promotes inequity and disregard for the rights of forest communities and indigenous peoples (Angelsen et al. 2012, FAO 2012). In addition, there is some apprehension concerning the subordinate role of biodiversity in relation to carbon storage and sequestration, which constitute the main focus of REDD+. Unclear targets for biodiversity and other ecosystem services may allow the occurrence of trade-offs instead of achieving the envisioned synergies, which are expected to arise as positive externalities from activities directed towards carbon storage and sequestration (Visseren-Hamakers et al. 2012). For instance, there is a risk of leakage (i.e., intensification of activities in areas not covered by REDD+ projects) and of inadequate implementation of REDD+ activities, such as the establishment of plantations of exotic species (Visseren-Hamakers et al. 2012). Also, because there is some disconnection between the global distribution of carbon stocks and the associated biodiversity, the outcomes of REDD+ projects may be less effective for protecting biodiversity and other ecosystem services (Visseren-Hamakers et al. 2012).

Other PES mechanisms are emerging at a regional scale and are aimed at conserving forest ecosystem services. For instance, in the European Union, under the current European Agricultural Fund for Rural Development (http://ec.europa.eu/agriculture/cap-funding/budget/index_en.htm), a payment scheme has been implemented to support the development of multifunctional forests and the adoption of good management practices.

The rationale behind PES mechanisms lies in their attempt to internalize market externalities. As described above, many ecosystem services are not tradable in a market; thus, producers are unable to introduce their value into the price of the products they supply. Due to this market externality, producers will always be underrewarded for the services they provide in the absence of production incentives or subsidies. Through PES, central governments or private users and consumers make payments for the ecosystem services provided by landowners, producers, and other entities such as environmental agencies or nongovernmental conservation organizations. Examples involving the industry sector reflect how industries can often play a leading role as beneficiaries or buyers of ecosystem services, as in the case of the Nestlé Waters Programme (www.nestle-waters.com/).

Although the EU Forest Strategy is implemented at a regional scale, it nonetheless provides a good example of intergovernmental action to promote and support sustainable forest management through the coordination of forest policies by the member states and through community policies (CEC 2005). The strategy also acknowledges the multifunctional role of forests and their multiple services, and their relevance for the well-being of society. The Ministerial Conference on the Protection of Forests in Europe (http://www.foresteurope.org/) constitutes the political process at a pan-European level for the establishment of sustainable forest management. The Forest Europe strategy for 2020 developed by Ministerial Conference on the Protection of Forests in Europe has been signed by 46 countries, and lists among its main targets the valuation of multiple forest services and raising society's awareness of the importance of forests to human well-being. This strategy will foster cooperation among countries to develop and update their forest policies so as to secure and promote sustainable forest management.

Targeting different scales, the certification or information labeling of agroforestry products might also be a way to communicate environmental and other attributes that are not directly visible in the products, with the goal of promoting sustainable management of forestry resources. The rationale behind certification mechanisms is simple: if there is a market demand for differentiated agroforestry products, meaning that consumers are willing to pay for the price difference listed by producers due to their compliance with environmental standards and the consequent delivery of external benefits, then a market-based solution for sustainable rural development becomes possible. Countries like the Netherlands, Germany, and the UK are clear examples of major markets for certified forestry products, and specifically for timber. The most widely acknowledged example of certification mechanisms are the regional and national standards developed by the Forest Stewardship Council (https://ic.fsc.org/). Other less widely known initiatives include the Programme for the Endorsement of Forest Certification Schemes (http://www.pefc.org/), which is now a recognized label in Europe and in a few other countries such as Brazil and the United States. FAO (2007) estimated that around 270×10^6 ha of forests around the world, amounting to roughly 7 % of the world's forests, are certified for sustainability through an independent labeling organization. However, in less developed countries, the costs of complying with such environmental standards in addition to the costs of the certification process itself represent a great challenge for producers that want to enter a certification market.

At the local level, initiatives to manage forest ecosystem services can take many forms that depend on several factors, including land ownership, size and governance, forest location and condition, and (of course) the targeted services. Local initiatives can be independent or can derive from initiatives at larger scales, such as the global initiatives discussed above. Other examples of local actions include sustainable forest management and land-use planning, measures to improve forest resilience to disturbance (Fernandes 2013), and actions to restore degraded forests and deforested land, which can help to restore ecosystem services and biodiversity (Chazdon 2008, Rey Benayas et al. 2009).

Despite the scale of implementation, the success of these initiatives to manage forests and their multiple services will depend on the existence of governance structures and legal frameworks that safeguard access to the forest resources and respect the rights of all forest users, thereby avoiding inequity in the access to benefits (FAO 2012). It will also be very important to invest in building capacity to create conditions suitable for the implementation of sustainable forest management programs and to provide local peoples with the knowledge and tools they require to participate in the design and implementation of those programs (FAO 2012).

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Chapter 6
Carbon fluxes and storage in forests and landscapes

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Abstract We begin this chapter with a discussion of the major carbon fluxes (e.g., gross primary production, ecosystem respiration) and stocks (e.g., aboveground biomass) in forest ecosystems, as well as their relationships, and provide examples of their values from selected case studies. We pay special attention to the magnitudes of these fluxes and stocks in different forests and biomes. However, studies of carbon cycling at a landscape scale lag significantly behind those at an ecosystem level. The objective of this chapter is to provide a glimpse of current knowledge of carbon fluxes and storage in forests at both ecosystem and landscape scales. Due to the overwhelming literature on this topic, we have limited our review to lessons from selected empirical studies that demonstrate the temporal and spatial variations of the carbon cycle in a range of representative environments. We further discuss our current understanding of carbon cycles across forests and landscapes in the contexts of climate change, the impact of natural disturbances, and regulation of the carbon cycle by management actions. We present a new conceptual framework for the changes in net ecosystem production following a disturbance as a foundation

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to guide future studies. Finally, we share our vision of the direction of future carbon cycle research from both basic and applied perspectives. We support our review by citing relevant papers that provide important references for readers.

6.1 Introduction

Ecosystem play a major role in the global carbon cycle, as they store 45 % of the terrestrial carbon and account for ~50 % of soil carbon sequestration (Bonan 2008). A recent report based on long-term global inventory data indicated that the total forest carbon sink since 2000 amounts to 22 % of the global carbon sink, and that this sink is offsetting 33 % of current annual fossil fuel emissions (Pan et al. 2011). However, both carbon fluxes and storage in forests vary significantly over time (e.g., annual, decadal) and space (regional, global), and both are directly regulated by natural events (e.g., climate change, drought, wildfires, pest or disease outbreaks) and human activities (e.g., deforestation, plantation establishment, urban sprawl, management practices). For example, tropical deforestation is responsible for the release of about 1.5 Gt C per year, accounting for ~15 % of total anthropogenic carbon emissions (Peters et al. 2011). As the international community begins to address the impacts of global climate change through the development of adaptation plans (IPCC 2007), a thorough understanding of the forest carbon cycle as well as the mechanisms that regulate coupled human and natural stressors becomes increasingly important for both the scientific community and the decisionmaking community (Baccini et al. 2012, Birdsey et al. 1993, Davidson et al. 2012).

Scientific investigations of forest carbon cycling during the past three decades have been conducted using different representations of carbon storage that were based on societal needs. Prior to the 1980s, the carbon cycle was mostly investigated from the perspectives of timber yield and ecosystem production. In the 1980s, forests were hypothesized to be responsible for the missing carbon needed to close the global carbon budget, and some researchers believed that the ability of forests to sequester carbon had been significantly underestimated.

When ecosystem management emerged as the new paradigm in natural resource management in the early 1990s, researchers took advantage of the rapid advances in technology (e.g., remote sensing, eddy-covariance flux towers, stable-isotope analysis) and of new generations of ecosystem models to seek answers for questions such as the following: What determines the carbon sink strength of forest ecosystems under alternative forms of management? Can increased carbon sequestration be achieved through more intensive management? What is the relative importance of climate and disturbance in affecting the mean carbon flux and its variation? How do different fragmentation patterns affect landscape-scale carbon fluxes? Through the promotion of data sharing among research labs across the globe, the scientific community has made significant progress in understanding how forests differ in their carbon fluxes and stocks. This collective effort using open data sources has led to increasing studies of the carbon cycle at regional, continental, and global scales (e.g., John et al. 2013; Turner et al. 1995; Xiao et al. 2009, 2010, 2011; Yi et al. 2010; Zhang et al. 2012).

Recently, pressing issues arising from the high demand for renewable energy (e.g., fast-growing crops such as poplar (*Populus* spp.) and eucalyptus (*Eucalyptus* spp.) plantations to produce cellulosic ethanol) and the CO_2 emission-reduction targets adopted by many countries (e.g., IPCC 2007) triggered a new dimension in carbon cycle science (e.g., life-cycle assessment of the carbon cycle; Gelfand et al. 2011), emphasizing carbon's role in global warming (Robertson et al. 2008) and linking the carbon cycle with socioeconomic systems (e.g., carbon stocks, urbanization; Peters et al. 2011). In addition, the increasing magnitude and frequency of natural disturbances and extreme climatic events challenge our in-depth understanding of their roles in regulating carbon fluxes and stocks (e.g., Davidson et al. 2012, Gu et al. 2008). However, the core ecological research on this topic focuses on understanding the magnitude of carbon fluxes and stocks and identifying the underlying mechanisms responsible for changes in these factors in time and in space.

6.2 Carbon cycling in forests

Carbon enters a forest from the atmosphere, mostly through photosynthesis, and its storage in the forest is commonly known as "gross primary production" (GPP) or "carbon assimilation". A small amount is also input from the weathering of bedrock (M_c) and by lateral transfer by animals (A_c) and by the wind (W_c). GPP is simultaneously used to create biomass and to maintain plant metabolism through autotrophic respiration (R_A) of live tissues (e.g., leaves, stems, and roots). R_A can be broadly separated into aboveground and belowground respiration (i.e., R_{Aa} and R_{Ab}, respectively; Hanson et al. 2000). Net primary production (NPP) equals the difference between R_A and GPP, and can be divided into aboveground (ANPP) and belowground (BNPP) components. The remaining portion of GPP (i.e., NPP) can be divided into aboveground carbon allocation (AGCA) and belowground carbon allocation (BGCA), which serve as a food source for animals (A_c) and as a substrate for decomposition by decomposer organisms (D) into various trace gases (e.g., CO_2, CH_4) before returning to the atmosphere. Emissions from A_c and D are termed "heterotrophic respiration" (R_H). Forests include both live and dead organic matter (e.g., snags, dead branches, leaves), suggesting that a small amount of aboveground heterotrophic respiration (R_{Ha}) exists. This is especially true for the tropical and subtropical rainforests, where epiphytes are abundant for elevated decomposition of aboveground dead organic matter due to the high temperature (Clark et al. 2001). The sum of R_A and R_H is the total respiratory loss of a forest and is referred to as ecosystem respiration (R_e). The total amount of carbon loss from the soils—the sum of belowground autotrophic respiration (R_{Ab}) and belowground heterotrophic respiration (R_{Hb})—is termed "soil respiration" (R_s; Curtis et al. 2005, Hanson et al. 2000, Li et al. 2012). Most forests are on slopes and, therefore, the lateral fluxes of carbon

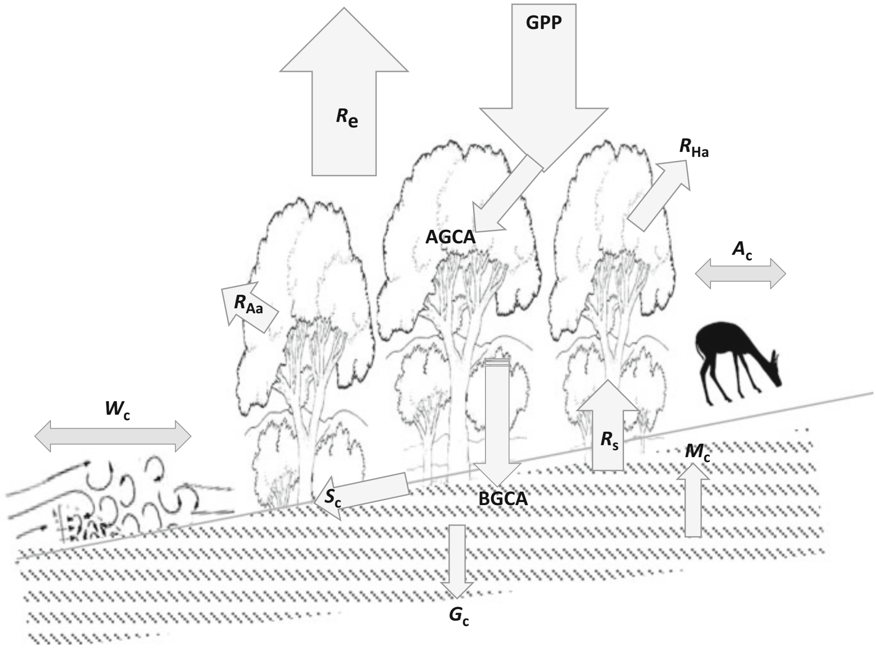

Figure 6.1 Illustration of the major carbon fluxes in a forest ecosystem, including gross primary production (GPP), ecosystem respiration (R_e), aboveground carbon allocation (AGCA), belowground carbon allocation (BGCA), soil respiration (R_s), aboveground heterotrophic respiration (R_{Ha}), aboveground autotrophic respiration (R_{Aa}), surface runoff (S_c), lateral fluxes of carbon through the wind (W_c) and animals (A_c), vertical water leaching (G_c), and upward movement through diffusion after weathering of bedrock (M_c) in the soil

through the wind (T_c, such as fine litter, leaves) and of organic materials through animals (A_c) may be significant. Finally, surface runoff (S_c) and vertical water leaching (G_c) will carry small amounts of carbon into or out of a forest (Fig. 6.1). These carbon fluxes and their relationships can be summarized as follows:

$$\text{GPP} = [\text{NEP} + R_e]$$
$$\text{NPP} = [\text{GPP} - R_A]$$
$$\text{NPP} = [\text{ANPP} + \text{BNPP}]$$
$$\text{ANPP} = \text{Vegetation Growth} - \text{Litterfall}$$
$$\text{BNPP} = \text{Root Growth} - \text{Root Mortality}$$
$$R_e = [R_A + R_H] - (M_c)$$
$$R_A = R_{Aa} + R_{Ab}$$
$$R_H = [R_{Ha} + R_{Hb}] - (M_c)$$
$$\text{NEP} = [\text{AGCA} + \text{BGCA}] + (S_c + T_c + G_c + A_c - M_c)$$
$$R_s = [R_{Ab} + R_{Hb}] - (M_c)$$

where NEP represents net ecosystem production, the flux terms inside the square brackets account for large proportions of the total, and those inside the round brackets are minor or difficult to quantify.

The magnitudes of these flux terms vary significantly among ecosystems and over time. Among them, GPP and R_e are the two largest fluxes, and the difference between them determines the carbon sequestration strength of an ecosystem (Chen et al. 2004, Schwalm et al. 2010). For example, Yuan et al. (2009) found that GPP explained a significant proportion of the spatial variation of NEP across evergreen needleleaf forests (also see Luyssaert et al. 2007). Conversely, R_e determines the magnitude of NEP for a range of deciduous broadleaf forests (Yuan et al. 2009). The global average GPP of forests is approximately 880 g C m^{-2} yr^{-1}, but varies from less than 500 g C m^{-2} yr^{-1} to nearly 3000 g C m^{-2} yr^{-1}, with the highest values in the humid tropics (e.g., Amazonia, central Africa, southeast Asia), where both temperature and moisture requirements are satisfied for photosynthesis (Sun et al. 2011, Yuan et al. 2010). Extremely high GPP has also been reported in plantations of loblolly pine (*Pinus taeda*; >2300 g C m^{-2} yr^{-1}; Gough et al. 2002, Noormets et al. 2012) and eucalyptus in Brazil (*Eucalyptus* spp.; 6640 g C m^{-2} yr^{-1}; Stape et al. 2008). The deciduous forests at high latitudes (e.g., the boreal region) have lower GPP levels, at 460 g C m^{-2} yr^{-1} or lower (Li et al. 2007a). The growing season length, annual precipitation, and temperature are the three most critical variables that determine GPP and its changes over time. Recent studies have shown that extended droughts (Xiao et al. 2009) and disturbances (Amiro et al. 2010) can substantially reduce NEP, primarily by reducing GPP while simultaneously altering R_e.

For forests that are carbon sinks, R_e is slightly smaller than GPP but of similar magnitude and varies from 300 to 600 g C m^{-2} yr^{-1} in boreal forests, from 600 to 900 g C m^{-2} yr^{-1} in temperate forests, and from 1000 to 2500 g C m^{-2} yr^{-1} in tropical forests (Yuan et al. 2010). The global average R_e is approximately 790 g C m^{-2} yr^{-1}, with the highest values occurring in the tropical moist forests and lowest values in the cold tundra and dry desert regions. Luo and Zhou (2006) also reported that the tropical moist forests have significantly higher R_e than other ecosystems, which results in mean NEP values of 400, 275, and 120 g C m^{-2} yr^{-1} for the tropical, temperate, and boreal forest biomes, respectively (Bonan 2008). In forest plantations, NEP can exceed 1000 g C m^{-2} yr^{-1}, making them good candidates for bioenergy systems for ethanol production (e.g., from eucalyptus or poplar). Consequently, alternative management practices are often sought to increase GPP or decrease R_e because forest NEP is determined by their balance. For recently disturbed or old-growth forests that release carbon into the atmosphere, R_e is typically larger than GPP.

For many forests, the amount of carbon emitted by forest soils as R_{Ab} and R_{Hb} (i.e., as R_s) accounts for the majority of R_e (60 to 80 %). R_s depends strongly on soil temperature, soil moisture, and total soil organic matter, which are important regulators of the metabolic processes involved in belowground R_{Ab} and R_{Hb} (Edwards and Sollins 1973, Martin et al. 2009). Consequently, soil temperature and moisture are often used to calculate R_s using simple temperature-based exponential models or other model forms such as the Lloyd and Taylor or Boltzmann–Arrhenius models (Davidson et al. 2005; Li et al. 2012; Noormets et al. 2008; Perkins et al. 2011; Reichstein et al. 2005; Richardson et al. 2006, 2007). Interestingly, the regulation of R_s by thermal and moisture conditions is not linear; instead, optimal and threshold values exist (Niu et al. 2012, Xu et al. 2011). In recent years, the scientific community has recognized that both phenology and GPP can directly affect R_{Ab} (DeForest et al. 2006, Högberg et al. 2001). Currently, we lack reliable methods to partition R_{Ab} and R_{Hb}, preventing us from estimating the magnitudes and dynamics of these two terms. For managers who are interested in increasing carbon sequestration (i.e., increasing the sink strength), soil seems to be the only place to store carbon in the long term because trees and understory vegetation will ultimately die and then decompose, releasing CO_2 back into the atmosphere (Noormets et al. 2012). Consequently, researchers who study the carbon cycle have focused on R_s (Euskirchen et al. 2003, Noormets et al. 2008, Xu et al. 2011).

Other carbon flux terms are typically small and have received significantly less attention despite their importance in some forests. For example, few studies have examined the amount of carbon lost through runoff and groundwater that will eventually leave the forests through streams and rivers (Bolin et al. 1979; Cardille et al. 2007; Hope et al. 1993, 1997; Roulet and Moore 2006). Richey et al. (2002) found that outgassing ("evasion") of CO_2 from the rivers and wetlands of the central Amazon basin constitutes an important carbon loss process, equal to 1.2 Mg C ha^{-1} yr^{-1}, which is equivalent to more than 30 % of forest NEP in the region. Two major studies on the efflux of CO_2 released from inland rivers and streams in the United States found that they were supersaturated with carbon and emitting 97 ± 32 Tg C yr^{-1} (Butman and Raymond 2011, Melack 2011). Nevertheless, the loss of carbon in most of the world's watersheds remains unknown. In addition, carbon fluxes associated with horizontal movements by wind and wildlife that directly carry carbon into or out of a forest have not been studied in the context of the complete carbon cycle.

The magnitudes of all of the components of the carbon cycle are not static, but vary greatly over time. Although pronounced seasonal changes are coupled well with interannual climatic variations, mounting evidence suggests that the variations over periods of two or more years (i.e., an interannual scale) or even at decadal scales are significant (Gough et al. 2008b, Richardson et al. 2007). For example, at the Oak Openings forest in northwestern Ohio, we found higher-than-average NEP, with values that varied from 1.9 to 4.1 Mg C ha^{-1} yr^{-1}, likely due to a combination of climatic variation, drought, and disturbances such as fires (Noormets et al. 2008). In a maple (*Acer* spp.) forest in Japan, Saigusa et al. (2005) estimated the annual NEP to be 237 ± 92 g C m^{-2} year^{-1} (mean ± SD) from 1994 to 2002, but NEP varied from 59 to 346 g C m^{-2} yr^{-1} between years (i.e., an interannual variability of up to 287 g C m^{-2}). In the Pacific Northwest of North America, Krishnan et al. (2009) found that a 57-year-old Douglas-fir (*Pseudotsuga menziesii*) stand was a moderate carbon sink, with annual NEP ranging from 267 to 410 g C m^{-2} yr^{-1} during a 9-year period. This variation was much higher than that in an old-growth forest in southern Washington State, which was generally a weak carbon sink and could occasionally become a carbon source (Chen et al. 2004).

The cumulative NEP is the amount of carbon stored in a forest without physical removal of carbon from the ecosystem by disturbances such as timber harvesting, commercial thinning, or wildfire (i.e., carbon storage = $\sum$[NEP – removals]).

Forests store a large amount of carbon, with 471 Pg C (55 % of total forest carbon) in tropical forests, 272 Pg C in boreal forests, and 119 Pg C in temperate forests (Pan et al. 2011). This totals an estimated 862 Pg C, with 44 % in the soils, 42 % in live biomass, and 8 % in deadwood. However, these proportions vary greatly among ecosystem types, climates, disturbance histories, land-use histories, management types, and soils (McKinley et al. 2011). Globally, tropical forests stored 56 and 32 % of carbon in their biomass and soil, respectively, whereas boreal forests store 20 and 60 % of the carbon in the biomass and soil, respectively (Pan et al. 2011). In the United States, McKinley et al. (2011) reported that the forests contained ~41 000 Tg C and that this storage increased at a rate of 192 Tg C yr^{-1}.

The major carbon pools in forests include living overstory and understory vegetation, dead biomass (e.g., coarse woody debris, snags, litterfall, dead roots), and soils. The amount of carbon stored in animals is small in most ecosystems and has rarely been studied or considered in the context of a forest's carbon budget. However, this distribution varies greatly among forests and regions. A few selected sites from the literature have total carbon storage (excluding animal biomass) ranging from less than 100 Mg C ha^{-1} to as high as 700 Mg C ha^{-1}, but most values are between 200 and 450 Mg C ha^{-1} (Table 6.1). On average, mineral soils contain the largest carbon pools in the national and north-central regions of the United States, where they account for approximately 42 and 52 % of total forest carbon, respectively (Turner et al. 1995). In contrast, live trees represented the largest carbon pool in the Missouri Ozarks and the Pacific Northwest, respectively, accounting for about 55 and 71 % of total forest carbon (Li et al. 2007b). The carbon pools of a mixed oak (*Quercus* spp.) forest in the southeastern Missouri Ozarks contain 182 Mg C ha^{-1} (Li et al. 2007a), with 80.2 Mg C ha^{-1} in living trees, 22.9 Mg C ha^{-1} in dead biomass, 20.0 Mg C ha^{-1} in roots, and 53.7 Mg C ha^{-1} in the soil (i.e., total soil carbon except roots). The mean live tree carbon pool at the site was ~17 and 21 % higher than the national average and the average for the north-central United States, respectively (Turner et al. 1995), but it was 16 % lower than the average for the Pacific Northwest (Smithwick et al. 2002). The mean soil carbon was about 16 % higher than that in the Pacific Northwest (Smithwick et al. 2002), but was 12 and 22 % lower than averages for the nation and for the north-central United States, respectively (Turner et al. 1995). On average, these results suggest that temperate forests store approximately 50 % of their carbon as aboveground biomass (AGB) and 50 % as belowground biomass (BGB). However, this estimate is imprecise because carbon pool estimates are influenced differently by site-specific disturbance regimes and because the definitions of some major carbon pools (especially for dead organic matter) vary significantly among studies (Bradford et al. 2008, Grier and Logan 1977, Matthews 1997, Schlesinger 1997).

The carbon storage in global forests varies greatly in both its magnitude and its within-system distribution (Table 6.1). Overall, tropical forests have high AGB but not necessarily high BGB (e.g., 305 Mg ha^{-1} AGB but negligible BGB for the Tapajos National Forest in the east-central Amazon; Saner et al. 2012). Keith et al. (2009) claimed that *Eucalyptus regnans* forests in Victoria, Australia, have the highest biomass in the world. In contrast, the BOREAL study found that up to 88 % of

Table 6.1. Carbon storage as aboveground biomass (AGB), belowground biomass (BGB), and coarse woody debris (CWD), and the total of these three components, in selected representative forests from the three dominant forest biomes.

			Carbon storage (Mg C ha^{-1})				
Biome	Region	Dominant species	AGB	BGB	CWD	Total	Source
Tropical	Tapajos National Forest	*Sclerobium chrysophyllum*	305.00	NA	NA	339.2	Nepstad et al. (2002)
	Sabah, Borneo	*Shorea* spp.	128.00	NA	70.60	210.75	Saner et al. (2012)
Temperate	WRCCRF, WA, USA	*Pseudotsuga menziesii*	313.23	174.22	NA	487.45	Harmon et al. (2004)
	MOFEP, MO, USA	*Quercus* spp.	80.20	73.70	22.90	182.7	Li et al. (2007b)
	Walker Branch, TN, USA	*Quercus* spp. *Acer* spp.	97.30	91.90	NA	189.20	Curtis et al. (2002)
	MMSF, IN, USA	*Acer saccharum* *Quercus* spp.	101.90	124.30	NA	226.20	Curtis et al. (2002)
	Harvard Forest, MA, USA	*Quercus* spp.	105.00	111.60	NA	216.60	Curtis et al. (2002)
	UMBS, MI, USA	*Populus* spp.	62.60	NA	NA	78.60	Curtis et al. (2002)
	Willow Creek, WI, USA	*Populus* spp. and *Acer* spp.	78.60	222.70	NA	301.03	Curtis et al. (2002)
	Victoria, Australia	*Eucalyptus regnans*	1819.0	1025.0	NA	2844.0	Keith et al. (2009)
	Chiloé Island, Chile	*Nothofagus nitida*	290.50	NA	158.00	448.50	Carmona et al. (2002)
Boreal	Saskatchewan, Canada	*Populus* spp.	93.34	35.99	291.10	158.44	Gower et al. (1997)
		Picea mariana	49.24	390.36	61.60	445.76	Gower et al. (1997)
		Pinus banksiana	34.55	14.20	202.30	68.98	Gower et al. (1997)
	Manitoba, Canada	*Populus* spp.	56.95	97.170	222.70	176.39	Gower et al. (1997)
		Picea mariana	57.21	418.36	38.10	479.38	Gower et al. (1997)
		Pinus banksiana	28.99	25.78	136.00	68.37	Gower et al. (1997)

the boreal forest ecosystem carbon was stored in the soil (Gower et al. 1997). This difference was more evident in the black spruce (*Picea mariana*) stands in Saskatchewan and Manitoba, Canada, and less evident in the aspen (*Populus* spp.) or jack pine (*Pinus banksiana*) stands within the same region (Table 6.1). Aboveground carbon pools at five AmeriFlux sites in the forests of the eastern United States (Curtis et al. 2002) differed significantly from those at more productive southern sites and from those in less productive northern hardwood sites in Michigan and Wisconsin (Table 6.1.). However, the Willow Creek Site in Wisconsin, which was dominated by aspen and northern hardwoods, had more soil carbon than other sites in the region (Curtis et al. 2002). In the southern hemisphere, old-growth Chilean forests were found to have greater biomass of coarse woody debris than most temperate forests other than those in the Pacific Northwest of North America (Schlegel and Donoso 2008).

6.3 Carbon dynamics in forested landscapes

Changes in carbon fluxes and storage across forested landscapes (i.e., across multiple ecosystems arranged in a cohesive mosaic) have been difficult to understand and measure due to the complex interactions between landscape structure and ecosystem processes and changes in these interactions over time. The two critical issues that must be accounted for in any landscape-scale research are heterogeneity and scaling. Although both topics have received extensive attention during the past 20 years, much less effort has been spent on their relationship to carbon cycles, due mostly to the high costs of such studies and a lack of effective methods. At the ecosystem level, several mature methods (e.g., the eddy-covariance technique, biometric sampling, chamber-based flux measurement, ecosystem modeling) can provide us with reliable estimates of both fluxes and storage (Chen et al. 2004). However, scaling-up of ecosystem-level carbon fluxes and storage to a landscape level is not always accurate because of the presence of many smaller elements (e.g., corridors) and of interactions among patches (Desai et al. 2008).

Intensive measurements of carbon fluxes and storage for the dominant landscape elements have attempted to support scaling-up of the estimates to the landscape level (Chen et al. 2004; Jenkins et al. 2001, 2003; Pan et al. 2009; Smithwick et al. 2009; Turner et al. 2011; Turner et al. 2004). For example, Euskirchen et al. (2003) measured the R_s, microclimate, and litter depth of six dominant patch types in a managed forest landscape in northern Wisconsin in 1999 and 2000. They found not only a significant difference among the patches but also a 37 % higher R_s in 1999 than in 2000, suggesting that the changes in any flux term over time must also be accounted for in any effort to understand the landscape-scale carbon cycle. A similar bottom-up approach for scaling up NEP was attempted by installing permanent and mobile eddy-covariance towers (Ryu et al. 2008) in an effort to include heterogeneous patch types and their associated characteristics in landscape-scale estimates. This effort was assisted by a cross-lab collaboration that combined spatiotemporal data from eddy-covariance towers (Desai et al. 2008, Noormets et al. 2008), R_s measurements (Martin et al. 2009), and models (Ryu et al. 2008, Zhang et al. 2012). However, the resulting carbon flux estimates remain problematic because no consideration was given to the influence of patch interactions or the contributions from minor elements of the landscapes (e.g., roads, small lakes). The results of these studies will nonetheless support scaling-up if they can be coupled with the spatially continuous characteristics of the landscape structure (Zheng et al. 2004) and will support the validation of modeled landscape-scale carbon fluxes and storage (Xiao et al. 2009).

Few studies have attempted landscape-level investigations of the carbon cycle. Several studies have been conducted in the Brazilian tropical forest region under the Large-Scale Biosphere-Atmosphere Experiment (http://lba.cptec.inpe.br/lba/site/). The researchers found that Amazonia constitutes a large global carbon store. Forest conversion in Amazonia is turning these forests into a net source of atmospheric carbon (Davidson et al. 2012, Tian et al. 1998). Recent measurements indicate that undisturbed Amazonian forest systems may be a net carbon sink, although the importance of carbon sequestration in regrowing forests on abandoned land is unclear (also see Pan et al. 2011). Dantas de Paula et al. (2011) found that carbon stocks varied greatly among landscape patches and that forest interiors retained nearly three times the carbon (202.8 ± 23.7 Mg C ha^{-1}) of forest edges due to edge effects. They found that 92 % of the forest stored only half of its potential carbon due to fragmentation and the resulting edge effects, including wind damage and exposure to drought. These findings contradict those of a study in the Delaware River landscape, where fragmented landscapes had higher NPP (Jenkins et al. 2001). In Northern Wisconsin, a 395-foot-tall tower was used to directly measure the net exchanges of carbon, water, and energy in a landscape dominated by northern hardwoods (Bakwin et al. 1998, Chen et al. 2008). The NEP and R_e reported from this tower represent the cumulative values for an eddy-covariance tower with a fetch length greater than 10 km in which different ages and types of patches coexist. To scale up the results to a regional level, both aircraft-based flux measurements (Stephens et al. 2007) and intensive field campaigns were conducted to quantify the C fluxes and storage, including the Midwest Intensive Field Campaigns conducted by the North American Carbon Program (http://www.nacarbon.org/nacp/).

Coupling remote sensing with ecosystem modeling and ground measurements of carbon fluxes and storage can also provide good estimates of carbon fluxes (e.g., Sun et al. 2011; Xiao et al. 2010, 2011) and pools (e.g., Blackard et al. 2008) at landscape, regional, and global scales because the emphasis is on the overall region, and several reliable satellites can cover the globe with a coarse resolution (e.g., MODIS). At the landscape scale (i.e., tens of kilometers solution), no satellite data can quantify the parameters (e.g., leaf area, microclimate) required to model carbon fluxes or storage with sufficient spatial or temporal resolution. Landsat imagery has the necessary spatial resolution (30 m), but has insufficient temporal resolution (due to the 16-day repeat cycle of the satellites and data gaps that result from cloud contamination) and measures only a limited number of spectral bands, thereby preventing accurate estimation of carbon gains and losses. A few promising, high-resolution remote-sensing technologies are being tested in carbon cycle research, such as LIDAR (Chopping et al. 2012, Parker et al. 2004) and AVIRIS (Roberts et al. 2004), although application of the latter technology outside of the western countries remains difficult. Predictions of belowground carbon storage and carbon fluxes based on remote sensing are not feasible. Consequently, our current knowledge of landscape-scale carbon fluxes and storage is based on the predictions of ecosystem models (e.g., belowground carbon; Gower et al. 1997) or on spatial interpolations between point estimates (e.g., Euskirchen et al. 2002; Pan et al. 2009; Turner et al. 2004, 2009).

A small handful of studies were conducted to link landscape structure with key carbon fluxes or storage pools (Jenkins et al. 2001, Noormets et al. 2007, Turner et al. 2004, Zheng et al. 2004). Based at the Chequamegon National Forest in Wisconsin, Zheng et al. (2004) produced a high-resolution map of stand age calculated from field measurements of tree diameter. Various vegetation indices were derived from Landsat 7 ETM+ imagery through multiple-regression analyses to produce an initial AGB map. This study is among the few in which AGB was estimated over a long study period (here, 30 years) based on near-infrared reflectance and the normalized-difference vegetation index. However, carbon fluxes and storage from other ecosystem components (e.g., the soil) may not be determined using this approach.

Scaling-up from trees and stands to landscapes (i.e., a bottom-up approach) appears to be more plausible than satellite-based approaches because many smaller structural elements cannot be quantified even from Landsat images, such as smaller woodlands, areas of edge influence (AEI, i.e., areas along the edges of fragmented stands where edge effects are significant), riparian zones, and narrow corridors. These structural features may be the dominant features of a landscape (e.g., dotted woodlands in the Midwest region of the United States) or may play significant roles in estimating landscape-scale carbon fluxes and storage. For example, integrating the terrestrial and aquatic components of regional carbon budgets in managed landscapes has been among the research foci (cf. Buffam et al. 2011). Giese et al. (2003) investigated the carbon pools of a managed riparian forest in the coastal plains of South Carolina and found a high potential for carbon storage, especially as BGB. A recent study by Rheinhardt et al. (2012) found that the carbon stored in riparian zones in the headwater reaches of a watershed in an agriculture-dominated landscape amounted to only about 40 % of the potential capacity.

As another example, forests influenced by clearcut edges were found to be responsible for a 36 % reduction of biomass in a Brazilian tropical forest (Laurance et al. 1998). Zheng et al. (2005) used the changes in land cover type and composition from 1972 to 2001 and an R_s model to assess the contribution of AEI to carbon emission in the Chequamegon National Forest in Wisconsin. They found that changes in land cover increased landscape R_s by approximately 7 % during the 30-year period. This is likely to be significant because of the large portion of AEI in the landscape. However, these pioneering studies are far from providing a comprehensive understanding of all major carbon fluxes and storage. After 14 years of investigating the Chequamegon National Forest landscape (Chen et al. 2006), we are still incapable of predicting the carbon fluxes and storage in AEI, roadside areas, riparian forests, and lakeshore forests. Li et al. (2007b) found that the total AEI amounted to approximately 48, 74, 86, and 92 % of the landscape with the depth of edge influence (i.e., the distance inside a forest stand to which the edge effect is significant) set at 30, 60, 90, and 120 m, respectively. AEI and roads accounted for 48 and 8 %, respectively, of the landscape in this study area, and their proportions had increased from 1972 through 2000 (Bresee et al. 2004). Across the United States, the total amount of AEI accounts for 42.8 % of our national forests (Riitters et al. 2002), but its contribution to the landscape carbon cycle remains unknown (Harper et al. 2005).

There are also many ignored landscapes for which our knowledge of carbon fluxes and storage is limited. This list includes urban areas, despite the important effects of intensive management, direct interactions between human populations and their environment, and the high potential of these areas to sequester carbon. This gap in our knowledge is particularly important because urban areas are growing at a faster rate than any other land-use type (Lal and Augustin 2012). Peters et al. (2011) argued that urban areas contributed 71 % of global energy-related CO_2 emissions in 2006. The United Nations reported that the global urbanization rate (i.e., the proportion of the population living in cities) was 49.6 % in 2007 and is expected to reach 70 % by 2050 (http://esa.un.org/unpd/wup/index.htm). Almost all of this increase will come from urbanization of developing countries, providing both a challenge and an opportunity to manage carbon emissions. Davies et al. (2011) examined the quantities and spatial patterns of AGB in Leicester, UK, after surveying vegetation across the entire urban area and reported storage of 3.16 kg C m^{-2}, with 97.3 % of this pool being associated with trees rather than with herbaceous and other woody vegetation. McKinley et al. (2011) stated that the carbon density of urban landscapes in the United States was similar to that of tropical forests. In summary, it is clear that the structure of and changes in land mosaics are important components of landscape-level carbon fluxes and storage (Noormets et al. 2007, Turner et al. 2009). Yet despite this importance, there remain many knowledge gaps for predicting the carbon cycle at this scale.

6.4 The roles of climate and disturbance

Forests and landscapes are not static; rather, they are constantly changing, resulting in large temporal changes in carbon fluxes and storage. Three driving forces for these changes often act together (Caspersen et al. 2000, Pan et al. 2009, Smithwick et al. 2009): changes in the environment (e.g., climate, soil, atmospheric chemistry) of the ecosystem or landscape, natural disturbances, and management practices.

6.4.1 Climate change and the carbon cycle

Global climate change now appears to be inevitable and will have profound impacts on natural ecosystems at all spatial scales. The feedbacks between forests and climate are complex, but a unique characteristic among the multiple feedbacks results from the longevity of trees and forests. Trees, in general, seem to be more tolerant of change than shrubs and herbaceous species (i.e., they exhibit relatively slow responses), but fast responses of carbon fluxes and storage to climate change have been widely reported because climatic factors directly regulate all flux terms for a forest ecosystem (Chen et al. 2002). The “fertilization” of trees by increasing atmospheric CO_2 will mostly likely enhance GPP (Pan et al. 2009), but elevated temperatures caused by increasing atmospheric concentrations of CO_2 and other greenhouse gases (CH_4 and N_2O) will also promote respiratory losses (R_e), resulting in an uncertain change in NEP (Bonan 2008).

Large-scale experiments to simulate the effects of climate change (CO_2, O_3, temperature, precipitation) have been initiated in several forests, including the cool-temperate Harvard Forest (Melillo et al. 2011), a poplar plantation in northern Wisconsin (Karnosky et al. 2003), and a loblolly pine plantation in the Duke Forest (Ellsworth et al. 2012; Oren et al. 2001), but the results from these experiments pointed to different trends for the different flux terms, with great uncertainties. One primary reason for the uncertainty is that no experiment has considered more than three factors related to the future climate due to the complexity and high costs of such modeling. Consequently, these predictions will need to be based on validated models. Interestingly, climatic extremes are predicted to be one of the major consequences of climate change, yet little is known about the effects of climate extremes on ecosystem processes (Ciais et al. 2005, Xiao et al. 2009), especially if multiple extreme events occur simultaneously (e.g., a heat wave plus drought). Although much experimental work has been conducted on the effects of chronic warming on ecosystems, most of these experiments were (understandably) conducted with short vegetation such as grasses and shrubs (e.g., Hovenden et al. 2008, Shaw et al. 2002). Few past studies have examined the effects of acute heat stress (short-term, high-temperature events) on naturally occurring vegetation (Melillo et al. 2011). Recent reviews have highlighted the significant negative impacts of heat stress on trees and forests (Allen et al. 2010, Rennenberg et al. 2006). In addition, researchers have not examined how landscape heterogeneity will respond to the changing climate, adding one more challenge for predicting changes in carbon fluxes and storage.

The responses of carbon fluxes and storage in forest ecosystems and forested landscapes to climate change are difficult to predict because the underlying mechanisms are much more complex than previously thought. Several particularly vexing challenges associated with climate change raise the following questions:

1. How the impact of climate change will extend beyond the effects of chronic warming and CO_2 fertilization to include interactions among multiple factors

(e.g., O_3, N deposition) and extreme physical and biological events (e.g., drought, asymmetric warming; Gutschick and Bassirirad 2010, IPCC 2007)?

2. How significant variation in both the driving forces and the ecosystem responses across temporal and spatial scales will affect forest processes (Jung et al. 2010, Martinez-Meier et al. 2008, Xiao et al. 2010)?
3. How our knowledge of the regulatory mechanisms for different fluxes that arise from feedbacks among the driving processes must be improved to allow these mechanisms to be incorporated in ecosystem models?

Ecosystem models have become increasingly important tools to answer these questions. Hundreds of ecosystem models have been developed during the past four decades and all have included a range of components in the carbon fluxes and storage pools. However, comparisons among the models and validation against field measurements of carbon fluxes and storage indicate that none of the models can be reliably applied to all ecosystem types or at all scales (Schaefer et al. 2012). Landscapes are composed of multiple ecosystem types; thus the modeling community faces the challenge of developing a new generation of models that accounts for this diversity. Another frontier in addressing landscape-scale responses to the changing climate will be to develop location-specific predictions of the future climate so that ecosystem models can be properly parameterized (e.g., regional downscaling modeling; Spak et al. 2007). This is because the spatial resolutions of the current global circulation models are too coarse (>100 km) and therefore cannot capture the effects of heterogeneous landscape elements, which frequently act at resolutions as low as 10 m. One well-known exercise is the Wisconsin Initiative on Climate Change Impacts (http://www.wicci.wisc.edu/), in which high-resolution regional predictions are being made to assess the impacts of climate change on Wisconsin's ecosystems. The program combines cutting-edge climate modeling capabilities with field expertise to assess the impacts on forest production, biodiversity, and the development of practical decision-support information at fine scales.

6.4.2 Disturbance and the carbon cycle

The responses of the carbon cycle of forested landscapes to natural disturbances have received much attention (Amiro et al. 2010, Balshi et al. 2009, Goetz et al. 2012, Kurz et al. 2008, Turner 2010). This is because natural disturbance often changes the landscape structure immediately, resulting in rapid changes in the magnitudes and directions of carbon fluxes and storage. Wildfires, outbreaks of insects and diseases, and windstorms are among the major natural disturbances in the northern hemisphere that have profound effects on forest carbon cycling (Amiro et al. 2010). Worldwide, fire is a key influence on global vegetation patterns, and especially on the distribution of forests; in the absence of fire, forest cover would about double, from 27 % of the vegetated land surface to 56 % (Bond et al. 2005). Thus, fire also has a profound influence on carbon storage.

Wildfires have been the most important disturbances in many regions. They not only directly produce carbon loss during the burn but also produce significantly different environments that, in turn, change the magnitudes and directions of subsequent carbon fluxes. Gower et al. (1997) used an ecosystem model to simulate the carbon balance of the Canadian boreal forest since the 1930s and found that the effects of CO_2, temperature, and precipitation varied interannually but generally balanced out over long time periods and large areas. Forest fires during this period had the greatest direct impact on carbon emissions from the system. Balshi et al. (2009) estimated that decadal-scale CO_2 emission caused by fires in the boreal region of North America will increase to 2.5 to 4.4 times the present level by the end of this century. Vasileva et al. (2011) found that wildfires in central Siberia are among the major factors driving the short-term (synoptic) variability of near-surface CO_2 during the warm season. At the stand level, Concilio et al. (2006) found that R_s not only varied in response to fire intensity but that its spatial and temporal variations were also greatly dependent on the patch patterns of the understory vegetation. One of the best examples of alteration of the carbon cycle at the landscape level is from Yellowstone National Park, where large wildfires in 1988 burned 47 % of the lodgepole pine (*Pinus contorta*) forests, a major forest type in the park that is prone to fires; it covers a total area of 525 000 ha. These fires caused a loss of 13.6 Mg C ha^{-1} (Kashian et al. 2006, Turner et al. 2004). However, postfire carbon accumulation can be rapid relative to historical fire intervals. In the park, about 80 % of the prefire carbon is typically recovered within 50 years and 90 % is recovered within 100 years, although ecosystem carbon is sensitive to variations in stand structure (e.g., basal area) and stand age (Kashian et al. 2013). Forests in the park would store substantially less carbon, however, if fire intervals decreased substantially as the climate warms (Westerling et al. 2011).

Deforestation caused by timber harvests, fuel-reduction treatments, and other types of land management are major anthropogenic disturbance agents that shape carbon cycles in the world's forested landscapes. Compared to natural disturbances, the influences from human activities on carbon cycling are direct, dramatic, extensive, and sometimes long lasting. For example, rainforest fragments in central Amazonia have been found to experience a marked loss of AGB caused by sharply increased rates of tree mortality and damage near the margins of the residual patches (Laurance et al. 1998). In the eastern United States, the current high carbon storage and NEP in forests are the consequences of forest regrowth after large-scale clearing of these forests between 1860 and the 1960s (Pan et al. 2009). However, management protocols during the late twentieth century were designed to maximize timber production, control erosion, prevent wildfires, and conserve species diversity. With increased awareness of other ecosystem services, such as carbon sequestration, our current challenge is to revisit the conventional management protocols at both stand and landscape levels to sustainably achieve multiple objectives.

Our knowledge of the carbon cycles in forested landscapes is not solely about the magnitudes of carbon fluxes and storage but also about how they change over time. Obviously, both human and natural disturbances must be included in the conceptual framework. These changes were first discussed in the pioneering research of

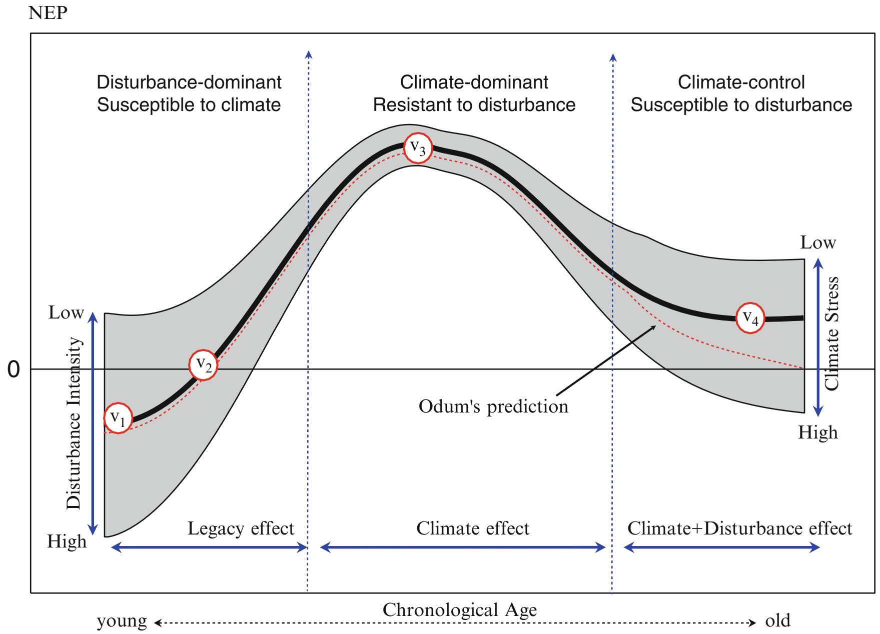

Figure 6.2 A hypothetical framework for predicting the changes in net ecosystem productivity (NEP) caused by climate variation superimposed on the effects of disturbances as a function of the time following a disturbance and four different stages (V_1 to V_4)

Odum (1969), but research has expanded greatly during the past two decades (Amiro et al. 2010, Chen et al. 2004, Euskirchen et al. 2006, Gough et al. 2008a, Harmon et al. 1990, Kashian et al. 2006, Pregitzer and Euskirchen 2004, Turner et al. 2011). Here, we offer a brief hypothetical discussion of NEP given that much of the current attention is on the strength of forest sequestration of carbon (i.e., on the magnitude of NEP).

Although the general predictions of Odum's (1969) succession theory explain ontogenetic changes, they do not address the variability among stands. Direct measurements of NEP have shown that considerable variability exists between stands of similar ages and developmental stages. A disturbance event is thought to move a stand forward or backward within the successional time series. The implicit assumption is that the sequence of conditions that constitute the successional series is constant and invariant. Here, we propose an alternative view: a three-stage conceptual framework based on the changes in NEP after a disturbance (Fig. 6.2).

During Stage 1 (V_1, Fig. 6.2), the nature and severity of a preceding disturbance are likely to be the major determinants of the ecosystem carbon balance. The increase in respiration caused by an increase in dead organic matter, changes in soil compaction and aeration, and changes in the ecosystem energy balance relative to the decrease in assimilation caused by a reduction in the effective leaf area and an altered radiation balance that affects the ratio of evaporation to transpiration may vary greatly depending on the disturbance type, disturbance intensity, and prior site conditions. Consequently, the range of variation of NEP is high during this stage (see Amiro et al. 2010, Chen et al. 2004, Euskirchen et al. 2006, Gough et al. 2008a). As legacy effects weaken during subsequent stand development and as respiration becomes dependent on new carbon inputs, the stand enters Stage 2 (V_2, Fig. 6.2), in which the magnitude of NEP depends most strongly on ecosystem composition and structure and NEP is increasingly sensitive to variations in climate. During late-successional stages (V_3 and V_4, Fig. 6.2), as the trees reach and pass their age of maximum growth rate, the site's nutrient and water availability are likely to render the forest increasingly susceptible to climate anomalies. Recently, scientists concluded that old-growth forests absorb substantial amounts of CO_2 from the atmosphere (Carey et al. 2001, Luyssaert et al. 2008)—a finding that contradicts Odum's theory and that has been touted as the basis for a global forest carbon management policy based on the preservation of these communities. However, with increasing mortality of overmature trees, the utilization of the dead organic matter in respiration will respond more strongly than assimilation to climate fluctuations, contributing to greater interannual variability of NEP (Chen et al. 2002, Gough et al. 2008a). Clearly, late-successional ecosystems have higher interannual variability in NEP that depends strongly on variations in the relationship between climate and disturbance.

Our hypothetical framework can be summarized as follows: variation in ecosystem NEP during the early development stages is primarily determined by the nature and severity of the preceding disturbance event (i.e., a legacy effect), the effects of climatic variability on NEP are most significant during the late-successional stages, and stands in intermediate developmental stages are most resilient against these influences and their NEP is determined most tightly by intrinsic vegetation properties and edaphic constraints.

The carbon cycle has long been a core component in many large-scale manipulative experiments that evaluated alternative management options. For example, the carbon sequestration capacity of a forest is broadly determined by the balance between its photosynthetic gains and its respiratory losses. To maintain optimal short- and long-term sequestration rates, the forest can be managed by retaining sufficient trees (i.e., leaves) to maintain a high rate of photosynthesis and provide a good buffer for the understory and soil microclimate (e.g., decreased respiration through lowered temperature). The foundation for this framework is that forests can be managed best by maintaining high photosynthetic rates (i.e., carbon gain) by retaining a sufficient number of green trees (i.e., leaves) and by reducing ecosystem respiration (i.e., losses) by moderating the forest and soil microclimate and structure. In the Missouri Ozark Forest Ecosystem Project, we first examined the changes in carbon storage under different management regimes and found that single-tree uneven-aged management and clearcut even-aged management of stands reduced total carbon storage from 182 Mg C ha^{-1} to 170 and 130 Mg C ha^{-1}, respectively. Although these changes are expected due to the removal of timber from the sites, the harvests reduced carbon pools in live tree biomass by 31 % under uneven-aged management and by 93 % under even-aged management, and increased coarse woody debris carbon pools by 50 % under uneven-aged management and by 176 % under even-aged management compared with the levels in the absence of harvesting (Li et al. 2007b). In a parallel study, Concilio et al. (2005) found that selective thinning in an experimental forest in the Sierra Nevada Mountains produced a similar effect on both mixed coniferous and hardwood forests by elevating soil respiration, moisture content, and temperature and, consequently, thinning increased R_s by 14 %. Xu et al. (2011) found that the summer mean R_s and soil moisture tended to be higher in wet years (2004, 2006, and 2008) and lower in dry years (2005 and 2007) under even-aged and uneven-aged management than in unharvested stands in the Missouri Ozark Forest Ecosystem Project experiment. Li et al. (2012) reported a significant difference in the various respiration fluxes among the treatments in this study. Altogether, it is clear that these management activities changed not only the total storage and carbon distribution in the forest but also the magnitudes and temporal dynamics of the carbon fluxes.

Landscape management, by definition, will alter the landscape's spatial heterogeneity and will consequently change both carbon pools and fluxes. However, we found only a few manipulative landscape studies that linked structural changes and carbon pools, preventing us from developing sound landscape-level management guidelines that would let managers design the temporal and spatial characteristics of landscape mosaics (Chen et al. 2006). Several investigations concluded that forest fragmentation and the resulting edge effects will produce negative impacts on carbon sequestration (e.g., Dantas de Paula et al. 2011). Therefore, future management should be designed to reduce fragmentation, a recommendation that agrees with the guidelines for conservation of biological diversity (Harper et al. 2005). Nevertheless, our knowledge of how alternative landscape patterns will affect the carbon cycle is still lacking.

6.5 Outlooks

Carbon studies have gained tremendous momentum in the past two decades because of their central roles in many pressing global issues that face society, such as climate change, energy security, shortages of natural resources, and rapid growth of the world's population and the global economy. Forest ecosystems will increasingly play a critical role in these issues, in large part due to the large carbon fluxes and storage in terrestrial ecosystems. Based on our literature review, future research on the carbon cycle in forested landscapes should be strengthened in the following three areas.

6.5.1 Temporal and spatial dynamics of carbon

The carbon cycle in forest ecosystems has been investigated for decades, yet there remain many unknowns about the distribution, temporal changes, and regulatory mechanisms for carbon other than the effects of climate. For example, the distributions and dynamics of carbon in complex terrain are characterized by many small carbon fluxes that are incompletely understood (Fig. 6.1). Limited data and knowledge are available regarding carbon dynamics in some ecosystem components (e.g., deep soils, wetlands, the urban–rural interface, the land–ocean interface, and other critical zones). From a theoretical perspective, the predictions by Odum (1969) about the responses of the carbon cycle after a disturbance have been challenged because of a lack of thorough validation. Although significant progress has been made in genetics, population and community ecology, and carbon cycle science, consensus on the interactions between the diversity of a forest ecosystem and ecosystem function has not been reached. Finally, understanding the carbon cycle more holistically by including indirect drivers and feedbacks should be explored.

6.5.2 Landscape-scale carbon cycles

Our understanding of carbon fluxes and storage at the landscape level has lagged significantly behind our knowledge at ecosystem and landscape levels. This is partially due to the limitations of existing methods and technology, which are both costly and labor intensive. Sound landscape-scale experiments have not been widely pursued; thus testing and validation of the basic concepts and principles of landscape ecology have been inadequate. Although carbon and water fluxes and storage are well coupled in both vertical and horizontal dimensions (Govind et al. 2010, Ju and Chen 2005, Sun et al. 2011), sound estimates of the horizontal flows of carbon as well as their relationship to landscape-scale processes are rare in current models. This lack of a satisfactory landscape-scale perspective is particularly unfortunate because most forests are owned and managed at a landscape level, and fragmentation is on the rise. Innovative proposals that can overcome these scientific and management challenges are urgently needed.

6.5.3 Humans and carbon cycles

The relationships between carbon sequestration and societal issues (e.g., global warming, fire management, urban growth) need to be studied more intensively from a more holistic perspective that couples humans with the natural systems that sustain us. The traditional approach of linking forest management and carbon cycles independently of human influences must be expanded to include functions that are relevant to human society, such as society's needs for carbon management (e.g., stock markets, biological conservation, bioenergy) and conservation of other ecosystem services.

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Chapter 7
Forest landscape change and biodiversity conservation

Santiago Saura, Emi Martín-Queller, and Malcolm L. Hunter Jr.

Abstract Forest landscapes are changing at unprecedented rates in many regions of the world. This may have profound consequences for the diversity and resilience of forest ecosystems and may impose considerable challenges for their management. In this chapter, we review the different types of change that can occur in a forest landscape, including modifications in forest habitat amount, quality, fragmentation, connectivity, and heterogeneity. We describe the conceptual differences and potential interactions among these changes and provide a summary of the possible responses of forest species depending on their degree of habitat specialization, dispersal abilities, and other factors. We review the main current drivers of change in different regions of the world and how they are affecting (often synergistically) forest biodiversity: deforestation, climate change, forest fires, abandonment of rural land, land-use intensification, spread of invasive species, forest management, and the increasing amount of plantation forest. We conclude by providing a summary of recommendations and strategies for mitigating and minimizing the undesirable effects of landscape change on forest biodiversity.

7.1 Introduction

Despite increasing conservation efforts (Rands et al. 2010), global biodiversity, which comprises the diversity of life in all its forms and levels of organization (Hunter and Schmiegelow 2011), has declined in recent decades (Butchart et al. 2010)

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and is projected to continue diminishing throughout the twenty first century (Pereira et al. 2010, Sala et al. 2000). Tropical forests are being converted to other land uses, mainly to agriculture, at high rates, and the remaining forest fragments are increasingly small and isolated. The consequent reduction in populations of many forest species may compromise their persistence in the future, in part because their growing isolation may lead to adverse metapopulation dynamics, and may even generate genetic bottlenecks. The decline in functional connectivity among forest populations is being exacerbated by a worldwide intensification of agricultural practices that makes the matrix in which forest fragments are embedded less permeable. Moreover, as the length of the boundaries between forests and adjacent non-forest lands increases in the landscape, the effective area of suitable habitat for many forest species will be reduced because they are not adapted to conditions found at the forest's edges. Fragmentation of tropical forests, combined with droughts induced by climate change, is also favoring an increased fire occurrence, possibly beyond the limits to which these ecosystems may be resilient. The resilience of fire-prone forest ecosystems (e.g., many Mediterranean and some North American temperate forests) may also be compromised by current and foreseen alterations in their historical fire regime. Another key global process that influences forest biodiversity is climate change. Climate change is expected to trigger shifts in species distribution poleward and upward in altitude, driving a worldwide rearrangement of forest species. Species responses to climate change will be idiosyncratic, especially given novel biotic interactions that may appear or be substantially altered as a result of climate warming. Furthermore, the capacity of forest species to adapt to changing climatic conditions may be curtailed by the aforementioned loss in connectivity. Forest species are already confronted by all these processes and by others, such as a reduced quality of forest habitats around the world, changes in landscape heterogeneity, or invasion by exotic species, leading to a complex set of interactions and synergies among these processes.

In this chapter, we describe how agents of global change influence forest biodiversity from a landscape-scale perspective, with a particular focus on conceptual mechanisms. By understanding these mechanisms, we may be able to anticipate and better avoid potential negative effects on each forest species. The responses to these processes are expected to differ among species, with their vulnerability depending on diverse aspects such as body size, geographical range, dispersal ability, reproductive rate, and niche specialization (Brook et al. 2008). This means that any particular landscape change that may jeopardize some species may also favor other species. In general, forest specialists are expected to be more negatively affected than generalist species by ongoing landscape changes, with a consequent potential homogenization of biota across regions. Overall, the potential future scenarios of global biodiversity loss addressed throughout this chapter provide an argument for the need to adopt political, economic, and social measures to reduce these pressures. For that purpose, we present some general management guidelines in the last section of the chapter.

7.2 Types of change in the forest landscape and their influences on forest biodiversity

7.2.1 Habitat loss and fragmentation: related but conceptually different processes

Habitat can be defined as the resources and conditions present in an area that produce occupancy—including survival and reproduction—by a particular species (Hall et al. 1997). Habitat is therefore species-specific. Habitat loss is the reduction of the amount of habitat for a particular species in a landscape, and therefore negatively affects the abundance of that species, sometimes even causing its disappearance. The habitat for a particular forest-dwelling species may correspond to a specific forest composition and structure (e.g., one or more successional stages or even non-forest vegetation in part of its life cycle). Therefore, habitat loss for a species should not necessarily be associated with the loss of forest cover in general. Nonetheless, forest cover is a critical element for the persistence of most forest species, and analyzing changes in its abundance and configuration is a helpful approach, as we will summarize in Sect. 7.2.5.

A related but conceptually different process is habitat fragmentation, which can be defined as the process through which large and continuous habitat patches are broken apart into multiple smaller pieces that are physically separated from each other (Haila 1999). The potentially negative effects of habitat fragmentation for biodiversity conservation have been widely described (e.g., Fahrig 2003), and are generally grouped in three categories: reduced patch size, patch isolation, and edge effects.

Habitat patches in the landscape become smaller with increasing fragmentation. From the perspective of an individual species, a reduction in the effective population size in smaller habitat fragments increases the probability that a species will go locally extinct, which is known as the "small-population" paradigm (Caughley 1994). Many theoretical and empirical studies have focused on evaluating the minimum number of individuals required for the persistence of a species within a specified timeframe, the so-called minimum viable population size (Shaffer 1981, Traill et al. 2007). Fragmentation may also increase isolation among previously continuous habitat patches as they become separated by unsuitable areas and as the distance between them increases. The small-population paradigm has traditionally assumed that populations are isolated. Yet both island biogeography theory and metapopulation theory (Hanski 1999) highlight the possibility that a small population can persist through immigration of individuals from other populations in the surrounding landscape. This has led to a more recent approach based on considering all of the populations in a landscape through the use of the "minimum viable *metapopulation* size" concept (Bulman et al. 2007). In short, the negative impact of habitat loss on biodiversity may be attenuated to some degree when functional connectivity is maintained. Theoretical studies predict that the extinction threshold will be reached later in the gradient of shrinking habitat amount in less isolated (more connected) sets of patches (Fahrig 2002), something we will discuss in Sects. 7.2.2 and 7.2.5. We will explore the impacts of habitat isolation on forest biodiversity in more detail in Sect. 7.2.2, on landscape connectivity.

Species richness declines with diminishing patch area (size), and this is one of the most consistent patterns in ecology (Begon et al. 2006). Much of the research on the effects of reduced patch size and patch isolation on community species richness has been framed within the theory of island biogeography (MacArthur and Wilson 1967), in which patch area and isolation are drivers of the extinction and immigration dynamics of populations. Apart from the island biogeography framework, many other possible underlying causes for decreases in species richness have been hypothesized, such as parallel decreases in environmental diversity, available energy, the target area for colonizers, or the number of sampled individuals (see Gardner and Engelhardt (2008) and the references therein). This consistent species richness–area pattern has allowed the common use of species–area curves to predict future species extinctions that will follow the loss of forest cover (e.g., Pimm and Raven 2000); we will discuss this in more detail in Sect. 7.3.1, on deforestation.

The “edge” of a habitat patch is the portion near the patch’s perimeter. Many forest species avoid forest edges or have lower population densities near them. These patterns, called “edge effects”, are driven by a variety of factors such as increased predation risk, modified microclimates, more intense human disturbances, and higher competition with generalist species at the patch edges than in the core areas (Gonzalez et al. 2010, Laurance et al. 2006). The distance from the border reached by edge effects is species dependent, but as forest fragmentation proceeds, all edge-sensitive species will begin to suffer from larger reductions in the area of their effective core habitat rather than in the total amount of forest in the landscape. Examples of edge-sensitive species include many lichens (e.g., Rocío et al. 2007) and bryophytes (e.g., Löbel et al. 2012), but specific cases are common for all taxonomic groups (e.g., vascular plants, birds, mammals).

Forest area loss and fragmentation are recognized as the main factors behind decreases in forest biodiversity, but disentangling their relative importance is not easy. Both changes usually occur simultaneously through the processes of deforestation and habitat degradation (the change from A, through B and C, to D in Fig. 7.1). This has frequently led to an overestimation of the actual effects of forest fragmentation on species persistence. Imagine, for example, that ten forest species were found in landscape A in Figure 7.1, and that the change process that goes from A to D in that figure (with B and C as intermediate stages) would have reduced the number of species to only two. This has been interpreted in many cases as the basis to conclude that forest fragmentation has caused the loss of 80 % of the original species richness (i.e., a loss of eight species). However, in the process of changing from A to D, fragmentation has not been the only important change; the amount of habitat has also decreased greatly for many forest-dwelling species. It would be interesting to know how many forest species would have been lost if a different change process had occurred (such as from A, through E, to F in Fig. 7.1); that is, if the same amount of forest area had been lost but with no fragmentation occurring (stage F, in

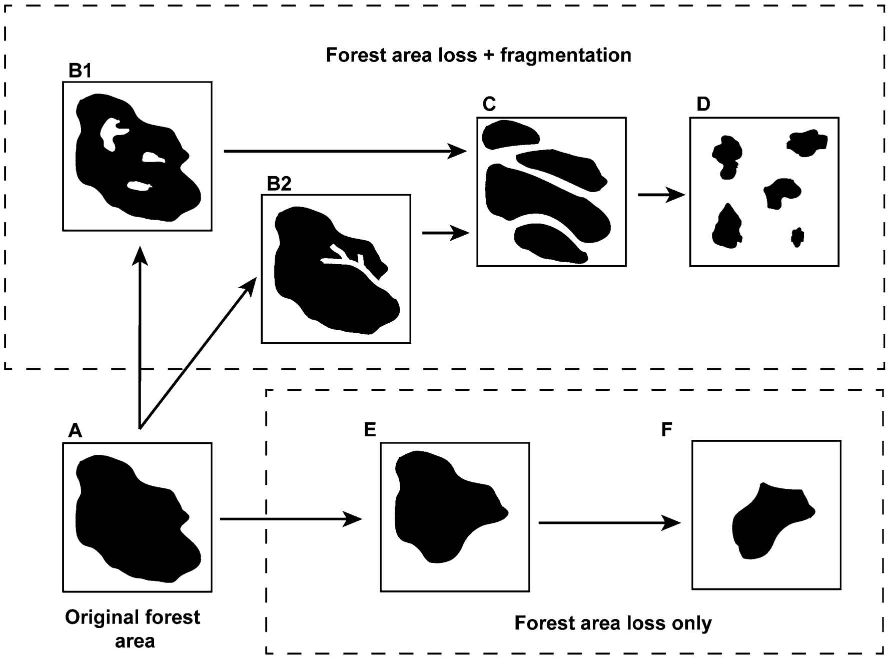

Figure 7.1 Changes in the amount and spatial arrangement of forest cover that can occur as a result of forest area loss and fragmentation processes. The upper box (changes from A, through B and C, to D) illustrates the typical progression of a process with combined loss of forest area and fragmentation, whereas the lower box (changes from A, through E, to F) corresponds to the case in which forest area is lost without causing any breaking apart of the remnant forest. B1 and B2 are two alternative possibilities for the typical spatial changes that would occur before separated patches are produced during the change from A to D. The final stages in both cases (D for the upper box and F for the lower one) have the same amount of forest area, but with a different spatial arrangement

which the total forest area is the same as the total in stage D). If, for example, four of the original ten forest species were found in stage F, this would mean that habitat loss alone has been responsible for the loss of six species, whereas the impacts of fragmentation per se have only caused the loss of two additional species (i.e., the difference in species richness between stages F and D). For habitat fragmentation to happen, some habitat loss needs to occur, even if this is only a small amount; for example, if the incisions in landscape B2 in Figure 7.1 continue to progress, they would break the forest into several separated patches with a relatively minor reduction in total forest area. However, the opposite is not true, since habitat loss can happen without any fragmentation or breaking apart of habitats (as in the change from A, through E, to F in Fig. 7.1). A meta-analysis by Fahrig (2003) showed that habitat loss has more prominent and consistent detrimental effects on biodiversity than habitat fragmentation. Forest fragmentation can indeed have important negative effects on biodiversity (e.g., Laurance et al. 2006) but, in general, fragmentation will be responsible for only a part of the total impacts on species and populations. For forest landscape management planning, it will be useful to disentangle the relative importance of these two processes for a particular species; that is, it is important to learn whether the focus should be placed more on the spatial configuration of the habitat patches or more on the total amount of habitat in the landscape.

7.2.2 Landscape connectivity

Based on Taylor et al. (1993), landscape connectivity can be defined as the degree to which the landscape facilitates movement among the existing habitat resources for a given species. Managing landscape connectivity is a key part of forest biodiversity conservation, as it is considered to be one of the best strategies for counteracting the adverse effects of fragmentation and facilitating shifts in species ranges in response to climate change (Araújo and Rahbek 2006, Opdam and Wascher 2004, Taylor et al. 1993).

Fragmentation and connectivity loss are related, but different, concepts. Fragmentation is a structural property in which patches of habitat are subdivided and physically separated from each other, and can be measured and assessed without considering the dispersal abilities of any particular organism. In contrast, landscape connectivity is a functional, species-specific property that depends on the dispersal abilities and behavioral traits of a given species (Theobald 2006, Tischendorf and Fahrig 2000). A given landscape might be perceived as strongly connected for an organism able to traverse large distances (e.g., a bird species), whereas it might be weakly connected for another species dwelling in the same landscape that only disperses over short distances, that lacks the ability to move through the land cover types in the landscape matrix that separates its habitat areas, or a combination of both (e.g., an amphibian). Fragmentation can occur without an impact on the connectivity among remnant patches; for example, for a bird species with a high movement ability, all the patches in landscape D in Figure 7.1 may still function as a single fully connected unit. On the other hand, connectivity losses can occur even with no additional habitat fragmentation. This will occur when a given landscape change does not directly affect the area of habitat, but impedes the dispersal of a species between habitats due to increased resistance to dispersal in the landscape matrix (e.g., as a result of road construction, urban development, or intensification of agriculture).

The concept of connectivity has often been associated with the presence of corridors, which are conceived as narrow, elongated strips of vegetation that physically connect larger blocks. However, the options to promote landscape connectivity go well beyond the maintenance or establishment of corridors. Ecological fluxes among habitat areas can also occur in a more diffuse but equally effective manner through wide stretches of a permeable non-habitat landscape matrix or by means of successive short-range movements facilitated by a series of stepping-stone habitat patches, such as small woodlots, or even single trees scattered throughout the landscape (Adriaensen et al. 2003, Lindenmayer et al. 2012, Manning et al. 2009, Rey Benayas et al. 2008, Uezu et al. 2008, With et al. 1997).

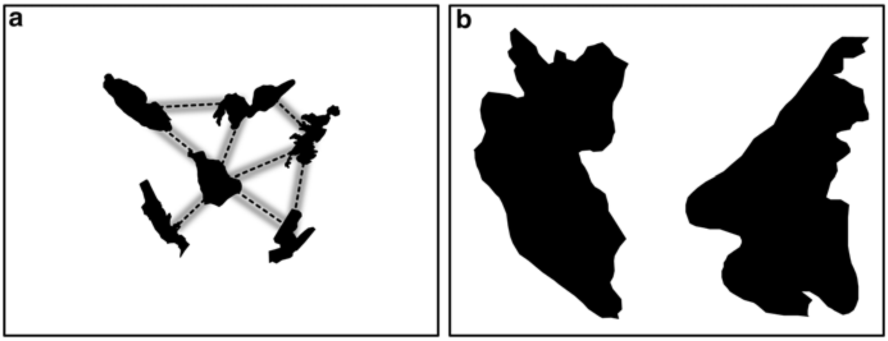

Figure 7.2 Two simple hypothetical landscapes (**a**, **b**) with different sets of habitat patches (shown in *black*) and links (direct connections) between them (shown as *dashed lines* with *grey* shadows) for a given species to illustrate the concept of habitat availability (reachability) at the landscape scale (see the main text for details). Links represent functional connections between the patches; that is, they represent the ability of a given species to move between patches, and may correspond to the existence of a corridor, of a permeable landscape matrix that makes movement of a species possible, or of a series of stepping stones that facilitate dispersal between source and destination habitat patches. Adapted from Saura (2008)

One of the classical definitions of landscape connectivity was provided by Taylor et al. (1993): "the degree to which the landscape facilitates or impedes movement among resource patches". This definition suggests that landscape connectivity can be successfully addressed and managed by considering only the number and quality of the connections among habitat patches. However, an approach that focuses only on the connections between habitat patches (interpatch connectivity) can mislead conservation managers when it deals with landscape changes that affect both the size and the spatial configuration of the patches. Consider the two landscapes in Figure 7.2, which shows the distribution of habitat patches and the links (functional connections) among them for a given focal species. Which landscape is more connected? It may seem obvious that connectivity is higher in **a** than in **b**, because in **a** there are eight links between patches, whereas in landscape **b** there are none. However, from a management perspective it makes no sense to consider **a** as more connected than **b** because no matter how well connected the patches are in landscape **a**, collectively they comprise less available (reachable) habitat than the area in only one of the patches in landscape **b** (Pascual-Hortal and Saura 2006, Saura 2008). In other words, a big isolated patch in **b** comprises a larger area of connected habitat within itself than all the area that can be reached through all the links in landscape **a**. As noted by Tischendorf and Fahrig (2000), some connectivity metrics suffer from the problem of indicating higher connectivity in more fragmented landscapes and zero connectivity in any landscape containing just one habitat patch, even if that habitat patch covers the whole landscape.

To provide an appropriate measure of landscape connectivity in changing landscapes: (1) the amount of connected habitat within habitat patches has to be considered (bigger patches have more intrapatch connectivity), even when the patches are completely isolated from all other patches, and (2) intrapatch connectivity must be considered along with the area made available by the connections with other habitat patches (interpatch connectivity). This is the concept of habitat *availability* (reachability) at the landscape scale (Pascual-Hortal and Saura 2006, Saura 2008, Saura and Pascual-Hortal 2007, Saura and Rubio 2010). Fundamentally, it means that connectivity should be considered as a landscape property that allows a particular species to reach a larger amount of habitat resources, no matter if these resources are provided by a single big patch (intrapatch connectivity), by the connections between different patches (interpatch connectivity) or, more frequently, by a combination of both. If connectivity is relevant for management, this is because it increases the amount of habitat that can be reached by a particular species in the landscape, not because it increases the number of connections between increasingly smaller and poorer habitat patches (as in landscape **a** in Fig. 7.2).

New metrics have been proposed that are derived from this way of conceiving and measuring connectivity (Pascual-Hortal and Saura 2006, Saura and Pascual-Hortal 2007), and they have been implemented in the Conefor software (http://www.conefor.org) and widely applied to support landscape connectivity conservation management in different countries. Among these, the equivalent connectivity area (*ECA*) is an intuitive and useful metric that is defined as the size that a single habitat patch should have in order to provide the same amount of reachable (available) habitat (i.e., connectivity) as the mosaic of habitat patches in a given landscape (Saura et al. 2011a, b). *ECA* will be equal to the total area of habitat in the landscape (A) for a particular species when either all of the habitat is concentrated in a single continuous habitat patch or when the habitat is dissected into different patches but the probability of movement between any two patches is equal to 1 for that species. With this approach, it is possible to directly compare the relative change in *ECA*, d*ECA* = (final − initial)/initial, with the relative change in the total amount (area) of habitat in the landscape, dA = (final − initial)/initial, after a given landscape change (Fig. 7.3). This allows an assessment of the degree to which a given change in the total amount of habitat would be beneficial or detrimental for ecological connectivity. For example, a net decrease in the total amount of habitat (dA < 0) may translate into a higher, lower, or equal loss of connectivity as measured by d*ECA*, as illustrated in Figure 7.3 (respectively) by the cases in which d*ECA* < dA < 0 (higher loss in the amount of reachable habitat than in the total habitat area), dA < d*ECA* < 0 (higher loss in the total habitat area than in the amount of reachable habitat), and d*ECA* = dA < 0 (both magnitudes decrease at the same rate, corresponding to a purely proportional effect of habitat loss).

Figure 7.3 (continued) analyzed period ("stable forest"), be lost due to conversion to other cover types ("forest loss," in which initially forested areas are no longer forested at the end of the period), or expand as a result of afforestation ("forest gain," in which areas that were not forested in the past are covered by forests at the end of the period). The six types of change correspond to the evolution of real Spanish landscapes, as they were selected from some of the samples in the SISPARES monitoring system (http://www.sispares.com). The figure was adapted from Saura et al. (2011b). See Saura et al. (2011a, b) for further details

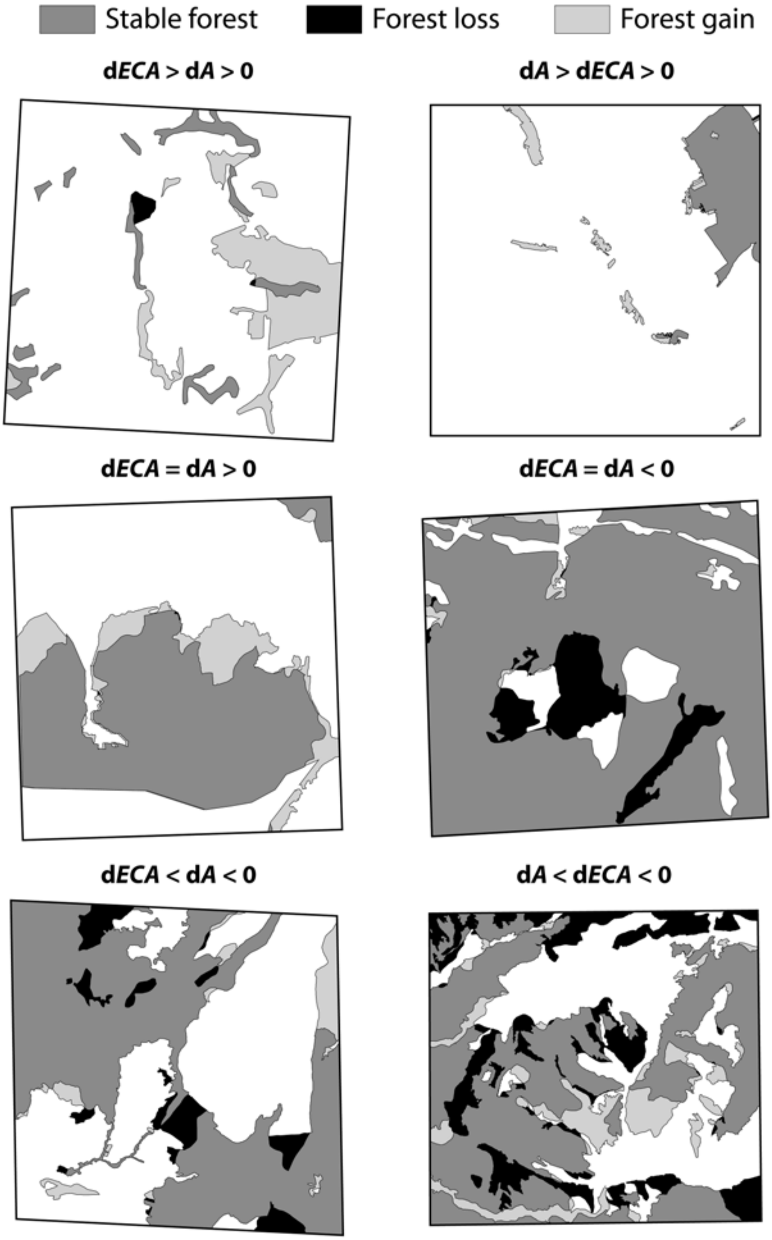

Figure 7.3 Six different landscapes (each covering 4 × 4 km) that illustrate the different ways in which the change in the total amount of habitat (d*A*) in a given time period can translate into a higher or lower change in the connectivity of the habitat in the landscape (d*ECA*), which is measured here by the equivalent connected area (*ECA*) for a species with a median dispersal distance of 200 m (relative to the landscape extent of 4 × 4 km). The examples in this figure assume that forest is the focal habitat. The areas occupied by forests may either remain stable during the

7.2.3 Habitat quality in the forest landscape

Even if habitat patches in the landscape are not completely eliminated or reduced in size, significant population declines or even species losses can occur due to the reduction in their quality as a result of natural disturbances or, more frequently, human interventions such as logging, grazing of livestock, or hunting. For example, for many forest specialist species, the abundance of elements characteristic of old-growth forests is an indicator of a forest's quality as habitat (e.g., Grove 2002, Lindenmayer et al. 2012). These elements include thick stems, dead wood in a range of diameter classes, an uneven-aged structure, and vertical (multilayer) or horizontal (spatial) heterogeneity. However, because habitat quality is, by definition, species-specific, the modification of some habitat characteristics may impair some species while favoring others. For example, microclimatic changes after harvesting may be detrimental to shade-tolerant plant species, whereas the increased availability of ground vegetation associated with early successional stages would benefit other organisms such as large herbivores.

For many ecologists and environmentalists, the quality of a forest ecosystem is largely determined by its degree of naturalness. This perspective requires an ecological baseline; that is, it requires historical information about the conditions under which the ecosystem developed. However, human influences on ecosystems may be difficult to disentangle from natural ones, especially in regions such as Europe, where centuries of human land use have left a deep footprint (see Hermy and Verheyen 2007 for a review; Rozas et al. 2009). For example, soil nutrient levels and the species composition in afforested patches can be influenced by former agricultural land use (Hermy and Verheyen 2007), and these changes have been observed to last as long as 2000 years (Dupouey et al. 2002). Even the use of pre-European conditions in North America as archetypes of pristine ecosystems has been criticized, since this probably underestimates the role of aboriginal peoples in shaping the landscape (Alagona et al. 2012). This situation is compounded by the dynamic and changeable nature of ecosystems even in the absence of human interference (Alagona et al. 2012). Furthermore, future uncertainty due to rapidly changing climatic and environmental conditions further challenges the search for an ecological reference to define the generic quality of a particular forest ecosystem.

7.2.4 Forest landscape heterogeneity

Different sets of species are associated with particular forest types and land covers, or combinations of types and covers. Therefore, it will generally be the case that heterogeneous forest landscapes, which comprise multiple forest and non-forest types, are able to harbor a relatively large number of species. Indeed, the importance of spatial heterogeneity for diversity has been long recognized as one of the central concepts in landscape ecology. The increase in species richness with landscape heterogeneity might be due to (1) a higher gamma diversity resulting from nonoverlapping sets of specialist species being present in different land cover types (beta diversity); (2) the fact that some species use resources from different cover types (generalist or heterogeneity-dependent species), such as raptors that nest in forests but forage in adjacent pastures; or (3) a combination of the two.

However, increasing heterogeneity is not necessarily beneficial for biodiversity conservation in many situations. Landscape heterogeneity cannot be increased without reducing the extent of some land cover types within the landscape, and this may or may not be a desirable outcome. The potential benefits of heterogeneity depend on the conservation value of the affected forest or cover types and their associated species. In fact, some regions of the world that are undergoing biodiversity loss are at the same time experiencing considerable increases in landscape heterogeneity, because they are in the initial stages of forest cover loss and fragmentation. These regions are shifting from large areas covered by primary, species-rich forests (which may be regarded as relatively homogeneous landscapes) to landscapes in which heterogeneity is increased by a variety of new cover types such as pastures, cropland, and urban areas. The assumed benefits of landscape heterogeneity may vanish when the conservation status of each species, and not just the total number of species, is taken into account. There is the risk of favoring generalist, cosmopolitan species by promoting landscape heterogeneity as a general management principle. Any landscape-scale change has losers and winners, and it is the identity and particular status of each of the affected species that should determine whether a particular type of change should be promoted in a given forest conservation management plan.

7.2.5 Responses of forest species abundance and diversity to landscape change: a summary of scenarios

In this section, we integrate the different types of landscape change described in previous sections to provide a conceptual summary and a comparative assessment of their potential impacts on the abundance of a particular forest species. We mainly borrow from Andrén et al. (1997) and Brotons et al. (2005), with several adaptations and additions for the purposes of this chapter. We consider different scenarios regarding the potential responses of a species to landscape change, depending on the species' habitat requirements, on its dispersal abilities, and on the dominant type of landscape change. Since the response of an entire forest species community to landscape change will be the aggregated result of the response of each individual species (and of the interactions among them), this conceptual synthesis will also help to predict how community richness might be affected by habitat loss, fragmentation, reduced landscape connectivity, decreases in habitat quality, or variations in landscape heterogeneity.

7.2.5.1 Potential responses of a forest specialist species to landscape change

First, we consider the case of a strict forest specialist species, which has a particular forest type (hereafter referred to as "F" for simplicity) as its only habitat in the landscape. It therefore follows that the maximum population levels for this species will be found when the entire landscape is occupied by F. In this case, there are three possible species responses to habitat loss:

– *A purely proportional effect of habitat loss (scenario S1 in Fig. 7.4).* Species abundance responds linearly (and in exact proportion) to the reduction of the cover of forest type F in the landscape because the population size is limited only by the total amount of habitat. There are no other additional impacts that compound habitat loss, even for low levels of habitat cover, such as those that might arise from habitat fragmentation. This might be the case for species that are largely insensitive to edge effects and that are able to satisfy their vital needs even in disturbed landscapes that retain a small amount of habitat area. These are typically species with small home ranges and small body masses, such as rodents.
– *A critical threshold in the habitat amount (scenario S2 in Fig. 7.4).* This is characterized as an abrupt decline in population size when the amount of habitat in the landscape drops below a certain threshold, as has been reported in simulation studies (With and King 1999) and, to a lesser extent, in empirical landscape studies (see Swift and Hannon 2010 for a review) for species such as the northern spotted owl (*Strix occidentalis caurina*) or the white-backed woodpecker (*Dendrocopos leucotos*). The proportional response in scenario S1 might hold until the amount of forest type F that remains in the landscape falls below a certain level, after which the species might go extinct or suffer from an abrupt decline. (See Sect. 7.2.1 for additional discussion.)

If the species is highly mobile and readily ventures through the non-habitat matrix, it might perceive all of the forest type F in the landscape as a single functionally connected unit. In this case, the amount of available (reachable) habitat in the landscape will be the same whether or not the landscape change corresponds to a pure habitat loss (the change from A to F in Fig. 7.1) or to the case in which fragmentation occurs with the remnant patches getting separated from each other (the change from A to D in Fig. 7.1). The critical threshold would therefore occur for much the same amount of habitat in both change types, assuming that no other compounding effects (such as an increased edge influence in the fragmentation case) affect the focal species. However, if the species is unable to move through cover types that differ from F, the available habitat area will correspond only to the size of the occupied habitat patch. In this case, an abrupt response could occur with just a small additional loss of forest type F if that loss fragments the habitat into several patches that are all smaller than the minimum viable size. In this case (low connectivity), an abrupt decline in population will appear earlier in the fragmentation process (during the change from A to D in Fig. 7.1). Therefore, the effects of habitat fragmentation and reduced connectivity may intensify the effect of these critical thresholds.

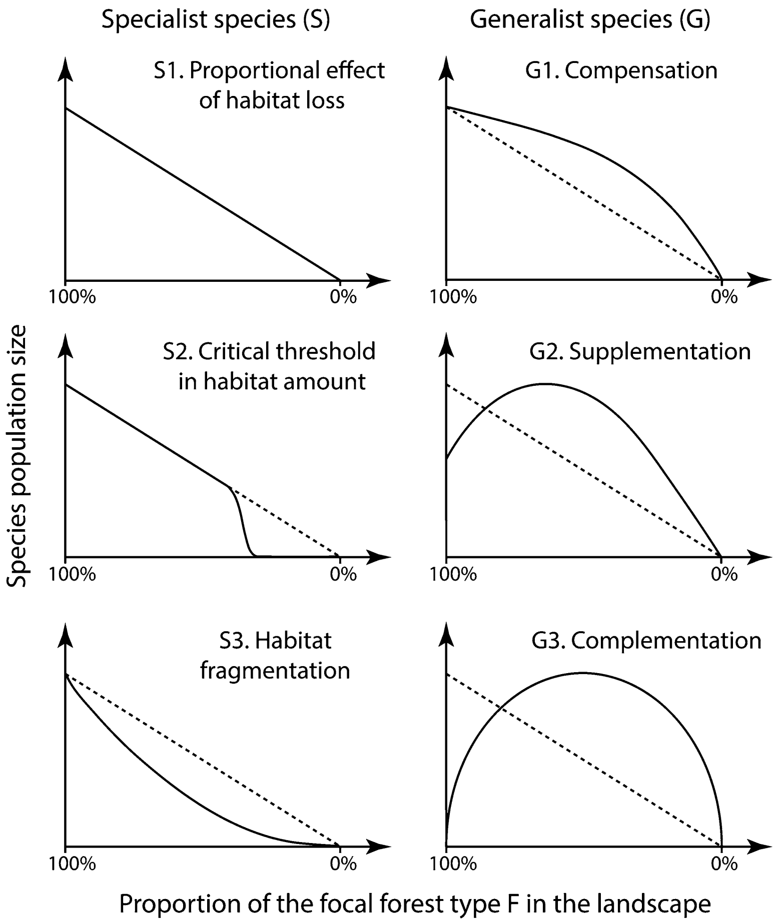

Figure 7.4 Hypothetical scenarios for the response of the population of a forest species to the loss of varying amounts of a given focal forest type F. The species is assumed to have F either as its only habitat (specialist species; S scenarios) or as a part of the habitat resources it requires to fulfill its vital needs (generalist species; G scenarios). The *y*-axis shows an arbitrary scale ranging from the maximum attainable population size to the absence (extinction) of the species in the landscape. The response curve for scenario S1 is added as a reference (*dashed line*) in all the other scenarios. See Sect. 7.2.5 of the text for a description and discussion of the six scenarios and their relationships to different species traits and types of landscape change. Adapted from Andrén et al. (1997) and Brotons et al. (2005)

However, such thresholds may also appear in the absence of fragmentation (during the change from A to F in Fig. 7.1; Swift and Hannon 2010).

– *Habitat loss has an amplified impact on the population size due to fragmentation and reduced connectivity (scenario S3 in Fig.* 7.4*).* A given percentage loss of forest type F translates into a proportionately larger reduction in species abundance, so the population size falls below the linear response depicted in scenario S1 (Fig. 7.4).

This is typically the case for fragmentation-sensitive species. Even in the early stages of the fragmentation process, when only a small amount of habitat has been lost and the forest has not yet broken apart (e.g., the perforations in B1 or the incisions in B2 in Fig. 7.1), the core area (the area away from the edges of the patch) will be reduced much more than the total patch area, leading to a comparatively large population reduction for edge-sensitive species. (For more discussion of edge effects, see Sect. 7.2.1.) In addition, some of the fragment sizes may fall below that required to support the minimum viable population size for some species. If a species has poor dispersal ability or if the habitat is embedded in a non-permeable matrix, the lack of functional connectivity will produce a larger decrease in the usable and reachable habitat area than the actual decrease in the total habitat area (e.g., see the case $\mathrm{d}ECA < \mathrm{d}A < 0$ in Fig. 7.3).

Typically, some fragments below the minimum viable size will be produced all along the gradient of habitat loss and fragmentation, and therefore such amplification of the impacts of habitat loss is likely to translate into a continuously decreasing response curve that falls below the linear change in scenario S1. However, the proportion of the habitat that falls below the minimum patch size may typically increase for lower levels of total habitat amount, and the increase in the distance between patches will generally be larger in landscapes where habitat is already sparse (Andrén 1994), making dispersal limitations translate more easily into effectively isolated populations in individual patches. This could lead, in the extreme, to the sharp critical threshold described in scenario S2, although it will probably lead to a milder response that lies somewhere between scenarios S2 and S3 (Fig. 7.4). Amplified impacts of habitat loss might also occur due to a reduced permeability in the landscape matrix (e.g., from agricultural intensification or road construction). Extinction debts (time lags in species responses) may also make the response curves vary from S1 to S2 or S3 over time (Tilman et al. 1994), even when no additional changes in the habitat or landscape occur.

7.2.5.2 Potential responses of forest generalist species to landscape change

In the previous three scenarios, we considered the case of a hypothetical forest specialist species that had forest type F as its only habitat. Now we relax that assumption to incorporate the effects of landscape heterogeneity on species responses. We now consider a generalist species that requires some amount of the focal forest type F to fulfill its vital needs, but that can also use resources in other forest or cover types. In this case, complete loss of the focal forest type F would lead to disappearance of the species, but it is not clear how its population size would react to smaller changes in the amount of F. This will depend largely on the species' traits and on the characteristics of the other forests or land covers to which F is converted. We can, however, conceptually differentiate three distinctive responses and link these to different ways in which particular changes in landscape heterogeneity could affect forest biodiversity:

- *Landscape compensation* (scenario G1 in Fig. 7.4). In this scenario, type F is converted to other forest or cover types that have some resources suitable for the species, although in smaller quantities or with lower quality than in F. Thus, they can partially compensate for the loss of focal forest F, and the population size would decrease, but at a lower rate than the amount of F that is lost; that is, the response curve would remain above the linear (proportional) response of scenario S1. Examples of compensation have been found, for example, for some bird species that had natural steppes as their primary habitat but that could potentially use extensive pastoral habitat (Brotons et al. 2005). This scenario would also apply when a decrease in the *quality* of F, rather than complete disappearance of this type, occurs. (See Sect. 7.2.3 for more discussion of the effects of habitat quality.) Depending on the amount of resources in the areas where F has been lost or degraded, higher or lower compensation levels will occur, with the response curve for scenario G1 in Figure 7.4 falling either farther from or closer to that for scenario S1.
- *Landscape supplementation* (scenario G2 in Fig. 7.4). This will occur when the other forest or cover types that substitute for F present some additional valuable resources for the species that are not available in F (Dunning et al. 1992). The species will therefore increase its resource intake in landscapes where both forest or cover types coexist, benefiting from that heterogeneity more than in the case in which only F is present in the landscape. In the study by Brotons et al. (2005), some steppe bird species were benefited by the presence of nearby pastures, probably due to the increased abundance of insects. However, at some point, loss of F is so large that the disadvantages outweigh the benefits, and the population of the species begins to decline again.
- *Landscape complementation* (scenario G3 in Fig. 7.4). Some species may be unable to find all the resources they require to complete their life cycle (e.g., foraging areas, shelter areas, winter roost sites, breeding patches) in a single cover type. Such species would depend on the combined presence of different forest and cover types, each providing different, complementary, and non-substitutable resources (Dulaurent et al. 2011, Dunning et al. 1992). Therefore, a landscape dominated only by F may not provide all the required habitat for some forest species, and the conversion of F into other forest or cover types, or their coexistence, would represent the optimal landscape setting for these mosaic- or heterogeneity-dependent species. As an example, many large herbivores or game birds such as *Perdix* spp. need to complement food resources from open areas with the proximity of forests that provide refuge (Choquenot and Ruscoe 2003). However, as in the supplementation scenario, there is a point at which the loss of F produces disadvantages that outweigh the benefits from a more diverse landscape, resulting in a decline in species population size.

Whether any of these effects occur (compensation, supplementation, complementation), and the intensity of the effects, will depend not only on the amounts of the different forest and cover types in the landscape but also on the ability of a species to move among patches. For species with movement limitations in a given landscape, the complementary or supplementary resources that may exist in other forest or cover types that are different from F may only be reachable when the different patch types are distributed in close proximity to each other; that is, there must be a fine-scale mixture between the different covers. For the same reason, reductions in the permeability of the landscape matrix, even when they do not directly affect the area or quality of any of the cover types that are used as habitat by the species, may bring any of the generalist response curves closer to the curve in scenario S1 (Fig. 7.4) or even below that curve if species are benefited by (G2) or require (G3) those other difficult-to-reach resources.

The combinations of different types of change occurring in the landscape and their interaction with the particular traits of a species will determine the final response of each forest species, or of the total forest species richness, in the landscape. The resulting response might be close to one of the six idealized response scenarios in Figure 7.4, but it is more probable that it would fall somewhere between these scenarios due to the aggregated result of the different processes described earlier in this Section.

7.3 Major contemporary processes driving forest landscape changes and their impacts on forest biodiversity

7.3.1 Deforestation

Forest loss (deforestation) is a key driver of current global biodiversity loss. Current rates of forest conversion, mainly to agricultural uses, have raised the alarm about the future of biodiversity, particularly in the tropics (Bradshaw et al. 2009a, Laurance 2007), with the expected effects even larger than those predicted as a result of future climate change (Sala et al. 2000). Although tropical forests represent only 7 % of the Earth's land surface (Bradshaw et al. 2009a), an important portion of global biodiversity depends on the persistence of tropical forest habitats (Dirzo and Raven 2003); therefore, deforestation in these regions is an important threat to global biodiversity. Many studies have estimated high rates of species extinctions in tropical forests based on rates of forest loss combined with species–area curves; for example, see some of the figures and references in Bradshaw et al. (2009a), Brook et al. (2008), and Laurance (2007). However, the wide discrepancy between predicted rates at a global scale and those that have actually been recorded (much lower) has fostered a debate about how to explain this disagreement (He and Hubbell 2011, Ladle 2009). Although there seems to be a better agreement at local and regional scales (Fattorini and Borges 2012), the species–area relationship, even when based on endemic species from the area of forest being destroyed, only estimates instantaneous extinction. However, many species that survive under suboptimal conditions in small forest fragments may already be committed to extinction when the conditions for their reproduction are no longer met—the so-called extinction debt (Tilman et al. 1994). A more realistic framework that includes potential extinction has recently been introduced to account for these issues (Tanentzap et al. 2012). Finally, it is important to note that, even if forest is restored, ongoing anthropogenic disturbances in deforested areas are expected to have a legacy effect on their habitat quality (see Sect. 7.2.3), for example, through an accumulation of persistent pesticides.

Deforestation often results in fragmentation of the remaining forest. (For a more detailed discussion of fragmentation, see Sect. 7.2.1.) The species composition and structure of remnant forests have been reported to differ from those of previously continuous forests in many studies (e.g., Benedick et al. 2006, Filgueiras et al. 2011, Watson et al. 2004). Furthermore, some authors warn about the convergence of species composition in small forest fragments to the composition of communities adapted to early successional vegetation, inducing the replacement of forest interior (edge-sensitive) species by generalist, disturbance-tolerant species across the landscape (e.g., Laurance et al. 2006, Lôbo et al. 2011). Although this speaks to the irreplaceability of large, continuous forests, some authors also highlight the conservation value of networks of small fragments, given their potentially substantial contribution to landscape-level biodiversity through increased heterogeneity (e.g., Bell and Donnelly 2006, Struebig et al. 2008) and functional connectivity (but see Sects. 7.2.2 and 7.2.4).

7.3.2 Abandonment of rural land

In some regions of the world (and particularly in some developed countries), the abandonment of agricultural land and of forest harvesting practices are leading to the encroachment of shrub and forest communities, although globally this process does not compensate for deforestation. Natural forest regeneration in marginal agricultural land and forest maturation represent an opportunity for the recovery of forest-dwelling species, as has been shown for birds (e.g., Gil-Tena et al. 2009, Preiss et al. 1997, Sirami et al. 2008). Moreover, reforested and more mature patches are expected to improve the connectivity among natural or seminatural forests. (For more discussion of connectivity, see Sect. 7.2.2.) In contrast, open-habitat species tend to disappear from abandoned agricultural landscapes as ecological succession occurs and openings disappear (e.g., Moreira and Russo 2007, Sirami et al. 2010), and farmland specialists are particularly vulnerable to this land-use change (Sirami et al. 2010). These changes might therefore be negatively affecting those species that are associated with the historical agriculture–forest mosaic (Blondel and Aronson 1999, Katoh et al. 2009, Scarascia-Mugnozza et al. 2000); in regions such as the Mediterranean, this mosaic hosts a significant portion of the endemic wildlife.

7.3.3 Climate change

Human-driven climate change during the twentieth century has already induced broad biological changes and represents a looming threat for biodiversity (Parmesan and Yohe 2003, Root et al. 2003), but to date, these changes have been small compared to those that have been driven by habitat loss (Parmesan and Yohe 2003). The global average tendency toward increased warming is projected to trigger the displacement of species ranges poleward in latitude or upward in elevation in response to the need of species for suitable climatic conditions. This rearrangement of species distributions has important implications. The dynamics of populations that inhabit the latitudinal margins of a species' range will be critical for its fate (Hampe and Petit 2005). For instance, northern populations of boreal forest species are projected to move into the Arctic tundra (Pereira et al. 2010), although some boreal tree species may be unlikely to find new areas with suitable conditions and will subsequently exhibit contraction of their range (Thuiller et al. 2006). As for populations that inhabit the low-latitude margins of a species' range, global niche-based models forecast a bleak future. However, in mountainous regions, many low-altitude populations may be able to persist through altitudinal shifts, as has been documented for many species during the Pleistocene (Bush et al. 2004, Hampe and Petit 2005); this will not be possible, however, for many high-altitude populations. In the case of tropical forest species, concern has been raised about the absence of species that are able to replace those species currently distributed in tropical lowlands given that these species are already living near the thermal optimum of their functional niche (Colwell et al. 2008). Finally, movements of a species up altitudinal gradients may also result in declines in population sizes, because the area of an altitudinal band diminishes with increasing elevation because of the typical conical shape of a mountain. (The species richness–area relationship is discussed in Sect. 7.2.1.)

Bioclimatic envelope models (niche-based models) have been used to assess the impact of climate change on biodiversity (Heikkinen et al. 2006). Projected range contractions combined with empirical species–area relationships, which are also used to predict the impact of deforestation (Sect. 7.3.1), have provided scenarios of the future potential extinction risk that is attributable to climate change (e.g., Thomas et al. 2004). But apart from the direct physiological constraints of projected warmer temperatures and protracted drought, the decline in overwinter mortality of some insects (e.g., Hódar et al. 2003, Kurz et al. 2008) and the weakening of some organisms by these constraints (Breshears et al. 2005, Pounds et al. 1999) may result in mass-mortality events (e.g., insect or disease epidemics). However, other aspects of vulnerability such as the sensitivity or adaptive capacity of a species should be considered apart from its exposure to climate change (Dawson et al. 2011). For example, phenotypic plasticity in response to climate change has already been reported for many species (Parmesan and Yohe 2003, Root et al. 2003), and the potential for microevolution may allow adaptation to new climatic conditions (Dawson et al. 2011, Malhi et al. 2008).

7.3.4 Forest fires

The implications of ongoing changes in forest fire regimes for biodiversity differ markedly among regions in the world because of differences in the type of fire regimes under which forest communities have evolved (Sousa 1984). For example, fire-prone forests such as those in the Mediterranean region have been subjected to the selection pressure of fire disturbance for millions of years, thus favoring the evolution of adaptive traits such as serotinous cones and high resprouting capacity (Lavorel 1999). In contrast, in tropical moist forests, fires have been a weak evolutionary force (Barlow and Peres 2004) and they are only recently becoming common due to a combination of climate change and human-induced ignitions (Malhi et al. 2009). The lack of fire adaptation by many tropical tree species may decrease their chances of survival if fire frequency increases (Malhi et al. 2008, 2009).

The dynamic equilibrium in the fire disturbance and succession cycle of fire-prone forests can be disrupted when a threshold of fire intensity, frequency, duration, or extent is exceeded. If this happens, disturbances can carry ecosystems into a different stable domain (Beisner et al. 2003, Holling 1973). In this context, the Mediterranean region's current tendency toward increasing fire frequency and extent (Pausas 2004) may compromise the persistence of fire-vulnerable species, and trigger different successional pathways, ultimately changing the structure and composition of the regional forest ecosystems (e.g., Pausas et al. 2004). For instance, short intervals between consecutive fire events (e.g., in the Mediterranean Basin) might prevent the regeneration of long-lived species with long prereproductive cycles (Whelan et al. 2002). Conversely, long intervals, which are more frequently found in North America, may limit species that rely on fire disturbance for their reproduction. Fire frequency also determines other structural aspects such as the presence of deep litter, logs, or cavities in trees, which are essential for many animal species (Driscoll et al. 2010). Fire extent is also an issue because small to medium fires may promote landscape heterogeneity, which might allow the coexistence of species with different tolerances of fire disturbance, those typical of different successional stages, or both (Moreira and Russo 2007). (See Sect. 7.2.4 for more discussion of the effects of heterogeneity.)

7.3.5 Plantation forests

The increased worldwide demand for wood products and the growing public concern over the loss or degradation of forests are the major causes of a steady increase in plantation establishment throughout most regions of the world, especially in China (FAO 2007). Recent research has shown that planted forests are usually species poor compared with natural forests (e.g., Armstrong and van Hensbergen 1996, Lindenmayer and Hobbs 2004, Moore and Allen 1999), which is attributable both to the decision to plant monocultures and to the lower structural complexity of the plantations (Brockerhoff et al. 2008); forests with more complex structures support more species by increasing the diversity of niches (e.g., Brokaw and Lent 1999, Ishii et al. 2004).

However, forest plantations encompass a wide range of positive and negative effects on biodiversity, depending on considerations such as the land use being replaced, type of management practices, time since plantation establishment, and landscape context (Brockerhoff et al. 2008). Although plantations have a negative impact on biodiversity when they replace natural forest or non-forest ecosystems, they can contribute to native biodiversity conservation when they replace agricultural land or other intensive land uses. (See Sect. 7.4 for more discussion of this point.) In addition, the conservation value of plantation forests varies broadly as a function of the management practices. Decisions about planting native or exotic timber species, mixed species versus monocultures, the method of land preparation, the abundance of biological legacies (e.g., seed banks, advance reproduction, and vegetative reproductive organs), and rotation length can imply very different scenarios for biodiversity conservation. The structural and compositional characteristics of planted forests can approach, with time, those of other more natural stands, so that they may be able to harbor a large portion of the biodiversity found in those reference natural forests, particularly when appropriate management measures, oriented to habitat quality restoration rather than to intensive timber production, are adopted. Finally, it should be noted that plantation forests can indirectly help biodiversity by satisfying enough of the demand for forest products, so that they alleviate the pressure for more intensive management of the remaining areas of natural forest, which potentially have a higher conservation value.

7.3.6 Management of adjacent non-forest lands

Agricultural land-use intensity is a decisive modulator of the degree of impact that deforestation, fragmentation, or habitat degradation has on forest biodiversity (Ewers and Didham 2006, Kupfer et al. 2006). The dramatic and widespread intensification of agriculture is creating landscapes with sharp contrasts between forests and other land uses in terms of ecosystem structure and microclimate. This intensive land use may constitute a filter for the movement of most forest-dwelling species, whereas less-disturbed non-forested lands surrounding forests can be experienced as permeable by many species, with the consequent beneficial effects of increased connectivity (Sect. 7.2.2). In addition, this contrast is expected to preclude forest organisms in deforested or degraded landscapes from supplementing or complementing their habitats or resources, such as food or shelter (See Sect. 7.2.5 for a discussion of complementation and supplementation from cover types surrounding a forest habitat.). Adverse edge effects could also be mitigated by more sustainable land use in the areas surrounding forest habitats, such as the creation of “softer” edges, with a less drastic transition between ecosystem types. Therefore, the detrimental effects on forest biodiversity of deforestation and fragmentation may be exacerbated by land-use intensification.

7.3.7 Forest stand management

In some regions, the increasing demand for wood products is leading to increased exploitation through timber extraction. Silvicultural disturbances can alter the composition of forest communities and trigger succession dynamics, but their exact impact on species diversity will depend on the frequency, intensity, extent, and duration of the disturbance and on how these factors interact with the characteristics of local species. The response of forest communities to the release of resources (primarily sunlight, water, and nutrients) after harvesting will also depend on the overall levels of resources in the system (Kondoh 2001); that is, the same practice will affect forest biodiversity differently on sites with different productivity (Martín-Queller et al. 2013). Despite these idiosyncrasies, some management practices have been found to benefit the species richness of vascular plants (Paillet et al. 2010a, Torras and Saura 2008). This phenomenon might be explained by the "intermediate disturbance" hypothesis (Connell 1978, Shea et al. 2004), which predicts that the maximum species richness will occur at intermediate disturbance levels. At intermediate levels, plants typical of early successional stages and their associated fauna can survive in canopy gaps and coexist with some shade-tolerant species (Shea et al. 2004). Furthermore, enhancing diversity in the overstory will promote microhabitat heterogeneity that will positively influence diversity of many other organisms that inhabit the forest (e.g., Gil-Tena et al. 2007, Kissling et al. 2008, Sobek et al. 2009, Vesseby et al. 2002). Nonetheless, what is an intermediate disturbance for some species may be severe for others and, therefore, mature, unmanaged stands are essential for the maintenance of species diversity of many organisms. (See Sect. 7.2.3 for more discussion of the effects of habitat quality.) This explains the decline of species richness of bryophytes, lichens, and saproxylic fungi in managed forests compared to unmanaged forests; see Paillet et al. (2010a, b) and Halme et al. (2010) for a review of the effects of forest management on the species richness of different taxonomic groups.

From a landscape perspective, ecological succession in forest communities is continuously fed by colonists from neighboring communities at different stages of the cycle or from different habitats ("metacommunity dynamics"; Leibold et al. 2004). Colonization by species adapted to the conditions in a particular stage of succession will depend on the presence of relatively nearby communities that are in similar stages or that are inhabiting habitat with similar conditions. Therefore, an extensive application of the same type of silvicultural treatments throughout the landscape may result in homogenization of the successional stages, and this may (for instance) hinder the persistence of more demanding shade-tolerant species. The reduction in the exchange of species among communities at different stages may also eventually reduce species richness in each local community (e.g., Martín-Queller and Saura 2013). These considerations emphasize the importance of widening the spatiotemporal scope when evaluating the consequences of any silvicultural operation on local species richness. If we are to protect or restore species richness in a forest, we must have an integral view of the whole span of successional stages in that forest and of the neighboring communities upon which persistence of some species may depend in the long term. Indeed, it may be inappropriate to focus on the biodiversity of an individual forest if doing so leads us to ignore the successional processes at a landscape scale that will naturally and eventually recreate that biodiversity elsewhere in the landscape. The preservation of unmanaged forests intermingled with managed stands in the landscape will help to ensure the persistence of disturbance-sensitive species, while providing sources for recolonization of disturbed sites by species that require disturbance during the successional cycle. In addition, an appropriate infrastructure of dispersal vectors must be assured by, for instance, avoiding excessive removal of the forest understory, since seed-dispersing birds will be attracted by an abundance of fruiting shrubs (García et al. 2010, Tellería et al. 2005). Similarly, the promotion of forest connectivity for mammals may attract species that disperse seeds via zoochory (e.g., Minor and Lookingbill 2010).

7.3.8 Invasive species

Invasive species are exotic species that establish and proliferate to the detriment of native species and ecosystems; however, they represent only a small portion of the larger number of naturalized exotic species (Mack et al. 2000). Although invasions are not a novel phenomenon, they are currently considered to be an important agent of global biodiversity change because of their unprecedentedly large geographical scale and the number and frequency of invasions (Ricciardi 2007). Invasive species have contributed to many animal extinctions in the last few 100 years (Clavero and García-Berthou 2005), with a particularly relevant role in the case of birds on oceanic islands (Blackburn et al. 2004, Sax et al. 2002). Extinctions are the extreme case of biodiversity loss; however, some authors point to the role of species invasions in global biotic homogenization (e.g., Sax and Gaines 2008). Even if exotic species increase local species richness, they are generally cosmopolitan species and do nothing to favor biodiversity at larger scales.

Examples of the negative consequences of species invasions on forest biota are numerous. Dramatic reductions of the population of a species can result from species invasions, as was the case during the destruction of almost all American chestnut (*Castanea dentata*) within their natural range by a fungus transported in imported exotic Asian chestnut (Mack et al. 2000). Another significant example is how the invasion of planted trees (*Pinus* and *Acacia* spp.) into the South African fynbos (native shrubland) transformed this endemic-rich ecosystem due to changes in water availability for native species (Richardson and van Wilgen 2004). Species invasions can also alter the functioning of forest trophic chains, as was observed in New Zealand with the invasion of two wasp species into southern beech (*Fagus* spp.) forests (see Mack et al. 2000 for this and other examples).

Because human-driven disturbances generally increase the invasion by exotic species (Alpert et al. 2000, Lozon and MacIsaac 1997), habitat loss or degradation usually act synergistically with species invasions to create a loss of biodiversity

(Didham et al. 2007). Furthermore, the invasibility of a forest community is influenced by the configuration and the composition of the surrounding landscape (see Vilà and Ibáñez (2011) and the references therein). For example, increased forest edge length in fragmented landscapes, logging roads, or subsidies from a highly disturbed matrix (see complementation and supplementation processes in Sect. 7.2.5.2), can all increase the risk of exotic species invasions in forest interiors (Didham et al. 2007, Kupfer et al. 2006, Vilà and Ibáñez 2011).

7.3.9 Interactions and synergies among different processes

Although we have presented a series of processes as if they act separately, any realistic scenario will be characterized by the interactions among these processes, frequently with synergistic effects (Brook et al. 2008). For example, the impact of fire in tropical forests is influenced by logging and fragmentation, which increase the flammability of tropical forests by drying the understory in canopy gaps and by greatly increasing the amount of dry, fire-prone forest edges and woody debris (Bradshaw et al. 2009b, Lindenmayer 2010). A greater length of forest edges also increases the probability of ignition. This logging–fragmentation–fire interaction is enhanced by severe droughts, whether natural and episodic or induced by global warming (Malhi et al. 2008). In contrast, in Mediterranean forests, increased continuity in the flammable area is being caused by rural abandonment; coupled with climate change, this may exacerbate the size and severity of wildfires. Small to medium wildfires, however, could compensate for the homogenization of the landscape derived from land abandonment by promoting heterogeneity (Loepfe et al. 2010). (See Sect. 7.2.4 for a discussion of the effects of heterogeneity.)

The impact of climate change also results from its interaction with the other agents discussed in this chapter. For example, the capability of a species to adjust its range to keep pace with high rates of climate change depends on its dispersal ability, on the availability of necessary resources in the new habitats, and on landscape permeability. In fact, for a given period, estimated distance shifts in temperature isoclines in some regions may be substantially higher than the estimated maximum dispersal distances of some species (Bacles and Jump 2011). This means that spatial discontinuities (range-shift gaps) between current and projected areas with suitable climatic conditions may preclude some species from shifting their ranges, which is especially likely in the tropics (Colwell et al. 2008). This shift will be curtailed by a reduced functional connectivity as forest landscapes become more degraded, fragmented, and subject to intensified land use (Brook et al. 2008, Colwell et al. 2008). For a species to reach new, climatically suitable habitats may therefore require "assisted migration", a complex and controversial topic that is discussed in Chap. 2 of this book. However, successful shifts in the distribution of a species may also indirectly promote tree mortality by leading to a higher prevalence of diseases due to habitat overlap between species that were formerly separated (Bradshaw et al. 2009a).

Although the complexity of these interactions and of others, some of which are discussed throughout this chapter, hinders our ability to predict the future impacts of landscape changes on the biodiversity of forest ecosystems, conservation policies focusing on single, separated processes will clearly be ineffective and should be avoided.

7.4 Conclusions and general recommendations for mitigating the impacts of landscape change on forest biodiversity

The complexity of landscape change and the uniqueness of every landscape and species make it impossible to offer a comprehensive set of management guidelines, but we can recommend some general principles that will be broadly relevant. We will start with a fundamental idea that may risk stating the obvious: maintaining or restoring forest cover will favor the abundance and richness of forest-dwelling species, and this will foster resilience in the context of landscape change. It is tempting just to say "the more forest, the better", but the real situation is not that simple. The portion of the landscape that needs to be covered by forest to sustain a population of a given species will vary greatly among species depending on their area requirements. (See the concept of "minimum viable population" (or metapopulation) in Sect. 7.2.1.) It is also important to recognize that the habitat needs of species that require other types of vegetation, notably grasslands, also constrain the idea that "more forest is better for biodiversity", especially in the context of afforestation. (See, for example, the responses of generalist species in Sect. 7.2.5.) One could ask, "Is there a minimum threshold for the amount of forest in a landscape?", but again, this is a species-specific question. To take an extreme example, some species might find habitat, or a key habitat element (e.g., a nest site), in a single tree.

Almost as important as the total area of forest is its spatial distribution. A landscape dominated by a large, contiguous tract of forest may provide optimal habitat for species whose survival is limited in small areas, but other species may thrive in a landscape of scattered forest patches, especially if they can move readily among patches and perhaps form metapopulations. (See Sects. 7.2.1 and 7.2.2 for a discussion of metapopulation dynamics and connectivity.) Species that are associated with edge environments—habitats at the interface between forests and other types of ecosystems—may also find superior habitat in a landscape of small, irregularly shaped patches.

Spatial distribution is important because it is a primary determinant of connectivity, which is measured largely by the ability of species to move across a landscape. Mobility is obviously important to species such as a carnivore that must travel widely to find sufficient food, but ultimately it is important to all species because of its effect on processes such as gene flow and the shifting of geographic ranges in response to climate change. Some forest species can move across a fragmented landscape by using forest patches as stepping stones, but open areas are strong filters for other species, and may even function as barriers to their movements.

Such species may be able to move using linear strips of forest such as the riparian forests that line many rivers or along the hedgerows that persist between agricultural fields.

One could argue that a landscape mosaic of numerous differentiated forest patches might be more resilient against change, as exemplified by the folk wisdom, "don't put all your eggs in one basket". For example, an isolated patch might be less likely to burn in a landscape-scale fire. However, as a generalization, the fact that most natural forest landscapes have a high degree of connectivity suggests that connectivity will generally improve resilience against change, by (for example) allowing species to recolonize sites following a local extinction event.

To summarize the previous paragraphs, having extensive forests that are well connected is fundamental to conserving forest biodiversity. In well-forested landscapes, this will require maintaining existing stands and minimizing fragmentation by roads and perforation by the conversion of patches into other land uses. In landscapes that have lost substantial forest cover, this will require forest restoration, undertaken with a particular focus on restoring connectivity by placing new forests in strategic locations such as along riparian zones or as a series of stepping stones that can maintain species fluxes between distant blocks of forest. (See Sect. 7.2.2 for a discussion of connectivity.)

In addition to the quantity and distribution of forests, managers must consider their quality (Sect. 7.2.3). From a biodiversity perspective, the most valuable forests are likely to be pristine, old-growth forests for two primary reasons: First, such forests are rare in most parts of the world and are thus likely to provide habitat for species that are absent or uncommon elsewhere. Second, old forests typically have features such as a sizable accumulation of biomass, high vertical diversity, and canopy gaps that let them support more species than other forest types. Conserving these forests is straightforward, at least conceptually; it requires identifying them and protecting them in a reserve system that is large enough to accommodate the natural dynamics of disturbance and succession.

Most forests are unlikely to be set aside because human demand for timber and other forest products dictate that they will be actively used. Fortunately, it is possible to extract timber in a manner that will sustain biodiversity with relatively modest compromises. There are two key paradigms for maintaining biodiversity in forests that are being managed for timber production (Hunter and Schmiegelow 2011). "Using nature's template" ("emulating natural disturbance") recognizes the coarse similarity between logging and natural forms of disturbance that kill trees and initiate secondary succession, and is based on designing silvicultural systems that will emulate natural disturbances to the extent that is feasible. The key idea is that if species have evolved to survive and even thrive in response to certain natural disturbances, then anthropogenic disturbances will have less impact (but certainly not zero impact) if they closely resemble these natural disturbances. For example, if fires have a return interval of 100 to 200 years in a particular type of forest, then a logging cycle of 100 to 200 years will have less impact than 1 of 50 years. A second paradigm for maintaining biodiversity in managed forests—"diversity begets diversity"—simply recognizes that a diverse forest landscape with stands of many ages, sizes, and tree species compositions will provide habitat for a greater array of species than a highly uniform forest. Such a forest landscape would probably also be more resilient against change agents than a uniform forest.

Plantation forests are widely seen as impoverished from a biodiversity perspective, and there is some truth to that generalization. However, there are some important caveats to note. First, some species, including some uncommon ones that are of concern to conservationists, such as the New Zealand falcon (*Falco novaeseelandiae*), find suitable habitat in plantations. Second, plantations probably constitute suitable temporary habitat for many dispersing organisms. Certainly, they are likely to be preferable habitat compared to wheat fields or parking lots. Thus, establishing plantations provides an important opportunity to restore connectivity by placing them between existing forests. Third, by producing large volumes of timber from a relatively small area, plantations can remove some of the pressure for timber production from natural and seminatural forests, thereby allowing them to be managed for biodiversity. Fourth, some species naturally grow in relatively homogeneous ecosystems, such as the fire-based jack pine (*Pinus banksiana*) ecosystems of boreal Canada. Although such ecosystems are typically more diverse than plantations, the similarity is closer than it is for more diverse ecosystems.

In short, maintaining forest biodiversity in the face of a complex suite of disturbance factors will be most likely to succeed if we can maintain landscapes of well-connected, extensive, and high-quality forests. This is a simple goal to articulate but a challenging one to implement. Ideally, a significant portion of these forests will be set aside in old-growth reserves, and the balance will be managed for both timber production and biodiversity conservation through the careful application of silvicultural techniques that are minimally disruptive and that maintain a diversity of forest conditions. Change is inevitable, but such a landscape will be quite resilient against undesirable change from a biodiversity perspective.

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Chapter 8
Landscape assessment and monitoring

Valentín Gómez-Sanz, Robert G.H. Bunce, and Ramón Elena-Rosselló

Abstract In the present chapter, we provide a basis for discussing some of the main issues concerning the dynamic behavior of landscape systems, and ways to assess their changes over time. We present an illustrative description of a particular Spanish monitoring program in which the authors have been involved. First, we describe landscapes as complex systems with ecosystems that exhibit inherently dynamic behavior. In the following section, we cover the topic of how to study landscape changes, and discuss some of the tools that have been most widely used in recent years. Section 2 discusses the main restrictions and limitations of these approaches, and Sect. 3 discusses the basic procedures used for landscape monitoring and assessment. Finally, Sect. 4 describes one assessment and monitoring program, the Spanish Rural Landscape Monitoring System (SISPARES), identifies bottlenecks, and assesses the system's strengths and weaknesses. The overall purpose of the chapter is to provide readers with methodological tools to identify and evaluate structural and functional changes in landscapes, thereby supporting the development of guidelines for effective and sustainable landscape management.

8.1 The problem of monitoring complex systems

Landscapes show high levels of structural and functional ecological complexity, as well as dynamic behavior. Most current landscapes consist of diverse systems composed of a range of different ecosystem types that have developed in a given area

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under the influence of the interactions between humans or ecosystems and their environment. Human factors, which include cultural factors, socioeconomic policies, and management processes, are particularly decisive in shaping such landscapes. These factors provide the conceptual scenario in which human demands interact with natural resources at both temporal and spatial scales, resulting in specific and complex spatial settings.

In this context, the current patterns and processes that exist in complex and dynamic landscapes, as well as their ecosystem functions, should be interpreted and understood within the framework of an integrated historical and ecological approach, in which time plays an important role (Bolliger et al. 2009, Bürgi et al. 2009, Ortega et al. 2008, Verburg et al. 2004).

From a functional perspective, a given landscape comprises a set of interacting, interdependent components linked by exchanges of energy, matter, and information. These interactions generate a number of associated processes that affect the behavior of a given component of the system, in which the spatial variation in the response generates the landscape pattern (Forman and Godron 1981, 1986; Turner 1989; Urban et al. 1987). These processes include disturbance and can be related to exogenous variables—the so-called driving forces responsible for landscape patterns (Turner et al. 1995). Together, these give rise to landscape patterns through internal interactions between the individual components of the system. Such patterns may involve small variations between time steps and locations that accumulate and lead to variations in structure, even in the absence of environmental heterogeneity (Bolliger et al. 2009, Lischke et al. 2009).

The state of a landscape cannot be explained simply as an equilibrium resulting from the present set of driving forces. Since the same process may produce many different patterns, two landscape patterns will rarely be identical; this makes comparisons difficult between different landscapes or between different time steps, so that methodological problems arise from the combined effects of biotic and abiotic processes (Wagner and Fortin 2005). In this sense, landscapes should be understood as dynamic, time-dependent entities rather than static associations of biotic and abiotic elements.

It is not easy to incorporate the temporal dimension (which is inevitably process-oriented) into landscape studies, as the concept of landscape is often treated as rather static, for example, as a series of images at fixed points in time (Bürgi et al. 2009). Drivers operate at a wide range of spatial and temporal scales, contributing to the emergence of heterogeneous systems (O'Neill et al. 1989, Turner et al. 2001). Complex feedback loops involving significant temporal and spatial lags may lead to discontinuities, thresholds, and losses of landscape components, revealing that local-scale processes and spatial interactions are crucial in assessing landscape dynamics. The existence of different causal processes at contrasting scales means that spatial interactions must be studied at a suitable level for each of the relevant factors.

Interactions among the various drivers of landscape patterns and dynamics can be linear, nonlinear, multidirectional, or unidirectional, and they can form positive or negative feedbacks between the driving factors and their effects on landscape dynamics, on the links between local and regional landscape dynamics, and on the spatial distribution of environmental conditions. Nonlinear and feedback interactions are a primary source of heterogeneity in many systems, and cause strong path dependence, with links to the system's initial condition. The resulting structure or pattern is often difficult to identify in many types of natural systems; sometimes, if one process operates at a slower rate than another process, a nonlinear interaction may appear to be unidirectional despite the true complexity of the interactions between the processes.

When this scenario of heterogeneity is applied, landscape patterns nearly always exhibit scale-dependent spatial autocorrelation in the landscape. This autocorrelation is mainly explained by the action of socioeconomic factors, which are affected by the clustered distributions of landscape features and gradients in environmental conditions that strongly determine the landscape pattern.

The different interactions and feedback mechanisms in different parts of a landscape lead to a dependence on current and historical landscape use, although a landscape's history is often uncertain, potentially leading to varying interpretations of the most important changes that have occurred and hampering the development of an intuitive understanding or direct assessment of likely cause-and-effect relationships. Small individual events may lead to markedly different outcomes, making prediction problematic. Thus, certain trajectories of land-use change may result in systems that exhibit what Verburg et al. (2004) referred to as autocatalytic behavior; that is, the patterns are reproduced in a progressive self-regeneration cycle to maintain the system's function.

The complexity of a landscape reaches its highest level when there are many and diverse driving factors, from abiotic to human, that result from interactions among natural processes, socioeconomic policies, and land-management decisions. Such complexity is both structural and functional, and each factor has a spatial and a temporal scale that dictates its longevity. Depending on the nature of the landscape and the proximity to large human settlements, natural physical and environmental factors may be dominant, or they may be subordinated to human factors to a greater or lesser degree. This has been empirically tested in analyses of the structure and dynamics of the different landscapes that have developed in large, heterogeneous geographical areas, especially when there are significant latitudinal or altitudinal gradients, or both (Ortega et al. 2008).

In this context, the term "landscape monitoring and assessment" represents the regular, long-term surveillance over different time periods of complex systems that show inherent heterogeneity (with spatial autocorrelation as a result of self-organization and emergent properties), and strong path dependence (Bolliger et al. 2009, Lischke et al. 2009, Verburg 2006). Such monitoring and assessment is a primary tool to support analyses of the causes and consequences of landscape changes that will provide a better understanding of the functioning of a land-use system and that will support land-use planning and policy development. As changes are often nonlinear and thresholds are important, landscape-change analysis should take into account the path dependency of a system's evolution, the possibility of multiple stable states on different sides of a threshold, and multiple trajectories between states (Urban 2002, Verburg et al. 2004).

Identifying emerging changes and trends is an important tool for counteracting deleterious future developments, and should allow differentiation between critical and less-important landscape changes. According to Urban (2002), monitoring should be seen as a prerequisite for working towards sustainable development, focusing on the effects of human impacts related to landscape functions and specifically on the capacity of natural processes and components of the system to provide goods and services that directly or indirectly satisfy human needs. However, there is increasing recognition that landscapes also serve important ecological functions that sustain their ability to provide both environmental and human services. Although the fundamental importance of landscape values has been recognized in both scientific and practical contexts, there is still a lack of empirical knowledge about their roles. However, values can be expected to be highly relevant in shaping the perception and management of a landscape, although few empirical studies have focused on their roles (Buchecker et al. 2009).

8.2 Approaches to monitoring landscape patterns

Landscape patterns are shaped by complex dynamic processes that act at various spatial and temporal scales. Currently, many landscapes have only been characterized spatially, although it is widely recognized that many landscape elements are not in equilibrium (Lischke et al. 2009). Thus, long-term landscape monitoring should include a strong focus on the assessment of landscape dynamics and change. Because landscapes are complex systems, systems theory suggests that analysis of their spatio-temporal dynamics and the resulting patterns must rely on knowledge from multiple disciplines, ranging from geography and economics to geology and plant ecology.

The purpose of monitoring and assessing landscape patterns is to identify key features and trends in their values, and this depends on a strict definition of the research objectives and the construction of appropriate databases. In general, three different approaches to quantifying the relationships between landscape patterns and the associated driving forces and processes can be distinguished (Verburg et al. 2004). The first approach analyzes the relationships based on theories and physical laws (Irwin and Geoghegan 2001). For integrated analysis of landscape changes, this approach is often unsuccessful due to the difficulty of including socioeconomic factors. The second approach uses empirical methods to quantify the relationships between landscape patterns and their driving forces. Some statistical techniques are used to quantify the relationships between landscape change and the driving forces, as in the approach used by Spain's "SIstema para el Seguimiento de los PAisajes Rurales ESpañoles" (SISPARES), which describes historical land-use conversions as a function of changes in the driving forces and location characteristics. Finally, the third approach involves the use of expert knowledge. One example would be models that apply expert knowledge to define the behavior of cellular automata that then shape the interactions between the landscape at a given location and adjacent land-use types (Li and Yeh 2002).

In each of these approaches, landscape ecology offers different tools for the characterization of spatiotemporal patterns of landscapes. Indicators and models are two of the most useful tools.

8.2.1 Indicators

Indicators are qualitative descriptors or quantitative measures that report key information that can be used to assess the structure, function, or composition of a system. They identify key attributes of the system based on selected criteria, and their values can be regularly monitored through time or space or both to gain information on the state of the system and on trends that are leading to changes in its state (Dale 2001).

Ecological indicators are usually selected or developed based on expert knowledge, and should be based on field observations to ensure their reliability, even though the long-term effects of environmental change cannot be predicted only from empirical data (Li and Wu 2004). Such indicators can be used to estimate species richness (Cousins and Lindborg 2002), monitor land-use change (Syrbe et al. 2007), or assess the influence of disturbance and management (Bolliger and Mladenoff 2005, Heinz Center 2008). The disadvantage of this approach is that the information derived from ecological indicators does not necessarily allow upscaling or generalization to larger spatial or temporal scales. However, indicators that characterize properties at a landscape scale supplement local-scale ecological indicators by providing information about factors that act at larger scales, such as the amount and spatial arrangement of different land-use and cover types or environmental quality. Thus, the primary goals of using landscape indicators are to quantify the amount and spatial arrangement of various characteristics of the landscape (i.e., patterns that are invisible at smaller scales) and to ensure comparability between different landscapes.

The Statistical Office of the European Community (Eiden et al. 2000) mentions three levels of landscape indicator: (1) statistical data on the area of various land-use and cover types, (2) trends in land-use and cover types that are often related to landscape patterns, and (3) landscape elements that are defined according to the needs of the user. Data are currently available for several countries at level 1, but indicators for levels 2 and 3 have yet to be developed at a national level (Heinz Center 2008, Lausch and Herzog 2002).

Landscape indicators should be based on landscape metrics that statistically represent the characteristics of landscapes or individual landscape elements, and are standard tools to analyze questions about the composition and configuration of landscapes. A wide range of metrics are available, including less familiar approaches from information or fractal theory (Forman and Godron 1986, Turner 1989, Urban et al. 1987), and they are based on the number, size, shape, and arrangement of patches of different types. These indicators are used together with areal statistics such as the distribution of patch areas, edge characteristics, and shapes. Many landscape metrics are now easily calculated (e.g., using FRAGSTATS; McGarigal et al. 2002) and have therefore been widely used in landscape ecology.

Shape characteristics are usually related to the overall heterogeneity of the landscape. Indicators that address landscape patterns and that are based on landscape geometry are often useful in this context. However, the area of an individual patch is ecologically relevant because it determines the space available to support viable populations of key organisms. Such metrics are valuable in studying habitat fragmentation, in which patch isolation may cause the extinction of entire populations because it reduces dispersal or colonization rates below a critical level. In addition, the degree and type of interactions between landscape elements are key factors that shape ecological systems and landscapes (Turner et al. 2001). The connectivity of patches is particularly important, and is often studied using the techniques of ecological network analysis (Bodin and Saura 2010, Fath et al. 2007).

The relationships among driving forces, processes, and landscape characteristics are associated with the scale—which includes spatial, temporal, quantitative, and analytical dimensions—that is used by scientists to measure and study objects and processes (Gibson et al. 2000). Landscapes must inevitably be considered at both temporal and spatial scales that are appropriate for the needs of the research.

For each process that is important to landscape dynamics, a range of scales can be defined to characterize the influence of the process on the landscape pattern. Preliminary studies have provided different conclusions on the magnitude of the effect of scale on the relationships between landscape dynamics and driving forces, and some problems have been identified (King and Perera 2007, Li and Wu 2004). Most land-use models are based on a single scale, and this choice is often based on arbitrary, subjective reasons or on scientific tradition; for example, the resolution of the analysis is often determined by the availability of data, the measurement technique, or the data quality instead of by the processes being studied. For this reason, observation scales often do not correspond to the scale at which the process operates, resulting in an imprecise or inaccurate description of the process, and the aggregation of processes measured at a detailed scale does not always lead to an adequate representation of the net effect of these processes at a landscape scale. Also, little attention has been paid to the interactions between spatial and temporal dimensions, as well as the influence of nonlinear pathways for change, feedbacks, and time lags.

The scale-dependence of these processes reinforces another common problem with using environmental indicators: the spatial units to which they refer. Whereas socioeconomic indicators are usually available for administrative entities or areas, many environmental phenomena do not coincide with the administrative boundaries of these entities or areas.

It is necessary to integrate landscape indicators (which tend to relate to cross-border phenomena) with socioeconomic indicators (which are usually available for administrative entities or areas). Assessment of the latter is difficult because humans act both as individual decisionmakers (which is assumed in most econometric models) and as members of complex social systems. These problems are especially difficult where the study area is large and complex. Inevitably, some of the objectives of these actors conflict with each other. Similar scale dependencies are found in biophysical processes, and the aggregated result of individual processes cannot always be directly determined by integrating their individual effects, particularly when those effects are measured at local scales.

Consequently, the ecological relevance of the available metrics must be determined in terms of their ability to meet particular objectives, thereby avoiding misleading conclusions. This requires the selection of a manageable set of appropriate indicators that embrace the structural and functional properties of a landscape. The signal to be detected is often uncertain, and the objects of concern may be difficult to locate, but the main criterion for success is the development of a sound scientific framework. In work involving landscape metrics, it is necessary to select indicators that are relevant both for the study area and for the problem under investigation.

In the field of landscape monitoring, the application of landscape metrics has been tested in a number of studies that represented a wide range of test areas and methods of data acquisition and analysis (Lausch and Herzog 2002). The most frequently applied landscape indices belong to the broad category of edge and shape metrics, and these are often related to the patch area and fractal dimension. Diversity measures are usually derived from information theory and often involve the use of Shannon's diversity index. The number and size of patches are also often measured, whereas metrics for landscape configuration (e.g., the contagion index, which represents the degree of aggregation of patches of certain types) have seldom been applied. Other methods are based on single indicators, such as land-use and cover-type change or the loss of a landscape element. Land use is one of the main factors through which humans influence the environment, but is often confused with land cover; the former is the way land is used (e.g., forestry, agriculture) whereas the latter concerns the cover of the land surface independent of its use (e.g., deciduous forest, grassland).

In conclusion, when a landscape must be monitored over time, researchers and managers must choose indicators that provide a suitable tool for understanding the functions of the observed system, as well as its patterns or diversity, and that can be used to assess the consequences of changes in individual system components or of the environment. They must also allow a consideration of alternative scenarios to facilitate decisions about land-use options and conservation and about whether intervention would yield benefits. To be useful in prediction of future states, they should also support the development and testing of hypotheses about the state of a system under past, current, or future conditions, such as in studies of the changes in forest cover or agricultural use over time (Lischke et al. 2009).

8.2.2 Models

In landscape ecology, models are tools derived using various methodologies in order to quantify, assess, and predict spatially dynamic patterns based on their underlying processes at a landscape scale. Today, these methodologies include an approach to characterize particular periods (static) or time series (dynamic) based on discrete or continuous representations of landscape heterogeneity, or as implementations of conceptual models based on empirical observations and experiments (Lischke et al. 2009).

A number of model types can be distinguished, although there is no generally applicable classification scheme for models. They range from purely conceptual, descriptive word or graphic models to semiquantitative graphical schemes and mathematically formalized models in the form of computer programs that yield quantitative descriptions. Model types also differ in how they define and account for landscape heterogeneity. In practice, many models not only ignore heterogeneity but do not validate their results (Li and Wu 2004). The quality of the input data is also rarely taken into account.

Landscape structure and dynamics can be studied using discrete or continuous models of spatial and temporal heterogeneity. Approaches to identify and quantify spatial landscape patterns are widely used for discrete landscape representations, whereas continuous views are less well developed.

Discrete categories are often seen as a form of generalization and assume that landscapes consist of discrete, nonoverlapping objects or patches that belong to mutually exclusive classes; the patches are considered either to be embedded in matrices that are assumed to be homogeneous or to form mosaics. Discrete landscape representations are widespread and are helpful to simplify and quantify complex landscapes. As a result, they have been successful in a wide range of ecological contexts, such as studies of habitat fragmentation and the monitoring of landscape change. Many of these studies assumed that a given habitat mosaic was constant in order to study its effects on shifts in the distribution of organisms, populations, or species.

However, whether a phenomenon appears as discrete or continuous often depends on the scale of the study, and especially on the spatial resolution (granularity), the measurement resolution, and the hierarchical scale. Many natural phenomena are continuous in character and exist as gradients rather than as features with discrete boundaries. Classifications of spatially continuous features into discrete units may result in information loss.

From a temporal standpoint, landscape modeling can use static or dynamic approaches. Static approaches assume that the landscape is in equilibrium with its environment and do not account for transient phases. Their general expression is of the form $Y(s_j) = f(X[s_j])$, where Y represents state variables such as landscape units or land cover types, X represents exogenous factors such as soil productivity, and s_j represents the positions within the landscape. Applied with a discrete time step t_i, the model yields $Y(t_i, s_j) = f(X[t_i, s_j])$ and represents the spatial distribution of the landscape unit at an individual time or over a series of times.

State variables are often linked with exogenous factors using various regression approaches, including logistic regression (Bolliger et al. 2000), classification and regression-tree models (De'Ath and Fabricius 2000), and general additive models (Brown et al. 2006). The simulated landscape heterogeneity is thus a simple mapping based on the heterogeneity of the exogenous factors that are used as inputs for the model.

The main advantage of static models is that, in many situations, they can be a good starting point for further modeling approaches. Modern algorithms can enable rapid statistical calculations and can account for spatial interactions by applying methods that consider the effect of spatial autocorrelation. In this way, the assessment of factors and factor combinations that are relevant for a given landscape pattern and predictions of this pattern are given in a geographically explicit form and are interpretable as maps using tools such as geographical information systems (GIS).

The main disadvantage is that the mechanisms that determine spatiotemporal patterns are not explicitly included in such statistical models, which suggests that the ecosystems are considered to be in equilibrium and that the models therefore ignore transient behavior and temporal factors. This limits cause-and-effect analyses and restricts extrapolations to the range of factors used to calibrate the model.

In contrast, dynamic models account for the transient nature of the states of a system. In these models, the landscape's state at a given time or location is then driven by changes in both exogenous factors and endogenous processes and their interactions. For each location within the landscape, these are based on assumptions about the underlying processes and describe the temporal changes in the state variable, Y, using models with a different form: $\mathrm{d}Y(t)/\mathrm{d}t = f(Y(t), X(t), \varepsilon)$. These models can be deterministic ($\varepsilon = 0$) or they can be stochastic ($\varepsilon = 0$), allowing them to account for random or probabilistic influences.

The disadvantage of dynamic models is that they may not be considered as spatially explicit because this would lead to prohibitively long computation times. This, however, has the advantage that simulations based on such models are usually fast, require only small computer storage capacity, and can simulate state variables in a highly detailed way. Another advantage is that they allow researchers and managers to visualize and interpret the temporal course of the state variables.

Dynamic regionalized models incorporate landscape heterogeneity by applying dynamic point–area or area models in parallel at many locations s_j within a grid. They have the following general form: $\mathrm{d}Y(t, s)/\mathrm{d}t = \mathrm{f}(Y(t, s_j), X(t, s_j), \varepsilon)$. Thus, this type of model combines the spatial and temporal aspects of the landscape. Applications of dynamic distributed models include the evaluation of global-change phenomena such as possible future land use and climate scenarios. Nevertheless, complexity and reliability are evident disadvantages of this type of model; in particular, many of the underlying processes embodied in the model may not yet be adequately parameterized for all parts of the region being studied.

8.3 The monitoring and assessment procedure

According to Brunt (2000), managing ecological data is a process that should start with the conception and design of the research project; continue with data acquisition, quality control, manipulation, and quality assurance; and conclude with analysis and interpretation. Subsequently, the gathered, processed, and derived data should be archived, published, and made directly accessible to those who can benefit from the information.

Consequently, a basic monitoring and assessment procedure should begin with the formulation of hypotheses and assumptions that are designed to provide approximate answers to the widest range of questions that are specifically related to the researcher's or manager's knowledge needs and the problems and environmental risks that must be addressed. Hypotheses must be formulated with respect to the appropriate spatial scale or scales and content of interest (i.e., land-use and cover types), and must be multifunctional, according to the ecological concept of landscape; if a monofunctional approach is chosen, this choice should be duly justified. The British Countryside Survey (Howard et al. 2000) and the American State of the Nation's Ecosystems (Heinz Center 2008) follow this approach.

The next step is to select a set of indicators that can be used to test the hypothesis and assumptions. This task is a key to successful monitoring and assessment, thus it is necessary to choose the most appropriate indicators based on the research objective and scale, and based on budgetary and time constraints, so that the indicators adequately mirror the system's structure, state, functions, and composition. Indicator selection should also include an assessment of the limitations of each indicator so that these limitations can be accounted for.

One of the main challenges facing monitoring projects is the development of an integrated method for data sampling and analysis at appropriate spatial and temporal intervals. The ultimate goal should be to establish sound data structures. These form the indispensable foundation for a statistical analysis of landscape patterns and will enable comprehensive exploration and modeling of temporal and spatial processes in a landscape (Lanz et al. 2009). Data is important, but it must also be transformed into information; here, we define information as raw data combined with knowledge about the context of the data, including interrelationships between existing data; about how the data was collected, processed, and used; and about how the data should be understood within a given application.

At a landscape scale, the sampling problem is to simultaneously capture both the fine- and coarse-grained patterns. This challenge is not met by conventional systems, in which samples are not based on knowledge of the variability within the system, so that details of the variation in fine-grained local patterns will be missed. The classical solution has been to obtain stratified samples that achieve balanced coverage of each domain being studied within the landscape. In landscape ecology, this stratification is primarily spatial, and the strata are usually derived based on biological or environmental factors such as vegetation types, topographic positions, and soil types. However, temporal stratification is also important to account for seasonal environmental variations (e.g., temperatures, rainfall patterns) and socio-economic variations (e.g., tourist seasons, harvesting seasons).

Nevertheless, these types of designs may be limited by their dependence on the sampling intensity, since many factors can constrain the sampling performance. Despite these methodological limitations, stratified sampling designs, based on preliminary multifactorial environmental classifications, have been widely applied in European national surveys (Bunce et al. 1996b, Elena-Rosselló et al. 1996, Ihse 1995). This model, as well as decisions made to control sample sizes at appropriate levels for a given landscape type, is a distinctive methodological feature of the European national systems (Ortega et al. 2012a). To sample variables with an unknown granularity or pattern, a multi-scale pilot study would be necessary to develop the most efficient possible design for subsequent sampling (Urban 2002).

The need for a statistical sampling strategy increases when monitoring systems are required for a large region that has unknown variability. In large-scale systems such as landscapes, indicators of the system's state can be monitored only through a time series of empirical observations made at particular time steps ("snapshots"); these may originate from successive field measurements or other snapshot data such as historical aerial photographs. For example, Spain's SISPARES (which we will discuss in more detail in Sect. 4) effectively incorporates these monitoring tools. In the future, remotely sensed images will become increasingly inexpensive and easier to obtain for large areas, but there are currently problems with the spatial resolution of these images and the accuracy of the image interpretation.

The need for strategic sampling was particularly important three decades ago, when computers were not powerful enough to process comprehensive sets of environmental data at a national scale (Bunce and Smith 1978). Since then, there has been considerable research in this field, assisted by the increasing availability of satellite imagery and powerful computers capable of running sophisticated GIS software. Despite these advances, it remains important to ensure that the selected data structures are appropriate for the project's objectives, since this approach is more powerful than unstructured analysis of large datasets only because they are available.

Monitoring programs often rely on complex hybrid designs to meet multiple objectives. For example, the SISPARES approach is designed to include sampling at multiple scales and in multiple phases. The initial sample is surveyed for readily measured, coarse-resolution variables. In a subsequent phase, a subset of these samples is revisited and a different set of more logistically demanding variables are measured. This second set is then related to the initial set and is later used to add detail to the initial, coarse-resolution dataset.

The sampling design of a monitoring program is strongly conditioned by budgetary limitations, which depend mostly on the spatial scale of the program. In the case of large study areas such as continents or large countries (e.g., Brazil, China, Australia), stratified random sampling is the most cost-effective design if the program involves field verification of the data (i.e., ground-truthing). This is true of the European Biodiversity Observation Network and the European contribution to the Group on Earth Observations Biodiversity Observation Network (Bunce et al. 2008).

The development of any environmental land classification in a stratified random-sampling design must be based on objective information about biophysical factors. Land classification is the backbone of the monitoring program because it is needed not only for the sampling but also for the data integration and processing, and for the analysis, interpretation, and display of the results. Examples include the GB Land Classification by the Institute of Terrestrial Ecology (Bunce et al. 1996b), the Clasificación biogeoclimática territorial de España classification applied in SISPARES (Elena-Rosselló et al. 1996), and the Environmental Classification of Europe used for European Biodiversity Observation Network (Metzger et al. 2005).

After the retrieval of an adequate number of observations, the next step should be data screening to ensure that the data quality is sufficient to allow calculation of the indicator values. The goal is to improve the comparability, reliability, and accuracy of estimates of landscape values over a long period. Some observations are more informative about a specific hypothesis whereas other data might not provide adequate detail. In local field studies, ecologists sometimes over-sample by using a “shotgun” approach that collects the appropriate data together with extraneous data that may not be necessary to support the specific objectives of the study. This should be avoided because it may introduce excessive “noise” in the data or may use resources inefficiently. An alternative approach is to use a model to help discover which observations will be most useful for a specific application or research task.

All of these tasks must allow the implementation of suitable techniques to assess and quantify heterogeneity, as well as to assess landscape functions and components and to detect driving factors. In the portfolio of tools and techniques that are available, models are the main method that is used to unravel the dynamics of landscape systems and improve understanding of the structure and dynamics of a landscape (Verburg 2006). All models implemented for this purpose should include a temporal component for changes in the state variables, either in discrete time steps (e.g., years, generations) or continuously.

Unfortunately, many modeling studies have limitations on the interpretation of their results. It is necessary to establish modeling approaches whose complexity (in terms of the number of state variables, parameters, and processes) best suits the research question and the available data (Bolliger et al. 2005). The search for relationships between patterns and processes requires careful evaluation, especially since we currently lack a thorough understanding of the degree of landscape changes required to provoke ecologically relevant consequences (Turner et al. 2001, Wu and Hobbes 2002). Effective landscape models should be able to relate a spatial and temporal pattern to both exogenous and endogenous drivers, integrate current knowledge about the influences of drivers and the interactions among them, rank them, reveal inconsistencies and uncertainties, and make simplifications explicit. In addition to supporting the interpretation of data, landscape models should allow scenario testing by assessing the effects of different degrees of change in a particular study area. This may lead to confirmation, rejection, or generation of hypotheses and can support environmental decisions and policymaking.

A major challenge in landscape modeling relates to data availability. There is, on the one hand, rapid development of Earth observation and monitoring techniques to support large-scale spatiotemporal modeling. On the other hand, there is often a data scarcity because modeling was not planned or integrated with a sampling strategy.

Standardized data-processing techniques are vital to ensure the spatial and temporal comparability of results from different studies. Data for similar locations may be available from different data sources, although methodological differences may complicate the use of this data. This problem forces the use of data-integration techniques that extend or harmonize time-series data. Initiatives at the European level aim for standardization and consistency in data collection and data sharing. The INSPIRE initiative (Infrastructure for Spatial Information in Europe; http://inspire.jrc.ec.europa.eu/) promotes the availability of relevant, consistent, and high-quality geographic information. In addition, the European National Forest Inventory

Network initiative (http://193.170.148.89/enfin/) aims to promote new research on making definitions consistent, on data collection, and on estimation techniques applied in national forest inventories.

It is also necessary to improve data management to ensure comparability within and between different landscapes and scales, and data-integration techniques are needed in order to extend or ensure the consistency of time series. Furthermore, methods for dealing with heterogeneous data based on different semantics and with different degrees of uncertainty must be implemented along with methods for identifying sources of error.

Standards for quality control and quality assessment are needed to guarantee methodologically consistent data collection and data processing over long periods as well as to ensure adequate data quality for efficient integration and sharing. Quality assurance and control procedures should be implemented at all stages of a research project so that stakeholders will have confidence in the results. Quality assurance includes planning, documentation of methods, training, and analysis of information-management procedures. Quality control involves checking data quality, maintaining recording standards, and auditing analytical procedures. Examples of criteria that can be used to judge data quality are precision, avoidance of bias, completeness, comparability, homogeneity, representativeness, and reproducibility (Stierlin 2001). Most of these criteria can be assessed by using statistical tests and similar techniques developed in the fields of industrial production, training, and data-survey control (Kaufmann and Schwyzer 2001).

8.4 The SISPARES approach

Since the second half of the twentieth century, many studies have been published on landscape changes throughout the world. The availability of aerial photography with images captured over a series of dates made these analyses possible. However, most of these projects were performed at local scales and could not be extrapolated to larger scales. The need for national monitoring systems was officially agreed upon at the Florence Convention in 2000, where members agreed that every member state was responsible for monitoring its own landscapes.

However, some pioneering monitoring systems were developed before this convention. The GB Countryside Survey has been carried out four times since 1978 by the Centre for Ecology and Hydrology, and has monitored a range of ecological indicators (Howard and Bunce 1996). Three other European countries have also initiated nationwide landscape-monitoring schemes in recent years: Sweden, Austria, and Spain.

Spain's SISPARES project (which began in 1993) provides a good example of an integrated approach that illustrates methodological and conceptual advances in landscape monitoring. In the rest of this chapter, we will describe the system to provide readers with information on its key concepts and methods. We hope that this description will stimulate a new generation of large-scale landscape monitoring.

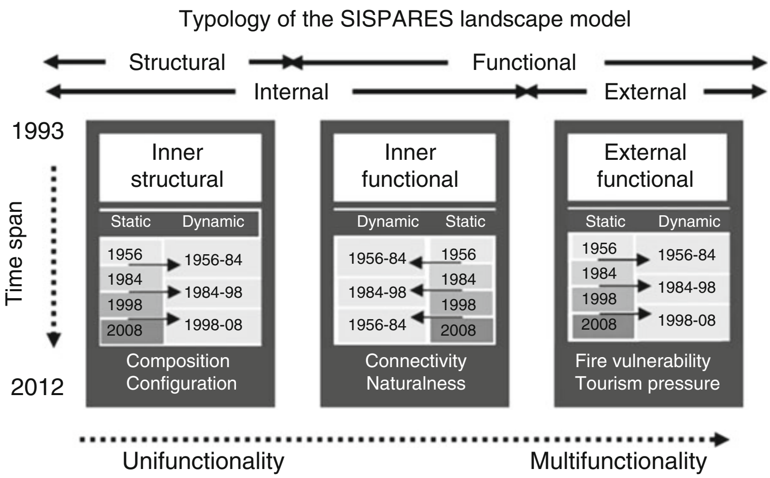

Figure 8.1 The evolution of SISPARES. The development process had three successive stages: (*Left*: the first model) In the first version of the model, SISPARES was based on building the simplest possible core model, compared only two dates (1956 and 1984), and had a primarily static structure. SISPARES subsequently included two later survey dates (1998 and 2008), thereby enlarging the time span for the static structural model to 52 years (1956 to 2008). Subsequently, dynamic models were built by comparing the static structural data in consecutive survey periods. (*Center*: the second model) In parallel with this first model, functional indicators were computed from recorded land-use and cover-type composition and configuration data. Models of the naturalness of a landscape and its connectivity for fauna were developed from both static and dynamic models during this period. (*Right*: the third model) Once the SISPARES database had been developed for three periods, it was possible to link its structural and internal functional values with available external databases of landscape functions such as nature conservation, ecosystem biodiversity, vulnerability to wildfire, tourism activity, and other factors

SISPARES is the acronym for the Spanish Rural Landscape Monitoring System (a translation of *Sistema para el Seguimiento de los Paisajes Rurales Españoles*). After a 19-year implementation period, SISPARES has evolved from simplicity towards the complexity of a mature system, from an emphasis on structure towards an emphasis on function, and from static to dynamic analysis. Early versions of SISPARES produced simple static structural models. The current version generates dynamic multifunctional models (Figure. 8.1). The time span required for this evolution has paralleled the changes in Spain during the twentieth century.

8.4.1 Land classification: the backbone of landscape-monitoring frameworks

SISPARES and the GB Countryside Survey share a common methodological base in that both models are built on the data recorded from environmentally stratified samples. The required environmental stratification is provided by a land classification system, which is an integral part of both monitoring systems. This approach has also been implemented at a continental scale as part of the design of a standardized procedure for surveillance and monitoring of Europe (Bunce et al. 2008).

The land classification system was an essential component of the methodological approach developed at the British Institute of Terrestrial Ecology: first, it was developed at a regional level in Cumbria (Bunce and Smith 1978) and then it was expanded for all of Great Britain (Bunce et al. 1996b). Both classifications were developed by applying multifactor automatic classification methods to existing biogeoclimatic data information (Bunce et al. 1996a). The spatial environmental database is provided by a climatic stratification for Europe (Metzger et al. 2005).

The land classification system in any monitoring system designed to operate at a national scale must fulfill two important conditions:

1. Environmental significance: the classes must match the environmental patterns that are present in the landscape and that have been proven to be key landscape factors.
2. Data-management coordination: the resulting databases must reveal spatial patterns consistent with other environmental or socioeconomic databases at the national scale.

If the classification has environmental significance, it becomes a useful tool for providing spatial integration with the databases generated by successive surveys. In the 1980s, land classification was also an indispensable requirement for spatial data management. Computers at that time were unable to process the huge landscape database for a medium-sized European country (ranging from 30 000 to 550 000 km^2). In the current century, this is no longer such an important limitation, and this has allowed models to become more sophisticated.

8.4.2 REDPARES: a permanent rural landscape network

Land classification models provided the basis for stratified sampling, first to assess landscapes and later to monitor them. Long-term monitoring implies the availability of a network of permanent landscape sample plots that have been surveyed in the past and that will be available for surveys in the future. This permits the development of a statistically reliable and cost-effective landscape-monitoring system (Bunce et al. 2008).

In SISPARES, a land classification known by the acronym CLATERES ("Clasificación biogeoclimática territorial de España") was used (Elena-Rosselló et al. 1993). This system was developed for peninsular Spain and the Balearic Islands. A similar but simpler model was applied to select the landscape samples in the Canary Islands. Afterwards, a network of 215 permanent landscape sample plots, referred to by the acronym REDPARES ("Red de Paisajes Rurales Españoles"), was outlined.

The design of the landscape sample units (size, shape, area, and data to be recorded) was initially based on the study of landscape changes in the American state of Georgia (Turner and Ruscher 1988). Accordingly, 4 × 4 km Red de Paisajes

Rurales Españoles landscape units were selected using the Clasificación biogeoclimática territorial de España land classes as sampling strata. The locations of the sample units were randomly selected within each stratum. The spatial parameters were determined by interpretation of aerial photographs taken simultaneously throughout Spain.

The availability of photographs was a key factor for deciding the survey dates. To ensure temporal consistency of the data, it was judged necessary for the data collection during a given survey to be completed in less than 3 years, with photos taken at a similar scale (around 1:30 000). Such conditions were met in 1956 and 1984 during the initial SISPARES period that began in 1993. Later, 1998 and 2008 were selected for surveys. SISPARES therefore has a time span of 52 years, with data from four survey dates (1956, 1984, 1998, and 2008).

The landscape information was recorded following the procedure designed by Turner and Ruscher (1988). Once the different land-use and cover-type patches were detected, a cover map was drawn. In parallel, linear and point elements were detected, identified, and mapped. The maps were digitized and analyzed in order to calculate composition and configuration indices. Following the fast evolution of GIS technology, the SPAN software was first used (Turner and Ruscher 1988), followed by the FRAGSTATS software (McGarigal et al. 2002), and analysis is currently performed using the patch-analyst extension (http://www.cnfer.on.ca/SEP/patchanalyst/) of ArcGIS (http://www.esri.com/software/arcgis). From the raw area, linear, and point data, SISPARES computes internal structural indicators (Table 8.1). After processing these internal structural indices and metrics, SISPARES calculates functional indicators such as accessibility, fragility, vulnerability, and connectivity. Recently, new indicators have been calculated using the CONEFOR-SENSINODE software (Saura and Torné 2009) and have been included in the updated version of SISPARES, providing a new generation of ecological connectivity indicators.

From the static information on spatial composition and configuration recorded at each survey date, SISPARES can produce a map of changes during each period between dates by overlaying the initial and final maps. Change maps can then be generated that show patches with different distinctive changes in land use, cover type, or vegetation density. Each change is a specific consequence of a given anthropogenic or ecological process taking place during the period between surveys. SISPARES has detected a catalog of 15 main landscape ecological processes, and defines and assesses each process in ecological terms (Table 8.2).

Finally, functional information available from external sources has been recently included in the landscape database, including data on wildfire occurrence and propagation recorded by the Spanish Ministry of the Environment from 1974 to 2012 (EGIF 2009, Ortega et al. 2012b).

Another critical requirement to be considered when selecting a survey date was the existence of changes in the sociopolitical drivers that could determine future landscape structure and functions. The probabilities of natural changes in the landscape's abiotic factors were relatively low during the study period because most of the landscape changes have been caused by human decisions. Such human impacts have generally been driven at a national scale by sociopolitical and economic

Table 8.1. Description of the indicators used in SISPARES.

Index type	Characteristic assessed	Calculated index
Composition	Landscape richness	PR—Patch Richness (McGarigal et al. 2002)
	Land-use and cover-type (LUCT) diversity	SHDI (LUCT)—Shannon Diversity Index for LUCT (McGarigal et al. 2002)
	Woodland	% W
	Cropland	% C
	Agroforestry	% A
	Forest recovery	% F
	Pasture	% P
	Shrubland	% S
	Urban	% U
Configuration	Fragmentation	PD—Patch Density (McGarigal et al. 2002)
	Mosaicity	% X (mosaic patches)
	Patch diversity	SHDI (PS)—Shannon Diversity Index for Patch Size
	Patch complexity	MSI—Mean Shape Index (McGarigal et al. 2002)
	Interspersion	IJI—Interspersion and Juxtaposition Index (McGarigal et al. 2002)
Combined	Accessibility	RTD—Roads and Tracks Density (García-Feced et al. 2008)
	Fragility	LFI—Landscape Fragility Index (García-Feced et al. 2008)
	Vulnerability	LVI—Landscape Vulnerability Index (Ortega et al. 2012b)
	Connectivity (functional)	PC—Probability of Connectivity (Bodin and Saura 2010)

Table 8.2. Most frequent and extensive change processes during the three time periods covered by SISPARES.

	Mean sample percentage		
Processes	1956 to 1984	1984 to 1998	1998 to 2008
Maintenance	31.3	39.1	44.7
Forest maintenance	25.9	36.6	53.5
Shrubland establishment	7.9	4.8	1.2
Afforestation	8.0	4.5	1.4
Crop establishment	5.3	2.7	0.8
Pasture establishment	3.1	2.1	0.9
Desertification	1.2	0.6	0.8
Forest densification	11.3	10.5	2.0
Forest clearing	6.9	4.3	2.8
Reforestation	12.4	3.6	2.7
Fragmentation	4.8	2.7	1.2
Flooding	1.1	1.0	0.3
Riparian vegetation establishment	0.7	0.5	0.2
Agroforestry establishment (*dehesa*)	4.1	2.6	3.2
Urbanization	2.1	0.6	1.3

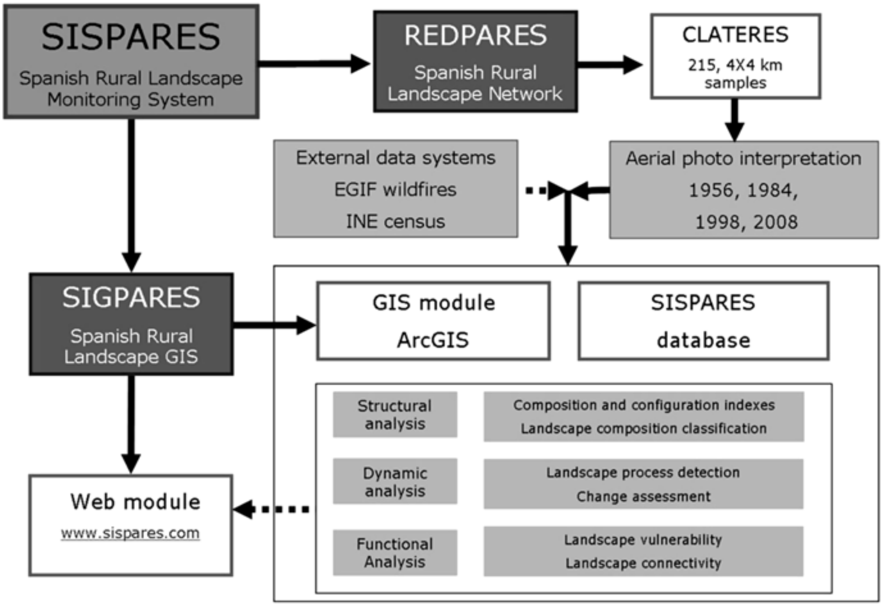

Figure 8.2 The structural components and functional processes incorporated in SISPARES, and a flow chart for the processing of information from inputs to outputs. REDPARES (the rural landscape samples network) and SIGPARES (the landscape GIS module) are the two operational components. REDPARES includes a network of 215 permanent landscape sample plots that were selected based on CLATERES, a biogeoclimatic land classification system that provides spatial integration at the resulting SISPARES model. SIGPARES is the data-processing component, which comprises a GIS module in which all the spatial information is stored and processed, and a Web-based module for disseminating the results (*EGIF* Spanish Ministry of the Environment, *INE* Spanish National Institute of Statistics)

policies developed by national and international organizations. Therefore, the survey dates and monitoring periods were selected according to key moments in Spain's socioeconomic evolution (Bolliger et al. 2009, Bürgi et al. 2009).

From this perspective, the SISPARES survey dates (1956, 1984, 1998, and 2008) were good choices because they coincided with important milestones in Spain's socioeconomic evolution during the last six decades. Between 1956 and 1984, the major change was that Spain joined the European Union. In 1998, Spain adopted the Euro as its official currency, and in 2008, a deep economic crisis started in Spain.

This preliminary selection was fully justified by subsequent analysis of the landscape's evolution during the three periods. All three periods showed clear changes in land uses that reflect visible imprints of human activities on Spain's ecosystems. In this context, SISPARES has helped researchers and managers to interpret the current patterns and processes in Spain's landscapes, as well as changes in ecosystem functions.

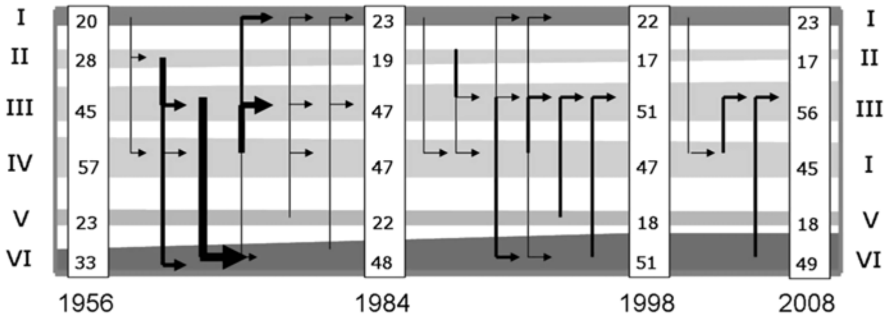

Figure 8.3 Sample outputs of the SISPARES model, illustrating the main structural changes that occurred in Spain's landscapes since 1956. The diagram shows the general evolution of trends in the agroforestry landscape composition assessed in the REDPARES rural landscape module. Two hundred and six samples located in the Iberian Peninsula and Balearic Islands were successively classified in 1956, 1984, 1998, and 2008. Landscape types are defined by Ortega et al. (2012b): type I, primarily agricultural; II, mixed forest with small agricultural fields; III, mixed forest with small and medium agricultural fields; IV, mixed forest with large agricultural fields; V, *dehesa-dominated* landscapes (i.e., a traditional form of Spanish agroforestry in which a few scattered trees are a dominant feature of the landscape); VI, primarily forested landscapes. *Arrows* represent a change from one of the six landscape types to another type; the thickness of the arrows represents the approximate change in the number of REDPARES samples. The evolution of each landscape type is represented by changes in the thickness of their horizontal strip. Numbers are the amount of samples of each type at each surveyed date

Figure 8.2 summarizes the current components of SISPARES. During its development, SISPARES has progressively evolved from a simple, static, structural core model into a complex, dynamic, structural, and multifunctional model.

8.4.3 What insights has SISPARES produced?

Since its launch, SISPARES has been used to produce many results and products. Figure 8.3 shows an example of the efficiency of SISPARES in detecting and interpreting the main landscape trends that have occurred since 1956. For example, the landscape changed more dramatically from 1956 to 1984 than during the next two periods. SISPARES has recorded major differences in the scale of the changes between the first, second, and third periods. Between 1956 and 1984, 26 samples completely changed their agroforestry landscape composition, whereas between 1984 and 2008, only 11 samples had changed (Ortega et al. 2012b).

This evolution was linked to the changes in Spain's sociopolitical and economic conditions. Until 1976, Spain was ruled by an authoritarian political regime with a highly centralized public administration. The regime developed land-management programs characterized by national decisions made regardless of the interests of local populations. Two national programs during that period were plans for irrigation

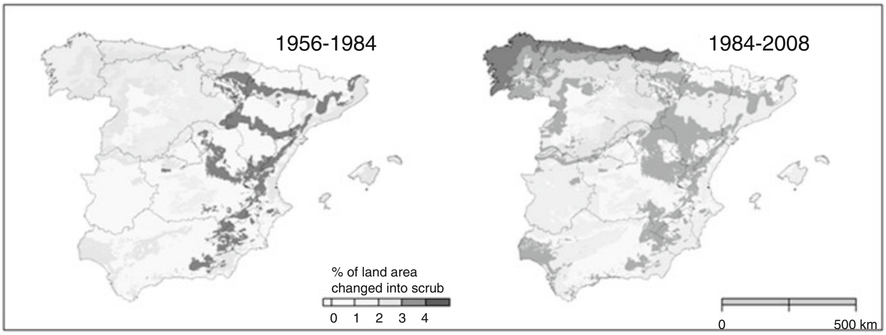

Figure 8.4 Sample output of the SISPARES model: The maps of Spain show the geographical distribution of a landscape process called "matorralization" (in which ecosystems evolve into scrublands as a result of repeated fires, overgrazing, or abandonment of marginal crops) for two of the surveyed periods. Extensive abandonment of marginal crops in parts of eastern Spain at moderate elevation was followed by secondary successional processes, resulting in matorralization of those regions between 1956 and 1984. From 1984 to 2008, this geographical pattern shifted from eastern regions to northern regions, due mainly to processes resulting from the high frequency of wildfires in those regions

of arable land and for reforestation. Both of these plans were directly responsible for many of the recorded changes.

In 1976, the political regime changed to a democratic and decentralized system, in which Spain's 17 regions gained a high level of political autonomy. As a consequence, the annual reforestation rates remained similar to those in previous years, but the geographical pattern was different. In addition, the average size of the reforestation projects was smaller, and the geographical dispersion was higher.

As an example of the potential for designing models of changes that result from combinations of human actions and autonomous landscape functional processes, Figure 8.4 shows the distribution maps for one of the detected processes that has taken place in the last five decades: the so-called "matorralization", which describes a common Mediterranean degradation process in which climax forest degrades into shrubland, due mainly to recurrent wildfires and overgrazing.

Because of its progressive development, SISPARES was able to analyze the stored data using increasingly broad time spans and a range of spatial scales. Two of the first results were permitted by the development of tools for better organization of the raw and combined information that SISPARES has stored and produced:

1. The first was the development of a landscape-pattern taxonomy based on the composition of land-use and cover-type classes (García del Barrio et al. 2003) and a classification of the agroforestry landscapes of Spain (Ortega et al. 2012b).
2. The second was the development of a methodology to divide forest districts into distinct landscapes (García-Feced et al. 2008, 2011).

SISPARES was designed specifically to produce geographical models of the Spanish landscape structure and of its functioning and dynamics. The functionality of the landscapes can be assessed directly from the structural and dynamics data, or

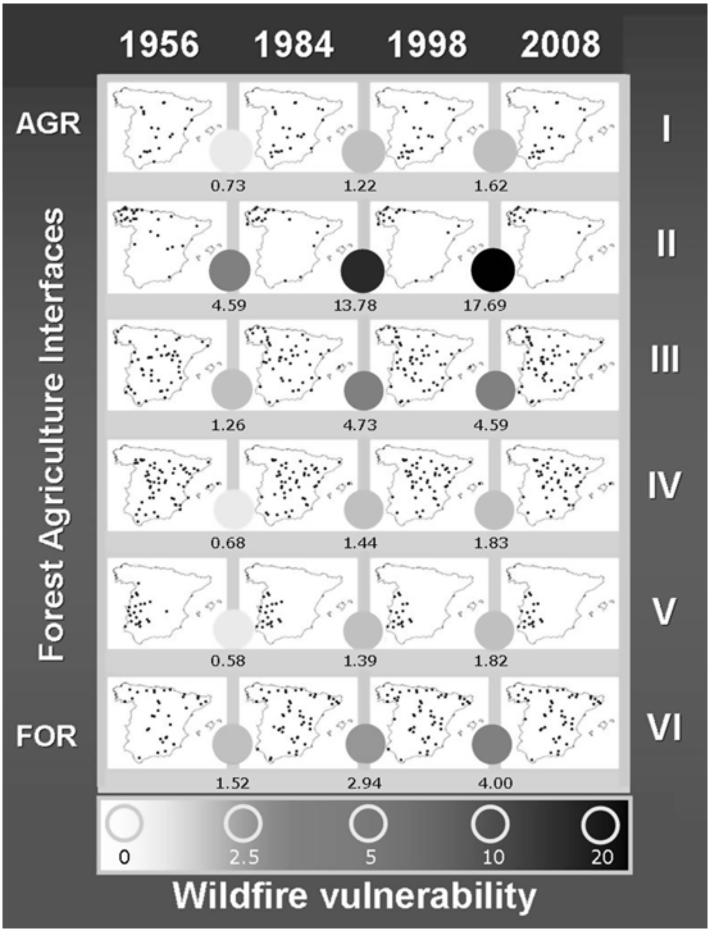

Figure 8.5 An example of the structural and functional model output from SISPARES. Maps show the distribution of the six structural landscape types defined by Ortega et al. (2012b): type I, primarily agricultural; II, mixed forest with small agricultural fields; III, mixed forest with small and medium agricultural fields; IV, mixed forest with large agricultural fields; V, *dehesa-dominated* landscapes (i.e., a traditional form of Spanish agroforestry in which scattered trees are a dominant part of the landscape); VI, primarily forested landscapes. This classification was developed to achieve the maximum discrimination in wildfire vulnerability, assessed as the number of ignitions per 4×4 km landscape square. Changes during each study period were assessed and geographically located. *AGR* agricultural, *FOR* forest

can be input into models. For example, Figure 8.5 shows the SISPARES model of wildfire vulnerability (Ortega et al. 2012b). As we mentioned earlier, the land classification plays an important role in spatially linking the SISPARES database with other available databases that supply additional functional information.

An important output from the model of vulnerability to wildfire has been the determination of a threshold for values of landscape structure (composition and configuration). The highest risk of wildfire is for the land-use and cover type with

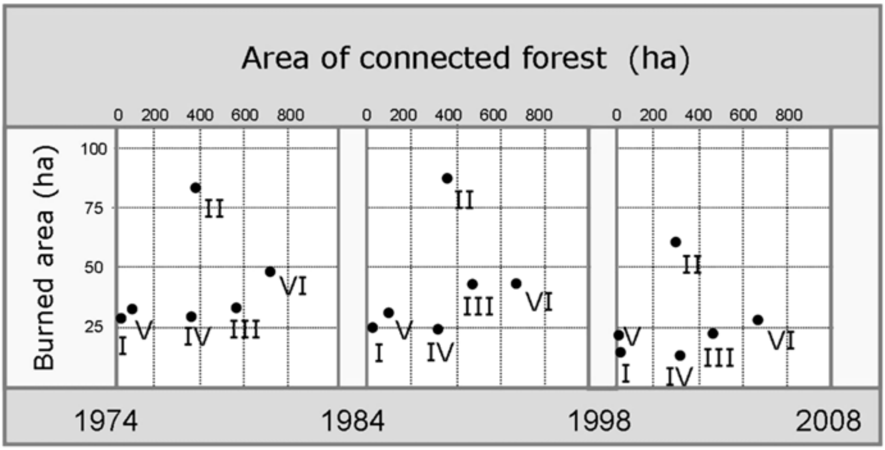

Figure 8.6 An example of the multifunctional model output from SISPARES. Diagrams show the changes in the burned area as a function of the area of connected forest in each land-use and cover type during the three study periods. The evolution of both functional features has been similar for the six structural landscape types. Landscape type II, a mixture of forest with small arable fields, is the most vulnerable to wildfire, but the wildlife connectivity is low. On the other hand, landscape type VI, which is predominantly forest, has a much lower vulnerability to fire and a much higher connectivity for wildlife than landscape type II

5 to 15 % of the small agricultural fields in a mosaic (fine-grained agricultural land use) within a matrix of more than 40 % of forested land cover.

In terms of modeling the landscape's multifunctional features, SISPARES has provided the geostatistical framework necessary to investigate functional effects of landscape structure that theoretically operate in opposite directions, such as the positive effect of forest connectivity on wildlife dispersal and the negative effect on forest wildfire vulnerability, as shown in Figure 8.6 (Martin-Martin et al. 2013).

Another important capability of SISPARES is its potential for investigating the similarities between Spanish landscapes and those from other countries. That potential has been explored in an initial comparative study of the structural and functional characteristics of the Spanish *dehesa* landscapes and Californian ranch landscapes. The results (Fig. 8.7) indicate high structural similarity between the two landscapes, especially in coastal Californian areas that were colonized by Spanish settlers and missionaries (Elena-Rosselló et al. 2013). The recent evolution of the landscape structures in both countries has shown a stable profile, which indicates that the land-use and cover types have stabilized, with only small changes towards increased recreational uses. Our analysis of the relationship between human population density and land use has shown a clear correlation with these trends.

Comparison between the landscape structures of the two areas is the first step in a deeper functional analysis of the vulnerability to wildfire in both areas. Preliminary analysis of the results indicates contrasting responses of the Spanish *dehesa* and Californian landscapes in terms of wildfire frequency: Spanish *dehesa* shows less-frequent wildfire occurrence.

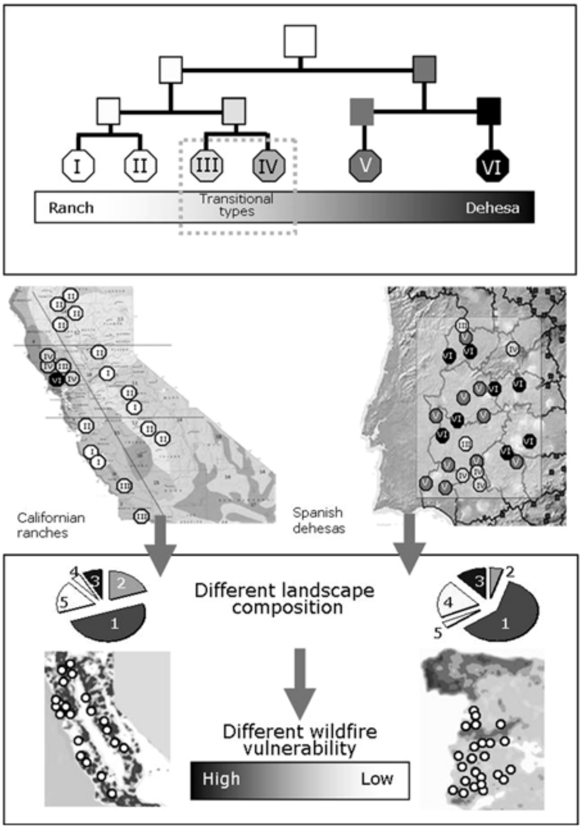

Figure 8.7 Comparison of Spanish *dehesa* landscapes from the REDPARES network with Californian ranch landscape samples assessed using the same methodology (Elena-Rosselló et al. 2013). A multivariate classification of landscape composition of both samples produced the dendrogram shown in the figure. The dendrogram ranks the landscape samples in a well-defined gradient from *dehesa* (*black*) to ranch (*white*). In the middle of the gradient, transitional landscapes exist in both study areas (*grayish*). The geographical position of the transitional landscapes clearly suggests a common historical origin. On the other hand, differences in landscape composition resulted in remarkable differences in wildfire vulnerability. (Land use/cover typology: (1) open woodland; (2) dense forest; (3) pastureland; (4) agricultural fields; (5) scrubland)

This difference can be explained by the different spatial patterns at a regional and national scale in terms of the physical environmental drivers (climate and geology). The *dehesa* covers an extensive area of relatively uniform plains in the southwestern Iberian Peninsula. In contrast, California shows elongated landscape patterns, with relatively narrow strips along the Pacific coast and around the San Joaquin Valley, as well as in the extended foothills of the Sierra Nevada. As a consequence of that spatial pattern, the average size of the *dehesa* farms is much larger than the size of ranches in the Californian landscape. Furthermore, the Californian ranches have a high proportion of flammable coniferous forest and chaparral ecosystems that increase the risk of wildfire compared with the Spanish vegetation. A final powerful driver responsible for the different responses relates to the distribution of the urban population. Although the human density is similarly low in both regions, the distance to metropolitan areas is significantly shorter in California.

8.5 Conclusions

The development of a holistic conceptual framework and the design of efficient monitoring procedures and their coordination have been undertaken in some projects in Europe, but more work is required, especially at the whole-Europe level. Fundamental research into the scientific basics of landscape monitoring should be seen as a priority (Urban 2002). The European Landscape Convention (COE 2000) encourages standardization of landscape-related investigations, so the development of suitable and comparable monitoring methods is necessary.

Any long-term monitoring system needs to be designed with the aim of detecting temporal and spatial trends to provide guidance in prioritizing management needs and determining the success of management activities. The objectives should include early recognition, assessment, and prediction of emerging landscape changes and the consequences in terms of future trends and risks (Syrbe et al. 2007, Urban 2002). Its key features should be reproducibility, standardization, and the ability to support the development and testing of hypotheses to determine those targets.

So that the data and results are easily available to researchers, partners, and the interested public, all the information generated by these initiatives should be prepared and communicated in a standardized, uniform, and unintimidating way. Standardization of data and models is perhaps the most important success factor for monitoring and assessing landscape change. The construction of versatile metadata systems should also facilitate sharing of data.

It is vital to remember that various difficulties may arise when developing a monitoring framework such as the one described in this chapter. For example, the temporal and spatial heterogeneity and the dependence upon initial conditions both require explicit attention. Readers should also remember that there will be a delay from the time when a decision is made to proceed with some form of management until the first results of that decision become apparent. The Spanish experience with SISPARES confirms that monitoring does not always yield fast results. The rate of change in many landscapes is slow enough that it may not be useful to specify a time interval between surveys that is less than 5 years. Longer periods provide enough time for detectable changes to occur, to record the necessary data, and to construct a database-management system to analyze the changes. In the Spanish case, the average minimum time between surveys has been 10 years (Martin-Martin et al. 2013, Ortega et al. 2012b). However, such problems should not discourage the development of monitoring programs.

After two decades of development, the most noteworthy conclusion from the Spanish experience is that SISPARES and its associated models have evolved towards greater complexity and greater power. As more information was introduced into the system, more complex and realistic models could be developed. These models have progressively shifted from static to dynamic forms, from structural to functional forms, from monofunctional to multifunctional forms, and from national to international scales. Consequently, SISPARES and its models have become increasingly useful for planning, restoration, conservation, and protection of Spain's landscapes.

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Chapter 9
Forest landscape management in response to change: the practicality

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Abstract In this chapter, we examined forest landscape management from a pragmatic (practical as opposed to idealistic) perspective. The discussion was framed in the context of the landscape: a spatially explicit geographic area consisting of recognizable and characteristic component ecosystems. This perspective provided two opportunities for management: the individual component ecosystems and the mosaic of ecosystems that form the landscape per se. A point of emphasis was that forest management is not a generic concept and requires specification of the purpose of management, the spatial unit(s) being managed, the type of forest being managed, and the projected desired outcome of management. Given these constraints, we considered how the principal drivers of landscape change (disturbances, climate, and domestication) influence forest management practices. We concluded with an examination of the concept of designed forest landscapes to provide human-valued goods and services and identified constraints to achieving this end.

9.1 Introduction

Forest landscapes exist in a variety of forms and are managed for multiple purposes. In this chapter, our goal is to examine forest landscape management from a pragmatic (practical as opposed to idealistic) perspective. The discussion is framed within the context of the principal drivers of forest landscape change: human intervention through domestication, natural disturbances, and climate. This approach is taken with the full recognition of contemporary and pervasive literature that deals with topics such as sustainability science (Wu 2013), landscape sustainability (Weins 2013), landscape services (Potschin and Haines-Young 2013, Termorshuizen

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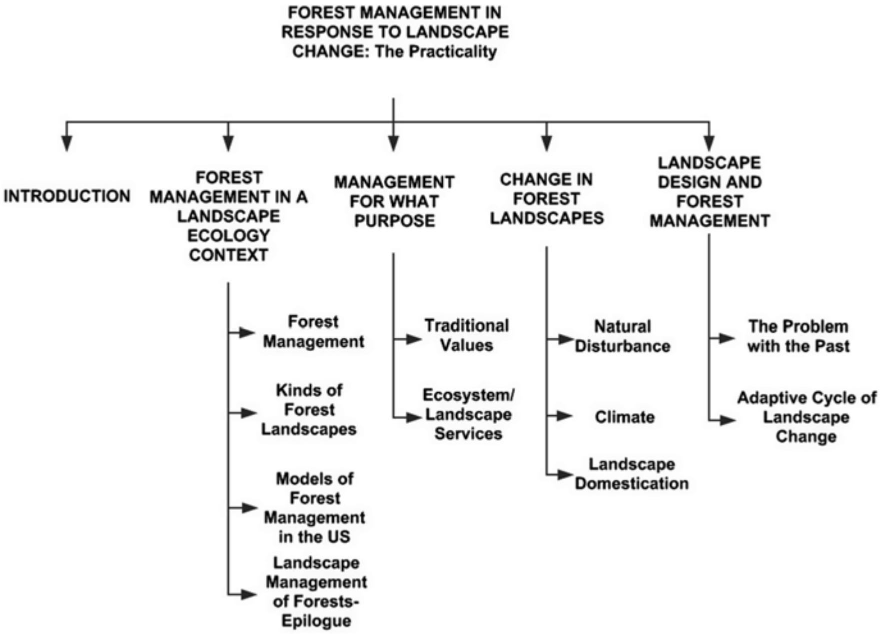

Figure 9.1 Summary of the topics considered in “Forest management in response to landscape change: the practicality”

and Opdam 2009), and designed landscapes (Musacchio 2009, Nassauer and Opdam 2008). Each of these subjects has relevance to forest landscape management in response to change, and in the following sections, we introduce concepts from this literature pertinent to the focus of the chapter. In particular we examine (1) forest management in a landscape ecology context, (2) management for what purpose, (3) change in forest landscapes, and (4) landscape design (Fig. 9.1). This investigation draws from domain knowledge associated with *landscape ecology*, which is the science that embraces the agenda of ecology in a spatially explicit manner (Coulson and Tchakerian 2010).

9.2 Forest management in a landscape ecology context

To *manage* is to take charge of or care of. As our focus centers on forest landscapes, how we define the spatial and temporal dimensions and the composition (structure) of the management arena, i.e., landscape, is of paramount importance. For our purposes, a *landscape* is defined to be a spatially explicit geographic area, i.e., an area defined by coordinates, consisting of recognizable and characteristic component

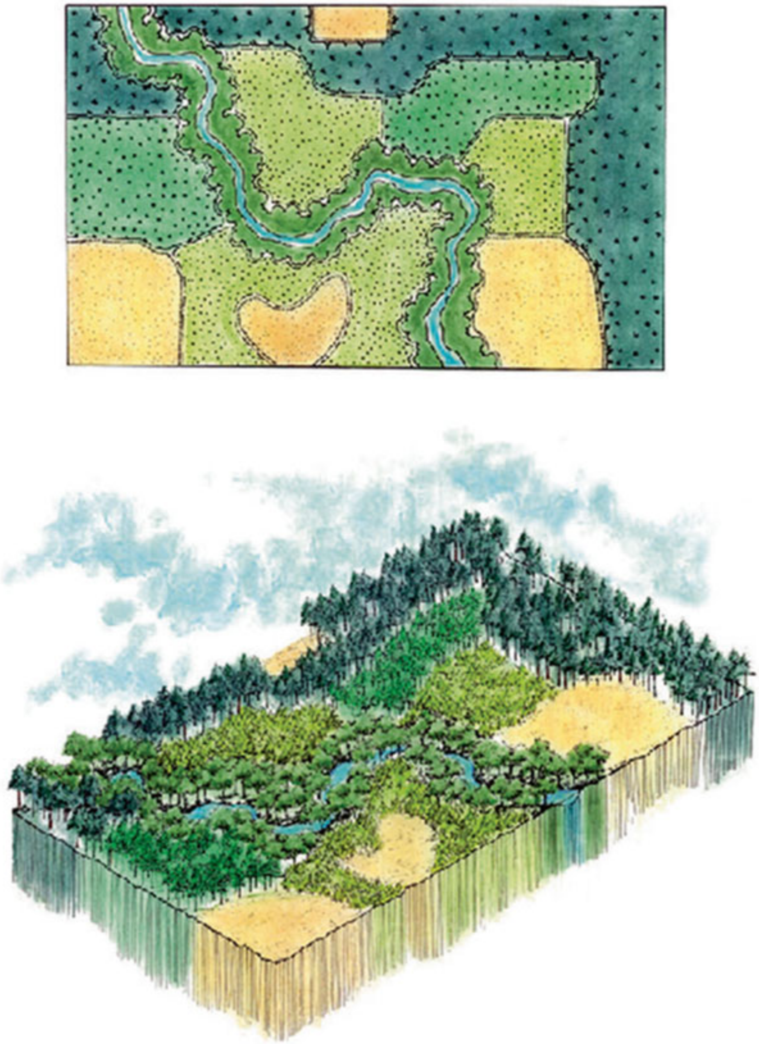

Figure 9.2 Overhead and plan views of a forest landscape, illustrating the component ecosystems and the mosaic pattern. *Source*: Coulson and Tchakerian (2010). Used with the permission of KEL Partners Inc.

ecosystems (Fig. 9.2) (Coulson and Tchakerian 2010). Given this definition, there are two opportunities for management: the individual component ecosystems and the mosaic of ecosystems that together form the landscape per se. Each of these management units is examined in the following sections.

9.2.1 *Ecosystems as building blocks of landscapes and management units*

The scientific literature dealing with ecosystems is inherently vague on spatial dimensions. Likens (1992) provided the first definition of ecosystem that included boundary as a component: "a spatially explicit unit of the Earth that includes all of the organisms, along with components of the abiotic environment within its boundaries". However, the issue of boundary and ecological functionality was recognized and dealt with in the European literature (reviewed by Zonneveld 1989) beginning in the 1930s through the concept of an *ecotope*, i.e., the smallest ecologically distinct landscape feature associated with a landscape mapping and classification system. Ecotopes are bounded ecosystems. The term *ecotope* is also synonymous with *site*, proposed by Bailey (1996).

Bounded ecosystems have historically been the fundamental unit of management in forestry and agriculture. The basic ecosystem processes (primary production, consumption, decomposition, and abiotic storage) are the targets for management. For example, gross primary production in forest ecosystems has been greatly increased by genetic selection, i.e., "tree breeding." For some forest tree species, the breeding programs have greatly altered the ratio of bole length to crown size and thereby increased merchantable biomass. Net primary production can be greatly increased by integrated pest management tactics that reduce herbivore consumption by pest species. Pesticide applications represent a common tactic in both forest and agricultural ecosystems. Decomposition in forest ecosystems is influenced by different types of site preparation practices: debris removal, furrowing, windrowing, etc. These practices also reduce export of nutrients by fluvial- and aeolian-mediated erosion and, along with fertilizer inputs, influence abiotic storage (Coulson and Tchakerian 2010). Finally, bounded ecosystems are generally the unit of harvest in commercial forests and therefore represent the source for one type of final ecosystem service: wood fiber for use by humans in building construction, paper production, and as fuel.

In summary, we define *forest ecosystem management* to be the orchestrated modification or manipulation of the basic ecosystem processes (primary production, consumption, decomposition, and abiotic storage) for desired human-defined ends. In traditional forestry, management centers initially on forest vegetation, and it is the domain of *silviculture*. The term *silviculture* is simply defined as the theory and practice of controlling forest establishment, composition, and growth. When practiced, this anthropocentric activity is place based, with the bounded ecosystem serving as the management unit. The knowledge base dealing with silviculture of many commercially important tree species is rich and extensive. This knowledge base is in part founded on a fundamental understanding of plant population and community dynamics and the tacit experience of forestry practitioners. Examination of how human intervention, climate change, and natural disturbance affect forest management practices begins with the component ecosystems that form the landscape.

9.2.2 Forest landscapes as management units

We are representing a landscape to be composed of multiple bounded ecosystems that together form a mosaic that is typically characterized by heterogeneity. The exchange of matter, energy, and information within the landscape is influenced by the kinds of ecosystems present and their spatial arrangement, i.e., the content and context of the building blocks. The ecosystems that characterize a specific forest landscape are defined fundamentally by topographic, climatic, and edaphic variables superimposed on an underlying geomorphology. As the climatic and edaphic variables change, so do the component ecosystems and their characteristic flora and fauna. By aggregating the component ecosystems that form a landscape, we have changed and expanded the basic management unit to include the mosaic. This landscape unit is considerably more complex than the ecosystem, and the management approaches and end points are much different.

Forest landscape management is the orchestrated modification or manipulation of landscape structure (components of the landscape and their linkages and configurations), function (the flux of energy, matter, and information within and among the component ecosystems), and rate of change (alteration in the structure and function of the ecological mosaic over time) to create dynamic mosaic patterns that provide human-valued goods and services. Again, landscape management is a place-based activity involving discrete human activities enacted within a spatially explicit land area organized as a mosaic of component ecosystems (Coulson and Tchakerian 2010). The *dynamic* feature of the definition implies that the mosaic patterns of forest landscapes develop and cycle through time in a predictable and sustainable way. However, there is no compelling evidence to suggest that forest landscape succession proceeded in this manner prior to the Anthropocene or that forestry practices today could be used to mimic the process if it did exist. The fundamental contemporary question is then how the drivers of change alter the processes that result in *observed* patterns of forest landscapes through time, i.e., the reverse of the pattern/process paradigm. The answers to this question represent the research, development, and applications agenda for landscape ecology and forest science for the foreseeable future. Musacchio (2009), Termorshuizen and Opdam (2009), Kates (2011), Turner et al. (2013), and Wu (2013) all provide lists of topics that identify different aspects of this agenda.

9.2.3 Kinds of forest landscapes

To this point, we have represented a forest landscape in a generic manner to be a mosaic consisting of multiple interacting ecosystems and management to be an anthropocentric activity directed to provision of goods and services. However, forests are also commonly classified to reflect specific management goals, structural complexity, spatial extent, different levels of human intervention, and ownership.

For discussion purposes, we recognize six different types of forests: conservation forests, extensively managed forests, intensively managed forests, specialized forestry settings, agroforests, and urban/suburban (*peri-urban*) forests. This classification differs modestly from that used by FAO (2011). The following is a brief description of the forest types.

The *conservation forest* is considered to be the natural or nominal condition. These forests are not managed for the production of goods and services, but, rather, are left to develop through natural processes of ecological succession. These forests are generally protected because of unique physical attributes and scenic beauty. Within the United States National Forest System, the USDA Forest Service sets aside conservation forests. These forests are designated as wilderness areas. Direct human intervention is minimal. Natural disturbances are left to follow their normal course. The biotic response to climate change is based on natural history traits that include inherent tolerance (or intolerance) to extreme and cyclic conditions. Persistence of conservation forests, given little human intervention, is often a function of their spatial extent. The cumulative impact of the disturbance regime has to be smaller than the area protected.

In *extensively managed forests*, human use of multiple resources (timber production, fish and wildlife, hydrology, recreation, grazing, etc.) is recognized. Extensively managed forests are often isolated from large human population centers, e.g., boreal forests of Canada, Alaska, and Russia. As such, these forests are used for the production of goods and services but, because of their remote locations, are modestly managed. Natural disturbances (such as wildfire and insect outbreaks), along with human intervention through harvesting, play important roles in shaping the structure and composition of extensively managed forests (Berg et al. 2006, Williams and Birdsey 2003). Again, climate change adds or extracts species as dictated by their tolerances.

Intensively managed forests are the focus of much of the commercial forestry practiced worldwide. The fundamental unit of silviculture is the bounded ecosystem (as defined above). Management practices often involve clear felling (cutting of all trees from a site), site preparation, fertilization, replanting with genetically selected species, etc. Landscape heterogeneity is reduced by single-species plantings. Age-class distribution is a function of harvest schedules. Human intervention is extreme. The forest landscape is highly modified relative to the nominal state. Management also includes efforts to reduce the impact of natural disturbances, e.g., fire control and suppression of insect and disease outbreaks.

Specialized forestry settings include nurseries (for the production of seedlings and ornamental plants), seed orchards (for the production of genetically selected seeds for reforestation), Christmas tree plantations, etc. The specialized settings require considerable care and maintenance and management activities resemble agriculture more than typical forestry practice, i.e., the emphasis is on cultivation rather than silviculture. Again, the fundamental unit of management is the bounded ecosystem.

An *agroforest* consists of a blend of agricultural crops and forest trees. *Agroforestry* deals with using trees on farms. There are two categories of the practice: simultaneous (trees, crops, and/or animals are grown together at the same time on the same parcel of land) and sequential (crops and trees take turns in occupying most of the same space, e.g., slash and burn agriculture). Agroforestry is an ancient practice primarily associated with tropical environments. It results in the production of both foodstuffs as well as forest products. Agroforests share some characteristics of natural forests in that they generally consist of multiple strata, contain large and mature trees, and have a shade-tolerant understory.

The *urban/suburban* (*peri-urban*) *forest* is a broad designation that includes a variety of settings, e.g., residential neighborhoods, parks, trees along city streets, etc. Urban/suburban forests can be remnants from commercial forests that have been encroached upon through development, they can be the result of plantings, or they can represent a combination of both native and introduced plant species. Their principal purposes are for esthetic enjoyment and as buffers to weather. Urban/suburban forests are highly modified by humans and are often created and designed to provide an illusion of the natural state. However, they can also serve to provide goods and services, e.g., refuges of biodiversity, habitat for wildlife, food (e.g., nuts and fruits), etc.

The point of this discussion of the different forest types is to emphasize that the ecological, economic, social, and political impact of the drivers of change is different for each landscape. Regardless of whether the forests are valued using traditional measures (timber production, water, hydrology, recreation, grazing, etc.) or ecosystem (landscape) services, the differences remain. In some instances, forest management can influence the degree of impact. For example, in intensively managed forest landscapes, natural disturbances such as wildfire and insect outbreaks can often be suppressed using remedial tactics. Human-mediated change, such as the introduction of invasive plant species, can be addressed through vegetation management practices. Response to climate change by conversion to another plant species or management approach at the ecoregion scale is generally not a viable solution for intensively managed forest landscapes. Guldin (2013) provides a compelling example illustrating factors associated with converting naturally regenerated pine (*Pinus* spp.) stands to intensively managed plantations in the southern United States. The conclusion was that a "climate-change conversion program" would be prohibitive given the cost, acreage involved, and public and nonindustrial private forest land ownership. However, for urban/suburban landscapes, species replacement is a viable approach. For example, Dutch elm disease (*Ophiostoma ulmi* and *O. novo-ulmi*) eliminated American elm (*Ulmus americana*) as a prominent landscape tree in urban and suburban northeast and north central United States. Over a period of several decades, American elm was replaced with a variety of hardwood species that reestablished many of the functional roles this tree once provided. Ironically, ash (*Fraxinus* spp.) was one of the recommended replacement species, and this genus is now host for the emerald ash borer (*Agrilus planipennis*). This introduced invasive species is a significant mortality agent for ash species throughout much of their range in the United States, and it is having an impact on urban and suburban forest landscapes comparable to that of the Dutch elm disease several decades earlier.

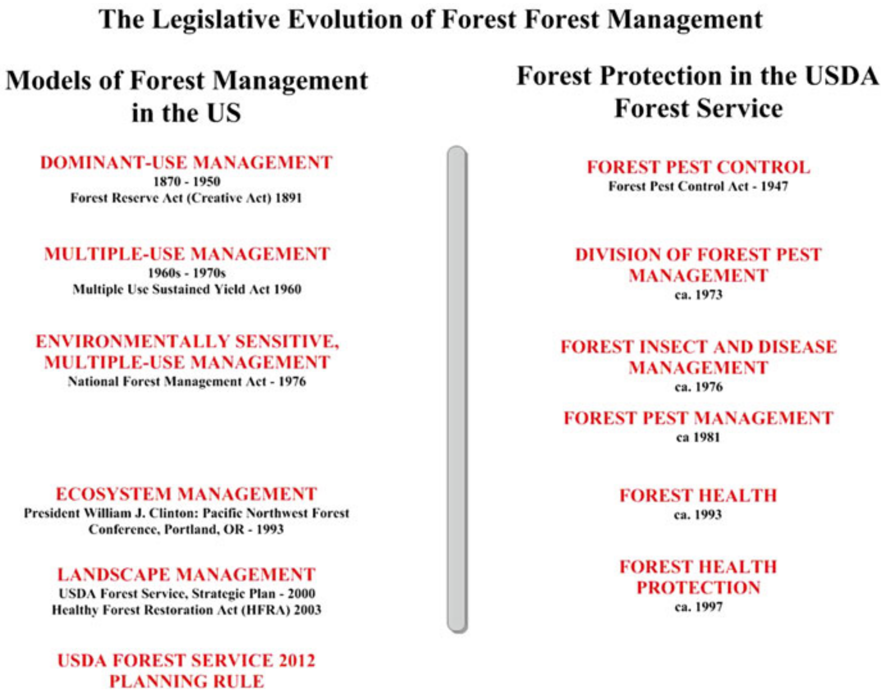

Figure 9.3 The history of forest management models employed by the USDA Forest Service in National Forests in the United States

9.2.4 Models of forest management

In the United States, several different models have been used in the past to guide forest management practice on public lands. Each model represented the prevailing thought of the time on how forests should be managed for the public good. Authorization came from legislative mandates and, in turn, each model was implemented throughout the National Forest System. Figure 9.3 summarizes the legislative history of forest management in the United States, as well as the response by the extension arm of the USDA Forest Service, Forest Health Protection, to the different models. Beginning in 1870, six different models have been used: dominant-use management; multiple-use management; environmentally sensitive, multiple-use management; ecosystem management; landscape management; and the forest plan. Coulson and Stephen (2006) examine the basic features of each model in detail, and here we simply identify the point of emphasis and the reaction of the Forest Health Protection agency. The purpose of the following commentary on the different models of forest management is to provide perspective for the prevailing view of forest management.

The first model of forest management in the United States was known as *dominant-use management*. This model followed from the Forest Reserve Act

(Creative Act of 1891) and persisted into the 1950s. The approach emphasized production of economically valuable species. Typically the goal was to maximize production. Protecting the means of production was also a goal, and this activity included fire control and insect suppression. In recognition of the complexity of forest protection, the Forest Pest Control Act of 1947 established the extension arm of the USDA Forest Service, and this agency was first named *Forest Pest Control* (Fig. 9.3).

The second model of forest management was known as *multiple-use management*. This model was authorized by the Multiple-Use Sustained Yield Act of 1960. The principal new feature centered on the recognition that forests provided a variety of goods and services that were valued by humans, in addition to timber production. The goal of multiple-use management was to maximize utilization of different resource values and do so on a sustainable basis. The term *sustainability*, in this context, meant continuous production of desired outputs, e.g., a non-declining and even flow in the case of wood fiber. No single resource was to be valued more than any other. This model provided the legal basis for management of United States National Forests in the 1960s and 1970s. In response to the broader management charge, the extension arm of the USDA Forest Service was renamed the *Division of Forest Pest Management* in 1973.

The third model was referred to as *environmentally sensitive*, *multiple-use management*. This model was authorized through the National Forest Management Act of 1976. The model represented forests as systems with interacting biotic and abiotic components and recognized that production was subject to ecological and environmental constraints. Important management concepts included sustained yield, minimizing negative environmental impacts, and protecting species diversity. This model recognized that different management approaches were possible, included a means to obtain input from stakeholder groups, and provided for the creation of a "Committee of Scientists" (to advise the Secretary of Agriculture on forest management issues). To accommodate the new model, the forest protection enterprise was renamed *Forest Insect and Disease Management* in 1976 and later changed to *Forest Pest Management* in 1981. The model was an abject failure but led to the next chapter, ecosystem management.

The fourth model was referred to as *ecosystem management*. This model followed from the Pacific Northwest Forest Conference in 1993, convened in response to controversy over forest management on public lands in the Pacific Northwest, United States. In contrast to the previous anthropocentric concepts of forest management, ecosystem management was a biocentric (biologically centered) concept. The goal was to maximize ecological integrity or "health", subject to the need to allow for sustainable human use. Ecosystem protection was the first priority and human-valued goods and services the second. Ecosystem management represented a significant departure from the production-driven models described above that emphasized forest resources. Although laudable in intent, ecosystem management was an elusive concept for both ecologists and foresters. Nevertheless, in 1993 the forest protection enterprise was renamed *Forest Health* and changed again in 1997 to *Forest Health Protection*.

The fifth model was referred to as *landscape management*. Authorization came from two sources: the Forest Service Strategic Plan of 2000 and the Healthy Forest

Restoration Act of 2003. In this model, the basic management unit was the landscape (as defined above). Emphasis was placed on the functional interconnections among landscape components as well as the production of human-valued goods and services. This integrative perspective resulted in an ecocentric management concept that combined both the anthropocentric and biocentric views of previous models. Implementation of the landscape management concept, again, proved to be problematic. However, the Healthy Forest Restoration Act did provide a set of guidelines for directed actions that were intended to adjust the landscape environment to approximate previous states, which were presumed to be better than the existing conditions.

The current view of forest management on public lands in the United States is defined by the National Forest System land management *planning rule*. The planning rule was implemented in 2012 and now serves as the guideline for USDA Forest Service management of the National Forest System. The intent of the planning rule is "to ensure that (management) plans provide for the sustainability of ecosystems and resources; meet the need for forest restoration and conservation, watershed protection, and species diversity and conservation; and assist the Agency in providing a sustainable flow of benefits, services and uses of National Forest System lands that provide jobs and contribute to the economic and social sustainability of communities."

The point of this discussion of the different models used in management of public forest lands in the United States is to emphasize the dramatic change in philosophy and practice that has occurred over a brief period of about 150 years. Because management of public lands in the United States is enacted through a legislative process and implementation is charged to a governmental agency (the USDA Forest Service), tracing changes in the model was straightforward and tractable. Other countries with a legacy of governmental management of public forest lands likely have undergone dramatic changes in approach as well. Why the models of forest management changed is subject to speculation. The following are five plausible reasons: (1) perceived deficiencies or inadequacies in the approach or outcome of management, (2) accommodation of advances in technical information and tacit knowledge of forest management, (3) expansion of the values for which forests are managed, (4) the perceived need to protect and preserve forest lands in perpetuity, and (5) recognition of the contribution forest lands provide in regulating global atmospheric processes.

9.2.5 Landscape management of forests: epilogue

In the preceding subsections, the goal was to frame forest management in an explicit manner. To this end, we addressed three issues. First, we specified and defined the actual spatial units of forests that are amenable to human intervention and that are directly affected by the drivers of change (ecosystems and landscapes). Second, we identified different kinds of forests and emphasized that each had specific management goals and objectives, constraints on the degree of management that is possible or desirable, and unique valuation systems for scoring the impacts of change. Third, we examined six different models of forest management employed in the United States (implemented over a period of less than 150 years) and identified that there has been an evolution in philosophy ranging from resource mining to an emphasis on a science-based approach for understanding the relation of the forest environment and the production of human-valued goods and services. The point of emphasis is that forest management is not a generic concept and requires specification that includes the purpose of management, the spatial unit(s) being managed, the type of forest being managed, and the projected desired outcome of management. The anthropocentric models of management were appealing for their simplicity, i.e., an emphasis on the production of human-valued goods and services. The models of forest management became intractable with the presumption that there was a well-defined scientific recipe (with ingredients from ecology) that could be applied to guide the enterprise.

9.3 Purpose of forest management

Forest management is a purpose-driven business. The specific values for which forests are managed can be summarized categorically (which is the traditional approach) or by the concept of ecosystem services (which is a contemporary view). Following, we examine each and also consider the relation between the two approaches.

9.3.1 Traditional values

The traditional purposes for forest management have been summarized categorically as “values”. The basic categories of management initially centered on timber production, hydrology, fish and wildlife, recreation, and grazing; and this list was later expanded to include real estate, biodiversity, endangered species, cultural resources, and non-wood forest products. Each of these categories represents a multifaceted subject domain, and all include as an endpoint something of value to humans. The value can usually be expressed in monetary terms, which facilitates a place-based calculation of the impact of the drivers of change. All of the constraints identified (forest type, location, spatial and temporal scale, management objective, etc.) come into play in the valuation process.

9.3.2 Ecosystem services

An alternative approach to forest valuation is summarized in the concept of ecosystem services. Simply defined, *ecosystem services* are "the benefits of nature to households, communities, and economies" (Boyd and Banzhaf 2007). The concept grew from an interest in a science-based approach to managing the environment to enhance human welfare. Scientific and social interest in the subject of ecosystem services followed from the publication of the *Millennium Ecosystem Assessment* (MEA 2005) and the topic has since received considerable commentary in the landscape ecological science and environmental economic literature, e.g., Boyd and Banzhaf (2007), Termorshuizen and Opdam (2009), Mace et al. (2012), Turner et al. (2013), Wu (2013), Marta-Pedroso et al. (2014), etc.

Given our focus on the purposes of forest management in relation to drivers of change, there are three features of the concept of ecosystem services that are relevant: the economic component, the relation between scientific and social perspectives, and the landscape context. Each of these topics is examined below.

9.3.2.1 The economic perspective on ecosystem services

There is an economic component of the concept of ecosystem services that is closely tied to interests in systems for environmental accounting and performance assessment. "Services" are the units these systems track and measure. The economic perspective is particularly useful for defining what constitutes an ecosystem service, given that there are several taxonomies. To be useful in environmental accounting systems, ecosystem services must be defined by quantity (units) and price. This constraint requires a precise definition of ecosystem services. To address this critical requirement, Boyd and Banzhaf (2007) distinguish between final and intermediate ecosystem services. *Final ecosystem services* are components of nature directly enjoyed, consumed, or used to yield human well-being, e.g., wood fiber, clean water, scenic beauty, etc. Final ecosystem services are end-products of nature. *Intermediate ecosystem services* are the biological, physical, and chemical processes that lead to the end-products. Nutrient cycling is an example of an intermediate ecosystem service. The value of intermediate services is in the provision of final ecosystem services. The *Millennium Ecosystem Assessment* (MEA 2005) defined ecosystem services as supporting, regulating, provisioning, and cultural. The first and second categories are examples of intermediate services, and the third and fourth are final services. Furthermore, there is a fundamental distinction between the quantity (or physical measure) of ecosystem services and the value of those services. The social value of ecosystem services is spatially explicit, i.e., ecosystem services are not spatially fungible or subject to spatial arbitrage (Coulson and Tchakerian 2010). The categorical forest management values identified above are examples of final ecosystem services in that they can be characterized by quantity and price.

9.3.2.2 The relation of scientific and social perspectives of ecosystem services

We have represented a landscape in a scientific context to be an eco-physical entity, i.e., an integration of the biotic and abiotic components within a spatially explicit boundary. An alternative perception considers the landscape to be a cultural unit (Nassauer 1997; Wu 2011, 2013). In this view a "landscape is… a heterogeneous mosaic of ecosystems that is constantly being adapted by humans to increase its perceived value" (Nassauer and Opdam 2008). The bridge between the ecological and cultural views of a landscape is through the structure–function–value chain (Termorshuizen and Opdam 2009). The ecological concept of landscape centers on the structure–function portion of the chain and deals specifically with processes. The processes are the intermediate ecosystem services, as defined in the previous section. From a management perspective, the basic question is how the drivers of change affect the governing processes of the structure–function (pattern/process) relationship that result in a desired forest landscape mosaic. The answers to this question are clearly the domain of scientific inquiry. The cultural concept of landscape centers on the function–value portion of the chain and deals with end-products of management. The end-products are final ecosystem services, as defined in the previous section. The basic forest management question is how the drivers of change affect the values placed on the end-products. The answers to this question still involve scientific inquiry but also require economic and social assessment (Termorshuizen and Opdam 2009).

9.3.2.3 Landscape context of ecosystem services

The acknowledgment that forest landscape management includes consideration of both eco-physical and cultural perspectives (linked through the structure–function–value chain) leads to an expanded view of the concept of ecosystem services. Final ecosystem services are often associated with a component ecosystem that is an element of the landscape mosaic, e.g., fish harvested from a lake. However, the clean water and habitat structure that provided the environment for the fish resulted from processes (intermediate services) associated with adjacent ecosystems, e.g., filtration, nutrient inputs, etc. Furthermore, in some cases the final ecosystem service is the result of an ensemble of interacting ecosystems, e.g., a scenic vista. In this case, the final ecosystem service, esthetic enjoyment of a viewshed, results from a unique placement of different ecosystems in the mosaic. Both ecosystem services can be managed for, e.g., protecting the intermediate services and regulating harvest, in the case of the fish in the lake; and preservation of the viewshed by excluding intrusions (e.g., roads, built structures, etc.), in the case of the scenic vista. So, in addition to the eco-physical/cultural perspective, landscape heterogeneity must also be included in any discussion of management for ecosystem services.

The concept of ecosystem services is evolving. Synonyms include terms such as natural capital, environmental services, green services, and landscape services. All emphasize a connection between the eco-physical environment (ecosystem and landscapes) and human values (Termorshuizen and Opdam 2009).

9.4 Change in forest landscapes

Landscape change deals with the alteration of the structure and function of the landscape environment in space and through time. Following, we examine three principal drivers of change in forest landscapes: natural disturbance, climate, and domestication. Coulson and Tchakerian (2010) provide an expanded discussion of the topic focused on landscape-cover change, landscape-use change, effects of landscape change on living organisms, and development of pattern in mosaic landscapes.

9.4.1 Natural disturbance in forest landscapes

The concept of *disturbance* is fundamental to a discussion of change in forest landscapes. The term is used interchangeably with *perturbation* and *stress*. Although variously defined in the literature, for our purpose, a *disturbance* is an initiating cause (a physical force, a process, or an event) that produces an effect (consequence) that is greater than average, normal, or expected. This definition requires a reference state (i.e., a mean condition bounded by a range in variation), as well as specification of spatial and temporal boundaries. The utility of a rigorous definition of disturbance is to separate circumstances where an initiating cause → consequence relationship is considered to be a disturbance in contrast to a normal or expected event. For example, when does a fire in a fire-climax forest (e.g., chamise chaparral, *Adenostoma fasciculatum*) cease to be a normal or expected event and become a disturbance (Coulson and Tchakerian 2010)?

A disturbance event can be characterized in a variety of ways: e.g., it can be biotic or abiotic in origin; it can be distributed, targeted, diffuse, or patchy in space; it can be frequent, rare, or periodic in occurrence; etc. The scope of the concept of ecological disturbance in forests is immense. However, there are several recurrent themes that center on the effects of disturbance on forest landscape transformation processes, primary production, nutrient cycling, biodiversity, endangered and threatened species, and population dynamics of selected species. The impacts can be assessed from ecological, economic, social, and political perspectives and also evaluated in the context of ecosystem services (Coulson and Tchakerian 2010).

Evaluating the consequences of disturbance events for a forest landscape requires observation over an extended time frame. Five human generations or approximately 100 years (20 to 25 years × 5) is often used as a reasonable temporal boundary.

The ensemble of disturbance types associated with a specific landscape is referred to as a *disturbance regime*, and impact evaluation involves an assessment of the aggregate regime.

9.4.2 Climate and forest landscape change

For our purposes, *climate change* is "a departure from the expected average weather patterns ('climate normals')" (NOAA 2013) for a specified forest landscape. As a driver of change, we are particularly interested in how variation in the expected state of the atmosphere (as defined by variables such as temperature, precipitation, wind speed, etc.) affects the biotic communities associated with forest landscapes. Effects on the biota are a function of tolerances to changes in the weather parameters (e.g., in means, extremes, variability, and seasonality) and adaptability to increased frequency and intensity of atmospheric-initiated disturbance events (floods, droughts, storms, fire, pestilence, etc.) (Bellard et al. 2012, Iverson et al. 2014).

The effects of climate change on a forest landscape are manifested in several ways: species can be eliminated, the timing of species life cycle events can be altered, the distributional range and extent of species can be expanded or reduced, trophic structure can be disconnected, and biotic regulation of ecosystem processes can be disrupted or eliminated. Within an ecological time frame, the living organisms have limited options to accommodate new climatic conditions: response in space through various dispersal mechanisms and response in time through adjusting life history strategies (e.g., phenology, diurnal rhythms, etc.). Forest management options in response to climate change are limited and again constrained by the purpose of management, the spatial unit(s) being managed, the type of forest being managed, the projected desired outcome of management, and the market value of final ecosystem services.

Forest trees are generally long-lived species, and they move twice in their life cycle, once as a seed and again as a pollen grain. Natural regeneration of forest landscapes is a function of the success of this movement and is constrained, as defined above, by space and time. Replacing species that are poorly adapted to a changing climate regime is feasible in agroforests, specialized forestry settings, and urban/suburban landscapes, but problematic in intensively and extensively managed forests. Certainly, forest managers have the option to substitute species at replanting following harvest or after a broadscale natural disturbance (e.g., from *Pinus* to *Eucalyptus*). However, the landscape ecological consequences of this action are speculative, and markets may not exist for the products of the substitute species. For the reasons outlined by Guldin (2013), orchestrated substitution of species at the ecoregion scale is not economically feasible.

Biotic responses to modest changes in weather parameters can have a profound effect on forest landscapes. Bark beetle herbivory in coniferous forests provides a good example. Small increases in temperature can trigger outbreaks through two different mechanisms: accelerated insect development time (reflected in voltinism, the number of generations the insect passes through each year) and range expansion that exposes greater numbers of hosts or uncommon host species. Berg et al. (2006) attributed in part the massive outbreaks of the spruce beetle (*Dendroctonus rufipennis*) in spruce forests (*Picea* spp.) of Alaska (United States) and the Yukon Territory (Canada) in the 1990s to elevated temperature that reduced winter mortality and increased insect development time from a 2-year life cycle to a 1-year cycle. Logan et al. (2010) documented persistent outbreaks of the mountain pine beetle (*D. ponderosae*) in whitebark pine (*Pinus albicaulis*) in high-elevation forests in Yellowstone National Park (United States). Generally, whitebark pine forests are inaccessible to the insect because the lower temperature regimes are unsuitable for its development. This insect has also been responsible for the massive outbreak in lodgepole pine (*Pinus contorta*) throughout the Pacific Northwest of the United States (U.S.) and Canada.

9.4.3 Domestication and forest landscape change

The term *landscape domestication* is defined as the activities of humans that structurally shape and functionally modify landscapes to satisfy basic human needs. With some concession to simplification, the basic human needs include adequate food, water, housing, energy, health, and cultural cohesion. In the context of forests, management actions associated with domestication are initiating causes that produce predictable changes in the forest landscape use. The management intent is for the changes to provide ecosystem services that directly translate to human needs, as defined above (Coulson and Tchakerian 2010). The subject of humans as agents of change in forest landscapes has been examined in detail by Farinaci et al. (2014) and is by far the most significant driver.

Climate, edaphic characteristics, and topographic features (surface geometry and landform) delineate logical physical boundaries for landscape-use change as directed to forest management. The different kinds of forest landscapes (described above) are largely defined by these structuring variables. The social boundaries for landscape-use change are rooted in issues associated with demographics, economic systems, sociopolitical policy, and technical and scientific developments (Farinaci et al. 2014).

9.5 Landscape design and forest management

Previously, we defined *forest landscape management* to be the orchestrated modification or manipulation of landscape structure, function, and rate of change to create dynamic mosaic patterns that provide human-valued goods and services in perpetuity. Landscape management was also described as a purpose-driven and place-based activity involving discrete human activities enacted on a spatially explicit land area organized as a mosaic of component ecosystems. This concept of management is perhaps praiseworthy in intent, but could it be implemented? In the following we consider the practicality of a science-based approach to forest landscape design for management purposes.

9.5.1 The premise of forest landscape design for the sustainable production of goods and services

The recognition that forest landscapes are structured as a mosaic of interacting ecosystems and that this ensemble forms a template that could be modified and manipulated in prescribed ways for the production of human-valued goods and services logically lead to consideration of the use of design concepts for optimization purposes. Nassauer and Opdam (2008) and Musacchio (2009) advocated the use of a science-based approach to landscape design for management purposes. The concept is defined as follows: "… design [is] an intentional change of landscape pattern, for the purpose of sustainably providing ecosystem services while recognizably meeting societal needs and respecting societal values. Design is both a *product*, landscape pattern changed by intention, and the *activity* of deciding what that pattern could be" (Nassauer and Opdam 2008). A fundamental component of this definition is the notion of sustainability, which is a term also subject to broad-based interpretation. Wu (2013) defined the concept as follows: "landscape sustainability is the capacity of a landscape to consistently provide long-term landscape-specific services essential for maintaining and improving human well-being". Wu (2013) further advocated the utility of a scientific enterprise based on the concept: landscape sustainability science ("a place-based use-inspired science of understanding and improving the dynamic relationship between ecosystem services and human well-being in changing landscapes under uncertainties arising from internal feedbacks and external disturbances"). Further commentary on this subject is provided by Kates (2011). The concepts of scientific design of landscapes, landscape sustainability, and landscape sustainability science are certainly laudable and, in part, represent visions that have accompanied the maturation of the discipline of landscape ecology. However, there are significant obstacles to their application in practical forest management.

9.5.2 The practicality of landscape design in forest management

The practical issues associated with implementation of design concepts that lead to the sustainable production of ecosystem services resulting from forest management center on the disutility of past experiences for predicting future events and the absence of a conceptual model of how change mechanisms influence mosaic pattern in space and time. These subjects are discussed below.

9.5.2.1 The problem with the past: interaction of the drivers

By definition, managed forest landscapes have been modified or manipulated through human intervention. Landscapes where climatic, edaphic, and topographic conditions are favorable for forest production have been utilized for similar purposes for many generations, although there is interplay among agriculture landscape use, forest landscape use, and landscape domestication. In the case of intensively managed forest landscapes, agroforests, specialized forest settings, and urban forests, the modifications and manipulations may have occurred multiple times. For example, the forest region of the Southern United States has undergone massive and multiple changes following European settlement. Initially, much of the native forested land was converted to cotton agriculture, and the revenue generated from this enterprise provided the financial resources that fueled the development of the United States economy. However, following depletion of soil fertility, cotton agriculture was abandoned and converted (or reconverted) to pine (*Pinus* spp.) production. The conventional forestry practice was to reestablish the forests, following harvest, through natural regeneration, and this approach was subsequently replaced by planting genetically selected pine seedlings. This practice has now been utilized through four to six forest generations in the Southern United States. Furthermore, modern agricultural practices and the competing values of cotton fiber vs. wood fiber have resulted in extensive plantings re-devoted to cotton production. The point of this example is to illustrate the fact that managed landscapes today often do not resemble the past state. Additionally, much of this chapter has addressed how the drivers of change (climate, disturbances, and domestication) create new conditional states. We have addressed each driver independently, but it is important to recognize that their influences on forest landscapes are complementary and the consequences of the interactions are unknown. In the context of forest management, how the drivers of change interact, coupled with multiple historical landscape uses, challenges the utility of past experience for predicting future events.

9.5.2.2 Adaptive cycle of landscape change

Using established design concepts to configure forest landscapes for management purposes is not a new concept. One of the first efforts in this regard was provided by Diaz and Apostol (1992) who defined a systematic approach for incorporating emerging concepts of landscape ecology into forest management planning. This approach was referred to as forest landscape analysis and design, and it featured a landscape analysis component (with five steps [landscape elements, landscape flows, relation between landscape structure and flows, process of landscape change, and linkages]) and a landscape design component (with three steps [landscape patterns from GIS databases, landscape pattern objectives, and forest landscape design]). The culmination of the process was a design plan tied to a spatially explicit forest landscape. This approach remains a useful planning tool today. The penultimate step in the process called for the definition of a landscape pattern that would lead to the production of specific ecosystem services in the future. This juncture is where current knowledge of landscape ecology, forest management, and cultural geography is inadequate.

The fundamental problem in using design concepts in forest landscape management is that we do not have a conceptual model (metaphor) that addresses spatially explicit dynamic development and change in landscape mosaics in space and time. An analogous situation exists in the computer science discipline of artificial intelligence (AI). One branch of this discipline, expert systems, has been useful in mimicking human problem solving in practical situations. It is possible to develop computer code that can process the logic associated with "if–then" rules and, with the addition of fuzzy mathematics (to deal with uncertainty), mimic successfully a problem solving approach used by humans. The branch of AI that deals with pattern matching did not fare as well. Humans see and recognize patterns very well, but how we accomplish this task is without a conceptual model. So, it has not been possible to write computer code that even remotely mimics the capabilities of humans in pattern identification.

The adaptive cycle (Holling 1992) and panarchy (Gunderson and Holling 2002) have been useful organizing constructs for conceptualizing change and the development of complex ecological and economic systems. Landscape ecology has not provided a conceptual model for the succession of mosaic pattern in natural (or managed) landscapes. Does mosaic pattern in landscapes follow the conservation, release, reorganization, and exploitation sequence of Holling's adaptive cycle, and how would this scheme play out in a spatially explicit forest landscape?

Simulation modeling for forest landscape dynamics is an active component of landscape ecology research, and the utility and limitations of this approach for studying forest landscape dynamics have been examined by Gustafson (2013). Forest landscape modeling was compartmentalized into two basic methodologies: phenomenological (empirical or statistical) models and mechanistic (process-based) models. Emphasis was placed on the limited utility of phenomenological models based on retrospective examination of past conditions. Process modeling, based on "first principles" (an approach advocated by P.J.H. Sharpe in the early 1970s), perhaps provides a means for understanding the complex behavior of forest landscape dynamics. However, the current state of understanding of forest management does not provide design principles that allow for the projection of the production of sustainable ecosystem services.

9.6 Epilogue

In this chapter, we examined forest landscape management from a pragmatic (practical as opposed to idealistic) perspective. The discussion was framed within the context of the principal drivers of forest landscape change. Four components of forest management were examined and are briefly summarized below: management in a landscape ecology context, management for what purpose, change in forest landscapes, and landscape design.

1. The discussion of forest management was considered in the context of the landscape: a spatially explicit geographic area consisting of recognizable and characteristic component ecosystems. This perspective provided two opportunities for management: the individual component ecosystems and the mosaic of ecosystems that form the landscape per se. Traditionally, silviculture has guided management of the forest ecosystem unit. Management of the mosaic pattern to produce human-valued goods and services is a study in progress. This study is complicated by the existence of different types of forest landscapes, each with unique management goals. Six different forest management settings were examined. Additionally, the philosophical basis for how to manage forests is unsettled, and we illustrated this issue by an examination of the approaches implemented on the National Forest System in the United States over a 150-year period. The conclusion was that forest management is not a generic concept and requires specification that includes the purpose of management, the spatial unit(s) being managed, the type of forest being managed, and the projected desired outcome of management.
2. As the purpose of forest management is of paramount importance, we next examined this subject from two perspectives: traditional values and ecosystem (landscape) services. Ecosystem services were defined to consist of intermediate services (processes) and final services (products). We emphasized that the purposes of forest landscape management included consideration of both eco-physical and cultural perspectives. The bridge for the eco-physical and cultural perspectives of landscape was through the structure–function–value chain. The eco-physical concept of landscape centered on the structure–function portion of the chain and dealt specifically with processes (intermediate services). The cultural concept of landscape centered on the function–value portion of the chain and dealt with the end-products of nature. The basic forest management question was posited as how the drivers of change affect the values placed on the end-products of management.
3. Landscape change was defined to be the alteration of structure and function of the landscape environment through space and time. The principal drivers of change included natural disturbances, climate, and landscape domestication. A *disturbance* was defined to be an initiating cause that produces an effect that is greater than average, normal, or expected. Disturbance characteristics and impacts on landscapes were examined. Climate change was defined as a departure from the expected average weather patterns for a specified forest landscape. Effects of climate change on forest landscapes and biotic responses were examined. The relations between climate change and disturbance were illustrated through examples of elevated herbivory triggered by a modest change in temperature. *Landscape domestication* was defined as the activities of humans that structurally shape and functionally modify landscapes to satisfy basic human needs. Landscape domestication was identified to be the most significant driver of change.

4. We concluded with an examination of the plausibility of using design concepts as a means of modifying and manipulating landscape mosaics for the production of human-valued goods and service. This “design-in-science” concept was considered in the context of the pervasive themes in landscape ecology that deal with sustainable landscapes and sustainability science. Practical issues associated with implementation of landscape design concepts were examined in the context of the disutility of past experiences for predicting future events and the absence of a conceptual model for how change mechanisms influence mosaic pattern in space and time.

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Chapter 10
Forest landscape ecology and global change: what are the next steps?

João C. Azevedo, Maria Alice Pinto, and Ajith H. Perera

Abstract In this chapter, we summarize current trends and challenges and future research directions in forest landscape ecology and in management related to global change. We discuss the available knowledge in forest landscape ecology and the possibilities of using this knowledge to support management under changing conditions. We also discuss the forest sector's preparedness to deal with changes in management and how forest landscape ecology can guide this management. Forest landscape ecology has gathered substantial knowledge on patterns, processes, tools, and methods that can support forest and landscape management during changing scenarios. We recognize that existing knowledge is incomplete and that a substantial portion of our knowledge is uncertain, that variability in landscape conditions and various forms of error compound the problem, that we still lack considerable knowledge in some fields, and that there are likely to be knowledge gaps we are not aware of. We nonetheless face the challenge of responding to change based on the available knowledge.

10.1 The promising role of landscape ecology in dealing with change

As the authors of previous chapters have discussed, more than 30 years of forest landscape ecology research has led to the development of a body of essential knowledge, theory, research methods, and tools that have improved our understanding of

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forest landscapes and management of these landscapes. We know much more today than we knew even in the recent past about how forest landscapes are spatially structured, how their structure interacts with physical and biological processes, and how patterns and functions are affected by many drivers of change. In addition, we now have access to an extraordinary array of tools for collecting, analyzing, integrating, and drawing inferences from large spatial and temporal data sets. Both our existing knowledge and these new tools are improving our ability to plan and implement sound forest management practices and to prepare ourselves to face global changes. But are we ready yet?

10.2 Are we ready yet?

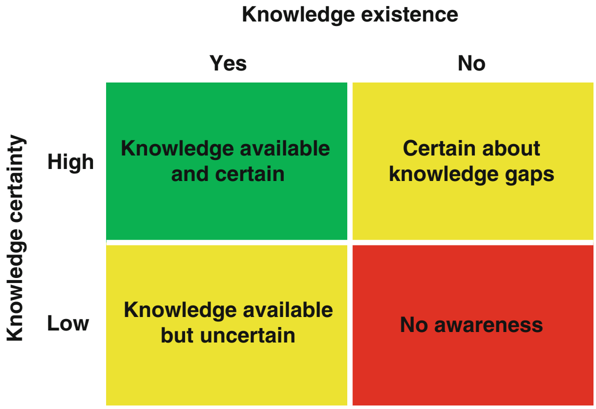
Population growth, climate change, land-use change, changes in management paradigms, and changes in management processes, among other proximate and ultimate drivers and processes of change, are creating increasing pressure on forest landscapes, which are already vulnerable or degraded in many parts of the world, thus creating additional stresses and threatening the provision of ecosystem services. Landscape ecology now has a much higher capability to inform management and decisionmaking in a context of change than ever before and can play a decisive role in mitigating or reversing ongoing degradation processes, thereby permitting sustainable or more sustainable provision of ecosystem services.

In spite of the enormous advances in landscape ecology, this field of research is still developing and maturing rapidly (Wu 2013), and the challenges facing this field of study are many. Questions such as “how much do we really know about change and its effects on landscapes?” or “how prepared are we to deal with such change in practice?” are not just legitimate; on the contrary, they are essential to ask, and the answers will define the future direction of landscape ecology and how we evaluate the role of this field from the perspective of practical applications.

10.2.1 How much do we really know?

The question of what we know and how well we know it is not just an epistemological question, in the sense of how much we are theoretically able to know about any topic, but rather is a pragmatic question whose answer constrains our ability to deal with real-world landscape change situations based on existing knowledge. Forest landscape ecology has advanced in many fields, thereby improving both the availability and the certainty of knowledge (Fig. 10.1, top left), but there are recognized knowledge gaps (Fig. 10.1, top right). Climate change and its effects on populations, ecosystems, and landscapes provide an increasingly important example. The process of climate change is not fully understood in terms of its causes, the underlying mechanisms, and the likely outcomes. In addition, research on the ecological

Figure 10.1 The balance between knowledge availability and certainty



effects of climate change has not yet provided sufficient information on basic physiological, biological, and ecological attributes of species that would let us address the impacts of climate change on biological diversity. Iverson et al. (2014) discuss this in Chap. 2 of this book. Since research in this field relies heavily on modeling, the uncertainty of the knowledge leads to high uncertainties in model predictions. Considering the large number of species and interactions in ecosystems and landscapes, gathering enough knowledge about these attributes seems difficult to accomplish within a reasonable timeframe, particularly considering that a large proportion of the known species are currently threatened and that many (perhaps most) existing species have not yet been identified.

Land-use change provides another good example. The available models can predict many of the effects of such changes on landscape patterns and on some landscape processes, but knowledge gaps are known to exist. For example, we do not understand the known interactions between forest composition and forest structure sufficiently well to account for these interactions in our assessments of the effects of landscape change on wildfires. Rego and Silva (2014) discuss this in Chap. 3 of this book. More importantly, we lack a full understanding of the complex feedback loops among the drivers of change and their effects. Farinaci et al. (2014) discuss this in Chap. 4 of this book. Furthermore, the lack of knowledge of carbon distribution, temporal changes in this distribution, and the underlying regulatory mechanisms for many ecosystem components limits our understanding of carbon cycles. Chen et al. (2014) discuss this in Chap. 6 of this book.

On the other hand, existing knowledge is seldom certain. Low certainty results from the fact that our knowledge frequently derives from research conducted at a particular temporal or spatial scale that prevents us from transferring those results to other scales. The knowledge may instead derive from particular landscape and experimental conditions that cannot be replicated or that differ from those in other landscapes or from the application of inappropriate analytical methods that produce misleading or uncertain conclusions. We are not sure, therefore, whether the knowledge gained from a particular setting will apply to a different one.

In addition, the complexity and natural variability of land systems make it very difficult to distinguish uncertainty in our knowledge from the uncertainty that is inherently associated with the behavior of complex systems. This may be more evident in modeling, in which variability of the system and uncertainty of model predictions are intermixed. The level of certainty of current knowledge is therefore often low (Fig. 10.1, bottom left). The fact that landscape ecology has not been able to produce scientific theories or laws that offer universal predictive power, like in many other fields of ecology, may not arise solely from our philosophical perspectives on ecological systems; rather, it may be at least in part due to the complexity and variability of the systems that we study and the lack of sufficient knowledge about how to apply our knowledge at a broader level, to different systems and scales.

The most striking knowledge challenge, however, is that we don't yet know what questions we have not yet identified and tried to answer (Fig. 10.1, bottom right). As science progresses and our knowledge grows, revealing what was previously unknown simultaneously creates the need for more knowledge to answer questions we had not formerly known existed, thereby revealing new gaps that become target areas for new research. These gaps are not known until a field evolves sufficiently to reveal their existence; therefore, they cannot be predicted. Although we don't currently know how much we don't know, it is reasonable to predict that there is, and will continue to be, unknown knowledge that may be critical for some future application.

10.2.2 Are we prepared to deal with change in practice?

Our preparation to deal with change in practice relies only in part on existing knowledge in landscape ecology and related scientific fields. It is mostly a function of the perceptions and willingness of society, as a whole, and particularly the economic and decisionmaking agents, to recognize change and the need to act in order to prevent or mitigate its negative consequences. In addition, we may be missing opportunities to harness the incredible energy of natural processes as a tool for coping with change. To answer the question about our preparation, we must consider landscape ecology as a scientific field separately from forest management at landscape and other levels, in the context of social and economic needs.

10.2.2.1 Forest landscape ecology

Although Wu and Hobbs (2002) identified "causes, processes, and consequences of land use and land cover change" as the second-most-important research topic in their "top 10 list for landscape ecology in the twenty first century", no other change-related issues were identified by the landscape ecology community at the turn of the century as particularly relevant for the near future. The term "climate change" was used in only 3.7 % of all papers published in all issues of the journal *Landscape*

Ecology in 2002, although its frequency of use had increased in recent years (Wu 2013). However, the top 10 research topics in the last decade as identified by Wu (2013) include several references to landscape change: land-use and land-cover change (ranked fifth), interactions between landscapes and climate change (ranked seventh), and ecosystem services in changing landscapes (ranked eighth).

Climate change has not been sufficiently addressed at the landscape level (Opdam et al. 2009), but change has been addressed frequently enough in the landscape ecology literature, whether directly or indirectly, through the analysis of change-related processes such as forest fragmentation or management, thereby providing relevant information that can be useful in an applied perspective under changing scenarios. Azevedo et al. (2014) discuss this in Chap. 1 of this book. Considerable limitations result from gaps in our knowledge and from areas of knowledge with low certainty, as noted earlier in this chapter, but knowledge gathered in recent decades can, at least in part, support management in terms of the design and implementation of prevention, adaptation, and restoration measures. Some of the syntheses presented in this book build a bridge between science and management to provide solutions that can be used in practical management to deal with change. See Chaps. 1 (Azevedo et al. 2014), 2 (Iverson et al. 2014), and 7 (Saura et al. 2014) of this book for details.

10.2.2.2 Forest landscape management

With the exception of climate change, all processes that are responsible for landscape change are driven by socioeconomic factors such as population growth or infrastructure development. Dealing with change in these cases mainly focuses on economics (both macro- and microeconomics), policy development, planning, and other fields that operate at scales above the landscape—often at global scales—and that focus on much more complex socioecological systems that combine aspects of human and natural systems.

The theoretical and technical foundations for management under ongoing and predicted change are available for the forest sector and other sectors that deal with forest landscapes in most parts of the world. However, there are clear limitations in our knowledge of forest management; for example, we currently lack sound silvicultural models that could be used to manage complex forests, particularly when it is necessary to meet multifunctionality requirements. Despite this, existing knowledge can support management of forest landscapes under changing conditions. For example, guidelines for forest management under climate change (e.g., Millar et al. 2007) are already available and have been applied in some parts of the world. Forest management philosophies have changed during the last decades of the twentieth century as a result of the introduction of systems analysis, consideration of multiple spatial and temporal scales, and concept of dynamics. By accounting for these new ideas, ecosystem management, sustainable forestry, and adaptive management are better suited to dealing with change and with its intrinsic uncertainty. See Chaps. 1 (Azevedo et al. 2014) and 9 (Coulson et al. 2014) of this book for more details.

In addition, the computational, logistics, and other tools that are currently available can be applied in managing forests that are being affected by processes of change, whether that change is physical, socioeconomic, or both simultaneously.

10.2.2.3 Barriers that arise from the interaction between science and society

Synthesizing these observations about the science and social contexts of landscape ecology reveals that, at the management level, preparation for change relies strongly on organizational or institutional culture, policy (national and local, public and private), planning, and knowledge transfer. The real degree and extent of the implementation of forest landscape management approaches that currently account for change is not fully known, since available examples of management that have been reported are usually restricted to the public sector in few areas of the world, and even in these cases, the information is sparse. Accounting for change is limited to a few cases, most of which are government-driven and in developed countries. Climate change in particular, although seen by the public and now governments as a major driver of change and a threat in many ways, has not significantly affected how forests and other land-use categories are managed. At the corporate and business management levels, the extent of plans to adapt management processes in response to climate change and other sources of change is unknown but is probably low.

At an institutional level, barriers exist that slow the incorporation of adaptation to change into management policies. This slowness results from several circumstances, including the following:

– Lack of awareness of change and its consequences
– Lack of management principles and methods that account for change and its effects
– Inertia, leading to an unwillingness to change how things are done in response to new challenges and processes
– Insufficient conceptual and technical preparation of individuals to deal with change
– Insufficient incentives from governments, markets, and others to account for change in planning
– Minimal pressure from the public

Some of these barriers are related to issues at a societal level, such as a lack of awareness and pressure from the public. Others are related to companies and government organizations that prevent or slow down the incorporation of change in their management activities. A particular group of barriers relates to insufficient development of an awareness of change, from scientific and management points of view, in academia, and, consequently, poor preparation of graduates to help institutions in areas that are being or will be affected by change, such as forestry.

10.3 What are the next steps?

From what we have discussed so far, limitations and barriers exist for both the sciences of landscape ecology and forest management and their practice at the landscape level. However, these obstacles also represent opportunities for landscape ecology and for society, and they are essential for helping us to define future directions for research and development.

10.3.1 Emerging fields and new directions in research and management

New fields within or related to landscape ecology that are under development will strongly benefit forest landscape ecology, particularly in terms of building up our knowledge and providing new tools to deal with change.

One of the fastest growing fields is landscape genetics. This field involves studying the interactions among landscape composition, configuration, and matrix quality in terms of evolutionary processes such as gene flow, genetic drift, and selection (Manel et al. 2003, Storfer et al. 2007). Spatially explicit data and spatial analysis tools are used to detect genetic patterns and to test their relationships with landscape patterns. The importance of the discipline, in a context of change, is very high. Many of the genetic patterns that have been analyzed using a landscape genetics approach resulted from changes in the landscape's structure, such as land-use change, forest fragmentation, intensification of forestry practices, and climate change. Changes in landscape structure therefore affect the genetic diversity patterns of populations and, often, the risk of extinction of these populations. Given the relevance of biodiversity in forest landscapes (see Chap. 7 of this book [Saura et al. 2014] for more details), landscape genetics will become a powerful approach for analyzing the effects of change processes on biodiversity (Manel and Holderegger 2013). Similarly, landscape genetics can provide knowledge to support management and conservation measures at landscape and regional levels to help prevent or minimize extinctions and to contribute to sustainable forest management.

Another emerging field that has grown extraordinarily is the study of ecosystem services. The ecosystem services concept and related methodologies can contribute powerfully to providing forest landscape ecology with many conceptual and methodological tools to analyze landscape change in terms of its impact on society and, through an analysis of trade-offs, to provide insights into how to optimize landscape structures and their management for the well-being of human communities. A great deal of ongoing research in landscape ecology relates to mapping the supply and demand for ecosystem services based on the landscape's composition, configuration, and processes. See Chaps. 1 (Azevedo et al. 2014), 5 (Marta-Pedroso et al. 2014), and 9 (Coulson et al. 2014) of this book for further discussion of this topic.

In addition to the ecosystem services approach, new directions in landscape ecology aim at the integration of socioeconomic factors in a broader landscape perspective. This is of utmost importance for the science of landscape ecology because change is often driven and carried out by the socioeconomic side of the human–nature system, because human societies are suffering from most of the consequences of change, and because solutions must be found on the socioeconomic side. Advances in multidisciplinary, interdisciplinary, and even transdisciplinary research are part of the required research agenda for the coming century to help us better integrate insights from the social and natural sciences within landscape ecology. This integration has been, at least in some parts of the world, a distinctive element of landscape ecology research. The promotion of interactions among scientists and with agents from fields outside the landscape ecology field of research, such as education, management, business, decisionmaking, and the public, is, therefore, a priority.

The incorporation of change in management and planning at a broader (landscape) scale should be an essential goal of forestry in the twenty-first century. Sustainable forestry has recently contributed to preventing or mitigating the negative effects of forest management on people, soils, water, wildlife, and landscape, thereby preventing degradation of forest landscapes in response to a growing demand for forest products in many parts of the world. Forest management can also anticipate changes by investing in species, rotations, harvesting technologies, and other management options to improve the ability of forestry to adapt to new biophysical, business, and market conditions, for example, and by improving efficiency and increasing innovation in the forestry sector. These are necessary directions for forest landscape management. On the other hand, the design and management of landscapes that will be resilient against climate change (Opdam et al. 2009) is another important goal of forest management and planning at a landscape level, particularly in terms of the effects of management on disturbance regimes and biological invasions.

10.3.2 Knowledge transfer

One aspect of forest landscape ecology that appears to have been overlooked by researchers is the transfer of knowledge to land managers and policymakers who practice landscape management. Although knowledge has been advanced steadily, energetically, and systematically by researchers, a noticeable gap has formed between the developers of knowledge and those who could apply that knowledge. This is a result of differences in educational backgrounds, focal scales, goals, and institutional cultures between landscape ecology researchers and forest managers (Turner et al. 2002). This state was recognized and brought to the attention of forest landscape ecologists almost a decade ago, with the goal of creating awareness and encouraging attempts to bridge the knowledge gap (Perera et al. 2006). Unfortunately, the topic of knowledge transfer has not gained much traction among researchers and remains a lower priority in formal discussion forums such as at scientific conferences and in publications.

However, the focus on knowledge transfer is even more relevant now, and its importance is likely to increase. As we explore the challenges to forest landscape ecology applications in a changing world, knowledge transfer will play a primary role. If a gap had formed between knowledge developers and practitioners who are consumers of that knowledge in the past, during a time when the context was less dynamic and more simple, imagine how this gap has widened in the present context of dynamic and complex changes, as has been discussed in the previous chapters of this book.

Here, we want to stress that forest landscape ecology researchers must actively engage in knowledge transfer, instead of passively expecting practitioners to seek out our knowledge. Many opportunities exist for us to do so. For example, we can aim to engage practitioners in a two-way dialogue from the outset of our research and to establish an ongoing feedback loop through practices such as adaptive management. We could reduce the time lag between detecting problems that affect practitioners and developing solutions through research by resorting to iterative options such as simulation modeling of scenarios. Fortunately, the task of transferring knowledge has become easier due to improved infrastructures: technological tools such as spatially explicit databases and analytical software and hardware, as well as skilled personnel who can use these tools, are now readily available to forest landscape managers.

There is another advantage of a dialogue between researchers and practitioners such as forest landscape managers: the benefit that researchers derive from the wisdom and experience of practitioners. This wealth of "expert knowledge", which is typically latent, can be now elicited and formulated quantitatively using advanced statistical techniques (Perera et al. 2012). Incorporating knowledge transfer as an essential component in forest landscape ecology research projects has an extra incentive: researchers are increasingly encouraged, and sometimes even required, to demonstrate the applications of their proposed research both to advance science and to advance the application of that science.

10.4 Summary

Forest landscape ecology has gone through a period of rapid development since the 1980s, leading to the development of a subfield of landscape ecology that deals with patterns, processes, and changes in forest landscapes and their close connection to forest management. Change has been part of landscape ecology from the beginning of the discipline, but its importance has recently grown due to increasing perception of new change processes, increasing and accelerating effects of processes that were already known, and interactions among different drivers and change processes, accompanied by a growing recognition of the state of degradation or vulnerability of forest landscapes around the world.

In this book, we have attempted to produce a synthesis of the most relevant topics within the study of changes in forest landscapes to provide readers with state-of-the-art information and to provide insights into how to apply the existing knowledge to prevent or mitigate problems related to change and to understand the limitations and challenges to the study of forest landscape change. Climate change is one of the relatively newly perceived drivers that is already affecting forest landscapes. However, its short- and long-term impacts on forest stands, on the landscape's composition, and on ecological processes are not yet fully understood, although we know they can significantly affect the distribution and functioning of these systems. Iverson et al. (2014) discuss this in Chap. 2 of this book. This and other drivers of change at stand and landscape scales are greatly affecting key processes, such as fire regimes, and are consequently affecting forest landscapes in most parts of the world. Rego and Silva (2014) discuss this in Chap. 3 of this book. Socioeconomic drivers of change are dominant factors around the world and operate at different scales and directions in different parts of the world. They are also affected by different drivers, such as climate change. Farinaci et al. (2014) discuss this in Chap. 4 of this book. Biodiversity, even more than other ecosystem and landscape components, has been affected by forest landscape changes of many different types and origins and potentially in irreversible ways in some parts of the world. Changes in the amount, quality, fragmentation, connectivity, and heterogeneity of forest habitats directly affect the forest ecosystem's ability to support populations and have significant implications for ecosystem resilience and the provision of a large array of ecosystem services. Saura et al. (2014) discuss this in Chap. 7 of this book.

Past, current, and future landscape changes can be described, analyzed, assessed, monitored, and modeled in diverse ways. The development of a relevant theoretical framework and set of methods for studying change is an important legacy of landscape ecology. Gómez-Sanz et al. (2014) discuss this in Chap. 8 of this book. A novel approach to evaluate change simultaneously from biophysical and socioeconomic perspectives is based on the ecosystem services concept. This has proven to have enormous potential for scientific use but also for decisionmaking in complex socioeconomic and ecological systems, in which economic considerations may be dominant. Marta-Pedroso et al. (2014) discuss this in Chap. 5 of this book. Among other services, carbon sequestration by forest landscapes is now widely recognized both by society and by the business community. The large amounts of carbon stored in forests and the vulnerability of this storage to forest management, as well as the complex dynamics that occur in forest systems and their effects on carbon cycling, make this a key issue in forest landscape ecology and other scientific fields. Chen et al. (2014) discuss this in Chap. 6 of this book.

The development of the topics discussed in this chapter and throughout this book provides valuable knowledge of potential applications in real-world management scenarios related to biodiversity conservation, carbon sequestration, fire management, evaluation of ecosystem services, and landscape monitoring. New directions in landscape ecology that are currently under development, such as landscape genetics and ecosystem services, can benefit forest landscape ecology by providing additional knowledge and tools to help us deal with change.

The available knowledge in forest landscape ecology related to change is possibly sufficient to support management under changing conditions, although identified and unidentified knowledge gaps exist. The preparedness of the forest sector to deal with change is currently insufficient. The incorporation of adaptation to change in business and forest management and planning should become a priority, and knowledge transfer is an essential but underused element in developing strategies to help organizations learn to deal with change.

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