EUROPEAN FOREST INSTITUTE

From Science to Policy 13

Forest Biodiversity in Europe

Bart Muys, Per Angelstam, Jürgen Bauhus, Laura Bouriaud, Hervé Jactel, Hojka Kraigher, Jörg Müller, Nathalie Pettorelli, Elisabeth Pötzelsberger, Eeva Primmer, Miroslav Svoboda, Bo Jellesmark Thorsen, Koenraad Van Meerbeek

Authors

Bart Muys, Division Forest, Nature and Landscape, KU Leuven, Leuven, Belgium

Per Angelstam, Swedish University of Agricultural Sciences, Uppsala, Sweden

Jürgen Bauhus, Institute of Forest Sciences, University of Freiburg, Freiburg, Germany

Laura Bouriaud, Universitatea Stefan cel Mare, Suceava, Romania

Hervé Jactel, INRAE, University of Bordeaux, Cestas, France

Hojka Kraigher, Slovenian Forestry Institute, Ljubljana, Slovenia

Jörg Müller, Julius Maximilians University Würzburg, Bavarian Forest National Park, Germany

Nathalie Pettorelli, Institute of Zoology, Zoological Society of London, UK

Elisabeth Pötzelsberger, European Forest Institute

Eeva Primmer, Finnish Environment Institute, Helsinki, Finland

Miroslav Svoboda, Faculty of Forestry and Wood Sciences, Czech University of Life Sciences, Prague, Czech Republic

Bo Jellesmark Thorsen, Department of Food and Resource Economics, University of Copenhagen, Copenhagen, Denmark

Koenraad Van Meerbeek, Division Forest, Nature and Landscape KU Leuven, Belgium; KU Leuven Plant Institute, KU Leuven, Leuven, Belgium

Acknowledgements

The report benefited from the helpful comments from external reviewers, Dr. Piero Visconti, International Institute for Applied Systems Analysis (IIASA) and Professor Dr. Bengt-Gunnar Jonsson, Mid Sweden University. We wish to express our thanks for their insights and comments that helped to improve the report, and acknowledge that they are in no way responsible for any remaining errors.

This work and publication has been financed by EFI's Multi-Donor Trust Fund for policy support, which is supported by the governments of Austria, Czech Republic, Finland, Germany, Ireland, Italy, Lithuania, Norway, Slovenia, Spain and Sweden.

ISSN 2343-1229 (print)
ISSN 2343-1237 (online)

ISBN 978-952-7426-20-3 (print)
ISBN 978-952-7426-21-0 (online)

Editor in chief: Helga Pülzl
Managing editor: Minna Korhonen
Layout: Grano Oy / Jouni Halonen
Illustrations 2,4-7: Gabriela Rueda
Cover photo: Goran / Adobestock

Recommended citation: Muys, B., Angelstam, P., Bauhus, J., Bouriaud, L., Jactel, H., Kraigher, H., Müller, J., Pettorelli, N., Pötzelsberger, E., Primmer, E., Svoboda, M., Thorsen, B.J., Van Meerbeek, K. 2022. Forest Biodiversity in Europe. From Science to Policy 13. European Forest Institute. https://doi.org/10.36333/fs13

Contents

Executive summary ... 4

1. Introduction ... 6
   - 1.1 Biodiversity as a natural capital supporting human well-being ... 6
   - 1.2 Global biodiversity crisis and European forests ... 7
   - 1.3 Aim of the report ... 8
2. Understanding forest biodiversity and its conservation ... 9
   - 2.1. What is biodiversity? ... 9
   - 2.2. Describing forest biodiversity ... 11
   - 2.3. Benchmarks for biodiversity conservation ... 13
   - 2.4. People's value of biodiversity and ecosystem services ... 17
3. Forest biodiversity monitoring: approaches and trends ... 19
   - 3.1 Forest biodiversity monitoring: from genes to ecosystems ... 21
   - 3.2 Overall trends in European forest biodiversity ... 24
   - 3.3 Data gaps and perspectives ... 25
4. Main threats to European forest biodiversity ... 27
   - 4.1. External threats ... 28
   - 4.2. Internal threats related to forest management ... 31
   - 4.3 Interaction between threats ... 34
5. Forest management promoting biodiversity ... 35
   - 5.1 The portfolio of measures for managing forest biodiversity ... 35
   - 5.2 Forest management systems supporting biodiversity ... 40
6. Forest biodiversity policies and incentives ... 45
   - 6.1 The role of public policy ... 45
   - 6.2. Public good characteristics and externalities justify policy ... 46
   - 6.3 Policy depends on context and history ... 47
   - 6.4 Public policy instruments, private initiatives and instrument mixes ... 48

7 Policy and practice implications ... 54

References ... 57

Executive summary

How can we effectively maintain and enhance forest biodiversity in Europe?

Forest biodiversity – what is at stake?

Forest biodiversity is more than just a mixture of species. It also concerns gene pools, structural and functional diversity as well as scale aspects that range from a single tree to entire regions. Forest ecosystems of Europe, which include both natural and managed forests, provide habitats for numerous plant and animal species and are havens for much of Europe's biodiversity. In addition to the cultural and aesthetic values of biodiversity, there is also clear evidence that biodiversity contributes to ecosystem productivity, stability and multifunctionality. Biodiversity loss is a major threat to ecological, social and economic stability.

Europe's forests have a long history of human use and management for the provision of wood, food, feed and fibre. The intensity and duration of these human impacts on biodiversity have varied across Europe both over time and by location. As a consequence, the types of forest biodiversity present and the interventions needed for its maintenance through conservation, management or restoration show a strong contextual dependency. In some areas, especially in Northern and Eastern Europe, there are still primary and old-growth forests where limited human interaction has left them in a substantially pristine state.

Forest biodiversity monitoring is key. Increased conservation and management of European forest biodiversity starts with effective monitoring. Most monitoring efforts are targeted at the species level, with very little attention given to monitoring other biodiversity aspects, such as genetic and functional diversity. In addition, the lack of coordination among European countries and institutions on what to monitor and how to monitor it has hampered efforts to build a standardised, EU-level overview of how forest ecosystems are responding to global environmental changes. Further support and investment is needed for the monitoring of biodiversity at different scales ranging from genes to ecosystems, the continued development of monitoring technologies and the harmonizing of indicators between European countries and institutions, as well as their long-term implementation.

Threats to forest biodiversity can be distinguished between 'external' threats that are beyond the direct influence of forest owners and managers (e.g. atmospheric pollution, climate change, biological invasions, land use change) and 'internal' threats that are directly related to forest management practices. Interaction between external and internal threats can increase the vulnerability of species. Different internal and external threats do not occur in isolation but interact, thereby increasing the overall vulnerability of individual species, ecosystems and even entire forests to each particular threat.

- **Forest biodiversity has suffered from atmospheric pollution.** It resulted in acidification, eutrophication and damage by tropospheric ozone, more than in any other continent.

- **Efforts to mitigate climate change continue to be both urgent and a challenge to preserve European forest biodiversity.** A combination of increasing temperatures, more frequent drought events, storms, megafires and pest outbreaks have all contributed to a sharp increase in tree mortality, threatening the survival of various associated species. Maladapted species that would normally migrate along latitudinal or altitudinal gradients are hampered from doing so by the extreme fragmentation of the European forest landscape.

- **Invasive species of plants, pests and diseases** that increasingly impact European forests can profoundly threaten biodiversity by dominating or devastating local ecosystems.

- In recent decades, **forested areas in Europe have expanded markedly through land abandonment and active reforestation**, particularly in Mediterranean and mountain areas previously used for marginal crop and livestock farming. Whilst initially appearing to be a positive, this can cause a loss of non-forest biodiversity associated with open ecosystems as light-demanding low vegetation is replaced by trees.

- **Different forest management practices have different impacts on biodiversity**. Compared to agriculture, forestry generally has an overall lower impact on biodiversity because of limited or no use of artificial fertilizers and biocides, and the much lower frequency of interventions causing ecosystem disturbance. Nonetheless, biodiversity in managed forests is often much lower than in natural forests, mainly due to the lack of diversity of tree species, of heterogeneity in tree age and of special habitats like deadwood or tree microhabitats. Smart management practices can further improve each of these aspects.

Forest (biodiversity) management – what should we do?

Current biodiversity is often related to legacies from the past, and in particular primary forests, old-growth forests, ancient forests and cultural forest landscapes are irreplaceable ecosystems that require **customised approaches to conservation**.

- **More stringent conservation measures are urgently required for forests that are home to species under threat, e.g. primary, old-growth and ancient forests**. The most effective way to secure biodiversity in primary and old-growth forests is to fully protect these forests. In ancient forests, nature-based management regarding the tree species composition and soil quality are often satisfactory conservation measures.

- While it is undisputed that forest management influences biodiversity, the **forestry sector has potential to include more biodiversity friendly measures** in its management portfolio. In fact, forest management is a key driving factor to restore, maintain and promote biodiversity in European forests.

- **Hands-on nature-positive management** is possible in every forest managed for wood production, including plantation forests. Doing so begins with increasing tree genetic diversity and tree species diversity. It can be further improved with decreasing the size of clearcut areas and the retention of more deadwood, trees bearing microhabitats like holes and fissures, and structural diversity in or between stands, all of which are within the means of every forest manager.

- **Ecosystem management, including variants of close-to-nature forest management**, needs to be promoted to embrace varied disturbances and support biodiversity reliant on the different development phases of the forest, including pioneer and decay stages. Modern integrated forest management seeks to mimic all types of disturbance regimes and the resulting development stages to allow the development and continued presence of patches of old and dead trees.

- **On a landscape scale,** a **Triad approach**, which is a combination of various management intensities, is a promising strategy to conserve a broad range of biodiversity while also achieving other important forest management objectives.

How should the policy landscape and finance respond?

Forest biodiversity is governed by a range of policy instruments, the form and use of which vary across Europe. The large variation in ecological contexts, as well as socio-economic, cultural and institutional settings that differ across forested areas in

Europe, needs to be factored in during policy formation:

- **Curbing biodiversity loss needs support from all sectors and actors at multiple levels**. Multisectorial collaboration is needed for efficient conservation and use of biodiversity in natural and cultural landscapes, combining agriculture, forestry, and human-cultural heritage.

- **A considerable time lag between new policies and their biodiversity responses** has to be taken into account given the slow pace of forest development and related management interventions.

- **The mix of tailor-made financial and other instruments needs to be expanded** to successfully secure Europe's exceptional biodiversity heritage. New instruments need to dovetail with existing governance frameworks and policy mixes at the global, EU and national levels to ensure that they are complementary rather than contradictory.

- **Market-driven instruments**, such as reverse auctions and biodiversity offsets are still underdeveloped and require further exploration and discussion.

Conserving and increasing biodiversity is a shared task that needs to be undertaken ambitiously by European forest managers and owners as well as public and private institutions, and the general public. It requires broad institutional and financial support.

1. Introduction

Chapter highlights

Biodiversity loss is a major threat to ecological, social and economic stability. Conserving and increasing biodiversity is a shared ambition of forest owners and managers, public authorities and the general public.

Europe's forests[1] represent a fascinating variety of ecosystems, from Mediterranean evergreen scrublands and lush, temperate, deciduous forests through to conifer-dominated boreal and mountain forests. This diversity is generated primarily by variations in climate, topography, soils and natural disturbances. In Europe, the majority of forests are, or at least have been, subject to some form of human modification or management over several millennia. A large variety of land tenure and governance systems, management styles and forest uses have transformed European forests by changing their extent, structure, species composition and function. Currently, forests and other wooded lands cover almost 40% of the continent, or 227 million hectares (Forest Europe 2020), and host a tremendous variety of living organisms that are of utmost importance for the conservation of biodiversity on the European continent. The knowledge, experience and range of tools available across the continent to forest managers shape forestry practices and are crucial to sustain forest biodiversity as a major natural capital.

1.1 Biodiversity as a natural capital supporting human well-being

In the last three decades, efforts have been made to place the importance of nature for human well-being higher on the international political agenda. The influential Millennium Ecosystem Assessment (MEA 2005) and The Economics of Biodiversity reports (TEEB 2010) have both drawn attention to a phenomenon known as ecosystem service cascade, where biological diversity is the basis for an array of cultural, regulating and provisioning ecosystem services which provide benefits and have value for humans, including in non-monetary terms, in addition to the intrinsic values of biodiversity. For example, biodiversity supports the provisioning service of wood production by providing suitable tree species and their genetic resources along with mycorrhizal fungi that boost the productivity of those trees. Such ideas are embedded in the Nature's Contributions to People (NCP) concept of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) supporting the implementation of the United Nations Convention on Biological Diversity (CBD) (IPBES 2015, Díaz et al. 2018). More recently, the Dasgupta review on the economics of biodiversity (Dasgupta 2021) emphasised the role of natural capital in our economies and made it clear that the species extinction crisis we face not only undermines the resilience of all life on the planet but has consequences for the human economic activity, which then impacts livelihoods and human well-being. Therefore, investing in biodiverse ecosystems is the basis for the sustainable development of a fossil-free circular bioeconomy (Palahí et al. 2020) while simultaneously providing insurance against abrupt environmental change (Paavola and Primmer 2019). The natural diversity of forest species, structures and functions is the only viable foundation upon which a sustainable flow of forest goods, services and values can be based. Indeed, the conservation of biodiversity could be viewed simply as best practice to retain the resilient, healthy and productive forest ecosystems that allow forests to better withstand the impacts of unprecedented climatic conditions and disturbances, such as increasingly severe storms, droughts, megafires as well as pest and disease outbreaks (Seidl et al. 2017).

1 Forests in the context of this report have no juridical meaning but are widely interpreted as ecosystems where trees or woody vegetation are dominant. We used the FAO definition (FAO 2000) that includes high forests and all of their developmental stages, but also scrublands and other wooded lands, excluding orchards and nut tree plantations.

1.2 Global biodiversity crisis and European forests

There is broad scientific consensus on the major global environmental threats caused by human activities. Even more than climate change, biodiversity loss is considered to have advanced to a more critically dangerous stage where the planet is no longer able to recover naturally (Steffen et al. 2015). The current loss of species is alarming as it is estimated that 1 million of the 8.7 million species on earth are threatened with extinction (IPBES 2019). More concretely, the latest IUCN global Red List classified 28% of all assessed species as threatened with extinction (40,084 out of 142,577), which means they are endangered, critically endangered or vulnerable (IUCN 2021). The major causes of this global biodiversity crisis are the 'evil quintet' of habitat loss, pollution, overexploitation, invasive alien species and climate change (Brook et al. 2008). The need for action to curb biodiversity decline has gained international profile since the United Nations (UN) Conference in Rio de Janeiro in 1992 where the UN Convention on Biological Diversity (CBD) was signed. One of the most significant concrete outcomes of the CBD was the formulation of the Aichi Biodiversity Targets in 2010 which were to be achieved by 2020. One of these targets was the development and implementation of the National Biodiversity Strategy and Action Plans (NBSAs). These have been backed up by important science-policy processes since the establishment of IPBES in 2012 as the latter has now become a source of reference information on the state of biodiversity highlighting for the countries as Parties of the CBD the urgency of undertaking more dedicated action in the post-2020 Global Biodiversity Framework (Diaz et al. 2020) as none of the Aichi targets were achieved and biodiversity continues to decline across all scales and regions (CBD 2020). Particularly for Europe and Central Asia, IPBES identified intensification in agriculture and forestry as the major cause of biodiversity loss (IPBES 2018).

To protect habitats and species while also meeting its international commitments, the European Union (EU) has been developing the Natura 2000 network of protected areas, based on the 1979 Birds Directive (on the preservation of wild birds) and the 1992 Habitats Directive (on the conservation of natural habitats). The main goal of the network is to offer a haven for the EU's most valuable and threatened species and habitats by setting conservation objectives, delineating protected sites and monitoring the conservation status of these sites. The network now includes almost 30,000 protected areas covering 18% of the EU's land area, of which 48% is forested, and 8% of its marine territory (EEA 2019). The Natura 2000 network is the cornerstone of the new EU Biodiversity Strategy for 2030 that aims, amongst other objectives, to securely protect and provide effective biodiversity management on 30% of the EU's land territory; to ensure the ecological restoration of forest and woodland areas in the EU by planting 3 billion biodiversity supporting trees and to make ecosystems healthier and more resilient to meet the challenge of climate change (EC 2020). Under the umbrella of the European Green Deal, other EU policies and strategies, such as the Farm to Fork Strategy, the Common Agricultural Policy (CAP), the EU Forest Strategy and the upcoming Nature Restoration Law set additional ambitious biodiversity targets. Biodiversity-related headline goals of the new Forest Strategy are the protection of primary and old-growth forests, the re- and afforestation of biodiverse forests as well as the promotion of closer-to-nature forest management practices (EC 2021, Larsen et al. 2022).

There is broad political consensus to support the increasing ambition of safeguarding and promoting forest biodiversity in Europe. However, stakeholder views on the data used to describe the current state and trend of European forest biodiversity, as well as on the strategies to protect and promote such diversity, strongly diverge. As to the data and views on the current state and trends, the indicators reported by the State of Europe's Forests, partly obtained from National Forest Inventories (NFIs), show an overall positive state or trend in an array of chosen biodiversity-related indicators, including the area of protected forests, amount of deadwood and stability of forest-bird populations (Forest Europe 2020). In contrast, the Habitats Directive related reporting streams show a grimmer situation with only 15% of European forest habitats in a favourable conservation status (EEA 2019). Scientists have also flagged the insufficient and inadequate protection of the last old-growth forests in Europe (Sabatini et al. 2020). This apparent inconsistency may illustrate different monitoring approaches supporting different purposes: the former provides systematic monitoring of national forest resources for sustainable forest production and multiple ecosystem services, while the latter monitors the conservation status of vulnerable species and forest types. As to the strategies for biodiversity conservation, there are also opposing views between proponents of land sparing (segregative) approaches targeting larger areas that exclude human activity to promote biodiversity within a matrix that focuses on provisioning services (Paul and Knoke 2015) and those supporting the land sharing (integrative) strategies using sustainable multifunctional forest management to conserve biodiversity using safeguards that still allow for activities such as the supply of wood (Kraus and Krumm 2013). Although these contrasting views are at the extremes of a broad spectrum of possible conservation options, they illustrate that retaining and increasing biodiversity in Europe is a complex, challenging and contentious issue that will require multiple nuanced solutions based on the specific social, ecological, economic and cultural contexts. Perspectives on biodiversity also vary along scales of space and time as well as being very much dependent on the viewpoint and values of the observer. This makes it difficult for decision-makers to get a good grip on the matter, its urgency and then find a viable way forward. This clearly causes a number of questions to arise, such as why positions on this issue are so divergent and how can biodiversity be effectively conserved and managed given this divide?

Finally, in addition to the differences in viewpoint, there are also large knowledge gaps and uncertainties, many of which are the focus of ongoing research. For example, scientists are yet to fully grasp the effects of climate change on forest biodiversity, the speed of the changes involved and the impact of such changes on the viability of the targeted habitat types when conserving, managing and restoring Natura 2000 areas. This understandably creates uncertainty regarding issues such as whether or not it is wise to introduce tree species and provenances from other regions to enhance the adaptive capacity of forests, even within protected areas? Or whether biomass extraction from forests for energy use is an opportunity to restore open or half-open habitats, or rather a threat to the last old-growth forests in Europe? This report will try to shed some light on these issues from a biodiversity perspective.

1.3 Aim of the report

This report **aims to explore the importance of biodiversity in the context of European forests and to make suggestions on how this biodiversity can be effectively maintained and enhanced through protection, management and restoration.** The term Europe in this document means European Union, except where mentioned otherwise. The report is meant for all kinds of decision-makers at the EU, national and local levels who are confronted with policy and management decisions related to biodiversity conservation and sustainable forest management. Although the primary focus is on the EU, most of the insights and recommendations made should be transferrable, with varying degrees of customisation where necessary, to non-EU countries as well. This report does not and cannot provide black and white guidelines on how to support forest biodiversity, rather it is designed to be a reference source for information and inspiration on the basics of forest biodiversity and forest biodiversity management. As such, it is a useful tool that highlights what is possible for evidence-based decision-making on this complex and dynamic matter. The report is purposely written and presented in a manner to stimulate dialogue on maintaining and restoring forest biodiversity while illuminating ways to bridge gaps between divergent viewpoints on the available options to avoid the loss of the irreplaceable and invaluable natural and cultural heritage inherent to European forest biodiversity.

2. Understanding forest biodiversity and its conservation

Chapter highlights

Forest biodiversity is more than just a mixture of species as it also concerns gene pools, structural and functional diversity as well as scale aspects that range from a single tree to landscapes and entire regions.

Current biodiversity is often related to legacies from the past. In particular primary forests, old-growth forests, ancient forests and cultural forest landscapes are irreplaceable ecosystems that require customised approaches to conservation.

Forest biodiversity is an indispensable underpinning factor for human prosperity and well-being. On the European continent, human societies have depended for millennia directly and indirectly on what forest and woodland ecosystems, which inherently includes their biodiversity, have provided. This involved deforestation to establish both settlements and clear land for agriculture on the most fertile soils. The remaining forest landscape was modified over time to a greater or lesser extent to provide multiple benefits. This included expansion of pastures for domestic animals, coppicing and pollarding of trees for feed and wood fuel as well as using or managing forests for building material and multiple non-wood forest products. Therefore, traditional knowledge, local stewardship and norms have fostered rules and regulations aimed at conserving the multiple benefits of forest landscapes over many centuries (Bürger-Arndt and Welzholz 2005). The advent of the industrial revolution ushered in an age where forests have been increasingly commercialised and exploited, a process which is ongoing and now expanding into the most remote forested areas, increasing the pressure on forest ecosystems. This is arguably most evident in forest management regimes focused on wood as a provisioning ecosystem service and often leads to a simplification of forest ecosystems into cropping systems. Only relatively unproductive, remote or inaccessible areas have retained forest ecosystems dominated by natural dynamics. All these transformations that have taken place at varying intensities over time and space have profoundly modified the composition, structure and functioning of European forests.

2.1. What is biodiversity?

The term biodiversity is a short form of **biological diversity** and was coined for the first time at a conference organised by the National Academy of Sciences of the USA in 1986. The report from this meeting explains the emergence of the biodiversity concept as being triggered by data on deforestation and species extinctions in the tropics and that forest loss accompanied by short-term profits were followed by local economic decline (Wilson 1988). There was a conscious effort at the time, as there still is, to (1) place evidence-based knowledge about the loss of life forms on the table of decision-makers, and (2) to discuss what actions and levers are required. Being a foundation for economy and society, the biodiversity concept needs to be well understood by citizens, stakeholders, actors and policy-makers.

Box 1. Definition of biodiversity

According to the UN Convention on Biological Diversity, biodiversity means "*the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems*". Biodiversity is thus a broad concept that is hierarchical and scale-dependent. It ranges from the springtails and earthworms on a decomposing tree trunk in an old-growth forest to the Gaia theory in earth systems science, seeing planet Earth as a single complex system (Lovelock and Margulis 1974).

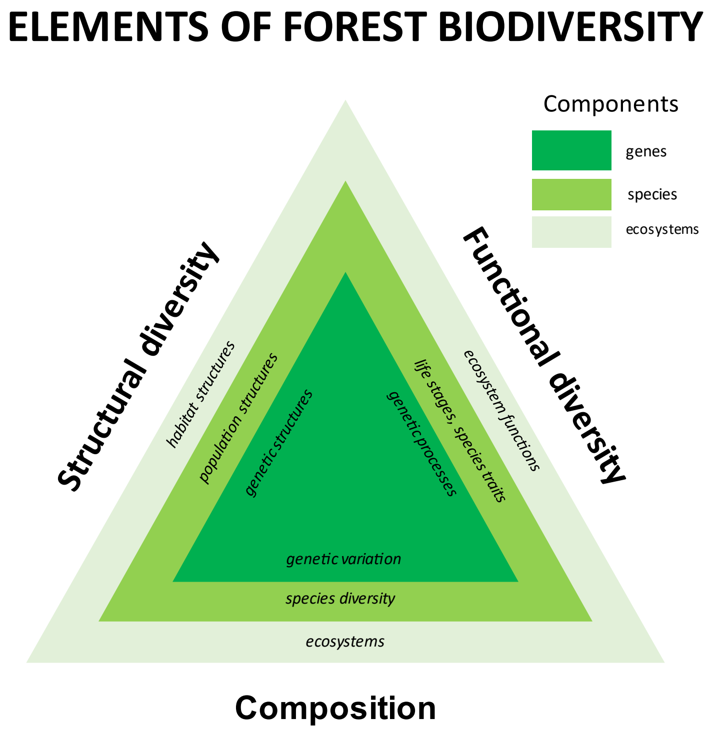


Figure 1. The main elements of forest biodiversity are represented as a triangle with three dimensions (composition, structure and function) that take account of the three hierarchical levels of components (genes, species and ecosystems). Modified after Noss (1990).

From the definition in Box 1, we can see that biodiversity has different **components** that correspond to the different levels of organisation of biodiversity, namely **genes, species and ecosystems**. Genes are the building blocks of life that explain the unique differences between species, individuals and populations within a species. Ecosystems are geographical areas, small or big, where different species live together and interact with each other and their environment. To make biodiversity understandable, these components can be looked at from three different **dimensions,** namely its compositional, structural and functional diversity (Figure 1). **Compositional diversity** looks at the occurrence and variation in genes, species or ecosystems by, for example, assessing bird species diversity in an area (alpha diversity), the differences in species composition between two separate locations (beta diversity), or the total species diversity in a landscape (gamma diversity). **Structural diversity** is concerned with the variation in patterns present in genes, species and ecosystems, for example, the number of vegetation layers or microhabitats (e.g., standing and lying deadwood, cavities of different sizes) present in a forest (Larrieu et al. 2014). Finally, **functional diversity** relates to the variations in processes active in genes, species and ecosystems (Franklin 1988, Noss 1990, Larsson et al. 2001, Turner 1989, Bracy Knight et al. 2020) by, for example, considering all life stages of a tree species from seed to an overmature tree or all functions in a complex food web including primary producers, herbivores, carnivores and detritivores.

Biodiversity can thus be seen as introducing a **language** for describing the **composition, structure and function** of ecosystems that are designed to help sustain the biosphere for its own sake and to allow humans to derive long-term benefits. Applying this language metaphor, biodiversity has *letters* (such as individual genes, species, ecosystems), which represent the composition of life. These different letters need to be structured into correctly *spelt* words (such as a viable population structure, a natural forest structure, representative habitat networks). Finally, there is the *grammar,* the process of arranging words and clauses in proper order to give them functional meaning (such as the presence of species with very different traits, which are characteristics ensuring different functions in the ecosystem) (see Figure 1). In short, forest biodiversity involves much more than just trees.

2.2. Describing forest biodiversity

In the following paragraphs, we will go into some further detail to better describe the dimensions of forest biodiversity and highlight some of these dimensions' key components. Following on from this, we will highlight the importance of spatial scale for studying and managing forest biodiversity.

Composition

Within an individual living organism in a forest, its genes are distributed in chromosomes and throughout its genome. Within a population, which can be defined as a group of individuals of a given species living in the same area, one can observe differences between individuals explained by **genetic variation**. Through reproduction processes, genes can be re-combined and modified. In pioneer tree species, such as the birch with light propagules (seeds, spores), the genetic exchange is strong, whereas in late-successional species, such as oaks that have heavy seeds, this exchange is lower. Among animal species, the degree of genetic mobility varies dramatically. In fragmented landscapes, populations of a given species can be so small and isolated that they suffer a loss of genetic variation, which leads to lower individual fitness and a higher risk of species extinction. Knowledge of the genetic variation of species forms the basis for *in situ* and *ex situ* conservation and, in the case of tree species, for tree breeding and climate change adaptation.

Species diversity in European forests is still enormous. Much attention is given to trees, herbs, birds and mammals, however, a much broader focus is required to truly appreciate the variation in life forms. Belowground biodiversity is largely hidden and therefore often overlooked, nevertheless, it includes innumerable plant roots, microbes, fungi, earthworms, nematodes, arthropods, etc. By way of a specific example, soil fungi can be either mycorrhizal (forming beneficial associations with plant roots), endophyte (living in the plant roots without harming them), parasitic (causing plant disease) or saprotrophic (thriving on dead organic material), all of which play their own crucial role in forest ecosystems. Mycorrhizal fungi link the whole belowground part of the ecosystem in a network that provides nutrients, energy and information exchange. At the stand scale, mycorrhizal species diversity can reach 20–120 species depending on the forest type (Schirkonyer et al. 2013, Primicia et al. 2016) and at the regional scale, this can rise to 350 species (Alday et al. 2017). Furthermore, tree species do not always associate with a single mycorrhizal species, changing their association as they transit through life stages (seedling, sapling, mature; Boeraeve et al. 2021).

The **diversity of forest ecosystems** is also important. Within the large biomes of evergreen forests in the Mediterranean, broadleaved forests in European lowlands to mixed conifer forests in mountainous and boreal regions, forest ecosystems further differ based on geology, soils and site types. However, ecosystems related to forests can also be a microcosm, as is the case with aquatic ecosystems that occur in **dendrotelms**, cup-shaped hollows in trees permanently waterfilled and containing specific fauna (Bütler et al. 2020).

Structural diversity

The range of habitats suitable for different species found in forests primarily depends on the abiotic environment (climate, geology, soil, hydrology and land form). In addition, organisms further modify this abiotic structure by adding **biotic structures** (e.g., vegetation layers with trees in different development stages, shrubs, herbs and mosses). Some organisms, called **ecosystem engineers**, produce structures that, because of their dimension, longevity or number, provide a habitat for many other organisms. In forests, trees are obviously the ecosystem engineers par excellence, however, other organisms, such as beavers, large herbivores and earthworms, are also able to drastically change their environment and consequently, create or modify the habitat for other organisms.

As a plant community dominated by trees, a forest may be either even-aged and single-layered, or uneven-aged and multi-layered. In a natural forest with a long absence of human interference, biotic structures are typically very heterogeneous and emerge at different spatial scales. These structures include trees of

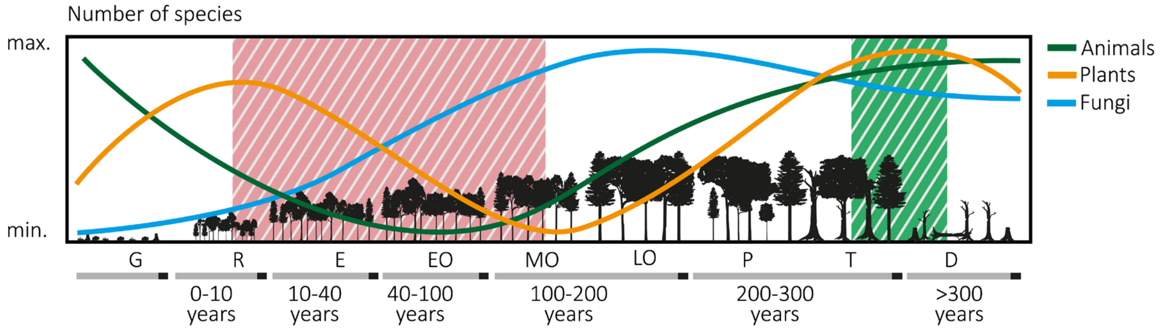


Figure 2. Forest stand development following a stand-replacing disturbance in European forests, with the hypothesised response of plants, animals and fungi (after Hilmers et al. 2018). Bands at the bottom of the graph represent an approximate timeline of the successional stages (in years after the disturbance). Stages: G, gap; R, regeneration; E, establishment; EO, early optimum; MO, mid-optimum; LO, late optimum; P, plenter; T, terminal; D, decay. The green hatched area identifies the development stage with the highest expected species diversity, while the red hatched area identifies the developmental stages wherein commercial forestry operates, demonstrating the mismatch between management for timber and biodiversity.

many size classes as well as standing and lying **deadwood** in different stages of decay, all of which have high importance for biodiversity (Stokland et al. 2012). They typically show complete cycles of **successional stages**, from pioneer stages with young trees to late-successional stages (Figure 2) with **habitat trees.** Habitat trees (also called veteran trees) are large and old trees that provide **microhabitats** (Figure 6). Microhabitats in the forms of hollows, crevices, cracks, decaying residues and so forth are key elements for biodiversity conservation in forests (Bütler et al. 2013; Larrieu and Cabanettes 2012; Larrieu et al. 2018). Similar to deadwood, knowledge about habitat trees and microhabitats and their importance for biodiversity has increased significantly (Fritz and Heilmann-Clausen 2010, Müller et al. 2014, Gossner et al. 2016, Asbeck et al. 2021, Courbaud et al. 2022).

The crucial driver shaping biotic structures in primary forests are disturbances. **Natural disturbances**, such as fire, wind, drought, flooding and insect outbreaks, create dynamic and heterogeneous habitat mosaics. The disturbances' extent, severity and frequency strongly affect the proportion of young, mature and old-growth stages in a landscape. This has direct bearings on species occurrences because certain species are more dependent on later-development stages than others which prefer disturbed or young forests (Figure 2). There is a continuum of disturbances in forests ranging from the occasional death of individual canopy trees caused by pathogens through to widespread low-severity mortality caused by drought or herbivory, up to rare high-severity stand-replacing fires, mass blowdowns, or major insect outbreaks. In forests managed for wood production, these disturbances are typically managed interventions and, as a consequence, all the structures become much more simplified in such forests (Figure 2).

Functional diversity

In European forests clear **positive biodiversity effects on forest productivity and forest stability** have been observed in dedicated tree diversity experiments (e.g., Van de Peer et al. 2018), observational studies in natural and managed forests (e.g., Jucker et al. 2014, Pretzsch and Schütze 2009) and large forest monitoring networks (e.g., Gamfeldt et al. 2013, Sousa-Silva et al. 2018). Mixtures of tree species show positive interactions, such as complementarity in water uptake and a reduction in infestations of particular pests and pathogens (Jactel et al. 2017). Much of the functionality of diverse forests depends on the identity of the species that compose it and the functional characteristics that are their **species-specific traits** (Ruiz-Benito et al. 2017), a factor that links back to their genetic makeup. In addition, diverse forests show higher levels of **multifunctionality** than monocultures (Van der Plas et al. 2016).

Forest biodiversity along scales

Biodiversity is scale-dependent and can be observed at different temporal and spatial scales (Brumelis et al. 2011), ranging spatially from the micro- to the continental scale and temporally from daily variations through to long-term epochal changes. While there are numerous possible spatial scales one can consider, for forest managers, the **single-tree**, **patch**, **stand** and **landscape scales** are particularly relevant. In the following paragraphs, we provide some examples of the three dimensions of composition, structure and function of biodiversity at these different scales.

Composition at the *single-tree scale* can be used to study a diversity of species with small area requirements (e.g., vascular plants, lichens, fungi in deadwood). At the *stand scale,* the genetic diversity of a tree population can be assessed along with the occurrence of species with intermediate area requirement (e.g., songbirds, insects, mycorrhizal networks). At the *landscape scale,* researchers can examine the genetic diversity of metapopulations (defined as a regional group of interconnected populations) of a tree species and can monitor species with large area requirements (e.g., raptor birds, large mammals).

Structure at the *single-tree scale* typically entails cavities and other tree-related microhabitats. At the *stand scale,* topographic and soil variation can be effectively researched, as can the horizontal and vertical tree canopy structure, **deadwood** profile (amounts by size and decay stages) and the amount and size of different forest habitats and forest succession stages. The landscape scale is relevant for assessment and planning of habitat network functionality, and when determining the occurrence of land-use legacies, such as cultural landscapes and ancient forests.

Function at the *single-tree scale* can provide insight into organisms with a broad range of functionality, such as primary producers, herbivores, carnivores, detritivores and parasites. The *stand scale* allows for research into **niche complementarity** and **portfolio effects** thanks to the mixture of tree species. Finally, the *landscape scale* is used to determine the functional significance of disturbances such as windstorms, fires and bark beetle gradations.

2.3. Benchmarks for biodiversity conservation

Two complementary visions of conservation management

Visions on conservation are subject to cultural contexts and may evolve over time (Mace 2014). In broad terms, European biodiversity conservation involves the maintenance of functional habitat networks at landscape scale that predominantly represent visions of either **forest naturalness** (Peterken 1996, Fritz et al. 2008) or **cultural forest landscapes** (Agnoletti and Emanueli 2016).

Forest naturalness is a term used to indicate the degree to which the attributes and dynamics of a forest are derived from purely non-anthropogenic factors (Peterken 1996, Kennedy et al. 2019). It is used as a concept to measure the bio-ecological integrity of an area and as such, naturalness indicators are being developed for different European forest biomes (Brumelis et al. 2011, Winter et al. 2012) and efforts are being made to map areas with high degree of naturalness as a basis for conservation efforts (Sabatini et al. 2021). An important aspect of forest naturalness is **forest continuity** (Nordén et al. 2014), a term that refers to the continuity of forest cover over larger areas in space and time. For forest continuity *in space* (also called **forest contiguity**) it is of utmost importance that coverage of the large variability in abiotic conditions, such as local climate, soils, hydrology and topographic conditions, which provide heterogeneous mosaics of different natural forest types, is ensured. Forest continuity *in time* ensures the occurrence of forest dynamic processes, including succession and biotic or abiotic disturbances, in an ever-shifting mosaic. Forest naturalness is a key factor in defining those areas assigned the highest conservation priority in Europe, these often being primary forests, old-growth forests and ancient forests (Box 2). In primary forests, all the biodiversity normally inherent to old-growth and ancient forests can be expected, however, old-growth forests have biodiversity features that do not necessarily occur in ancient forests and vice versa.

Box 2. Forests with a high degree of naturalness.

Primary forests are large naturally dynamic forest areas where forest continuity has been maintained over long periods and where human influence has been absent or minimal during that time.

Old-growth forests are generally smaller areas of forest with a long absence of human interventions. They may have been harvested, managed or even completely deforested by humans in the past, but have now recovered to the point where all the successional stages are present, including old trees and large amounts of deadwood.

Ancient forests are generally smaller forest fragments that have existed as far back as historical sources can determine and, therefore, possess a more or less undisturbed forest soil and forest microclimate. They often have some level of management and wood harvesting, and may therefore lack late-successional stages and old trees (cf. Hermy et al. 1999).

The definitions used here have been formulated in a way to clarify the differences between these categories of forests with high conservation value. They may deviate from definitions used in recent European policy documents, where confusion continues to exist.

Cultural forest landscapes are all kinds of agro-silvo-pastoral systems that may contain coppice forests, open woodlands, pasturelands with scattered habitat trees, scrublands and the like. Given how widespread human-altered landscapes are in Europe, these areas are crucial for biodiversity conservation in many parts of Europe. As an example, on the Iberian Peninsula, native oak woodlands are common and are subject to selective fuelwood and cork harvesting, cereal production, foraging by pigs, sheep and cows, and hunting (Campos et al. 2013). Although such cultural forest landscapes are human-made and retain limited tree cover due to cyclic human interventions, they still host a substantial part of Europe's native biodiversity. Using the light index from the plant bio-indicator table of Pignatti et al. (2005), it can be shown that 80% of total Italian native flora needs low or zero tree cover to survive while these plants in their turn provide suitable habitat to even more host-specific species of arthropods, fungi and so forth. This is in line with the hypothesis that a substantial part of Europe's current biodiversity is a result of co-evolution in low tree cover landscapes (Vera 2000). Open landscapes in Europe have existed for a long time thanks to native megafauna that was driven to extinction by human overhunting approximately 30,000 years ago (Araújo et al. 2017). Cultural landscapes kept open by human management form a kind of surrogate for these lost ecosystems (Sandom et al. 2014). In recent decades, cultural landscapes have been rapidly changing, sometimes due to agricultural intensification (Godinho et al. 2014) but primarily due to the rural exodus, abandonment of traditional practices and spontaneous forestation (Mauerhofer et al. 2018, Varela et al. 2020). Especially in southern Europe, contiguous pioneer forests and related hot fires represent an ongoing threat to local biodiversity, a situation further complicated by the emerging trade-off between ecosystem restoration designed to facilitate carbon sequestration and biodiversity conservation (Burrascano et al. 2016).

Depending on the conservation vision being considered, restoration and maintenance of natural processes, or low-intensity management practices is necessary (Halada et al. 2011). Both conservation visions require sufficient areas with appropriate quality, size and connectivity representing the range of site conditions and development stages of a particular region. One vision is not superior to the other, but they are complementary. Rather than being based on cultural preferences, adoption of either option should be

LANDSCAPE AND HABITAT FEATURES SUPPORTING EUROPEAN FOREST BIODIVERSITY

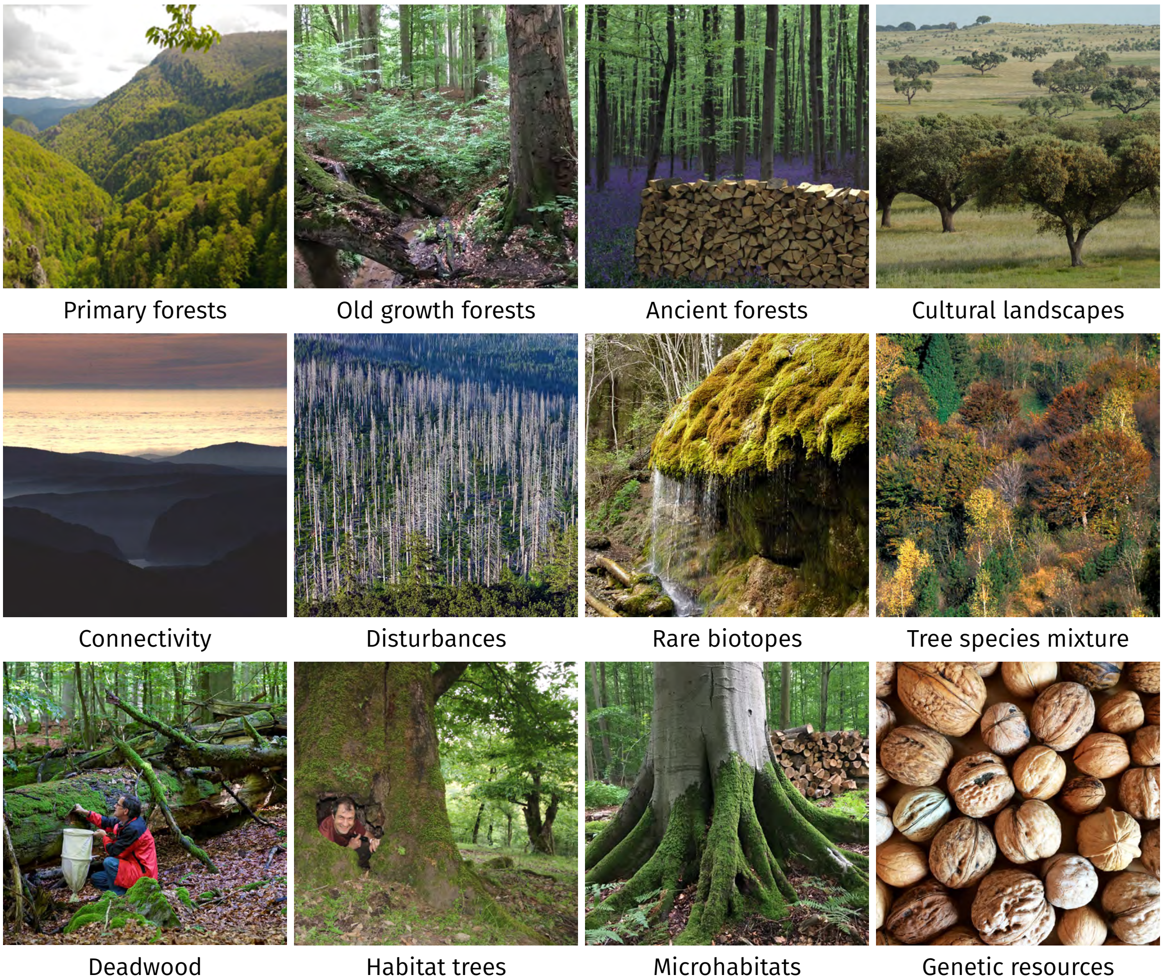


Figure 3. A visual overview of the major landscape and habitat features supporting forest biodiversity in Europe that need to be taken into consideration by forest management practices. Photos: Miroslav Svoboda (1), Bart Muys (2, 3, 5, 7, 8, 11), Per Angelstam (4), Ingo Arndt (6, 9), Sven Finnberg (10), Michele Bozzano (12).

based on the past land-use history and associated conservation priorities of the area under consideration (Angelstam et al. 2021; Van Meerbeek et al. 2019). Thus, a forest landscape brought into management for the first time in a recent past has the advantage of conservation practices targeting the conservation or restoration of naturalness, while the biodiversity of forests and woodlands with a long stable management history often have advantage of promoting the continuation of the cultural landscape management in place. This unavoidably results in the need for a complex conservation strategy where wood harvesting in some areas needs to be decreased or banned, while in other areas it can, or indeed, may need to be intensified. Particularly in the context of cultural landscapes, biodiversity conservation can create win-win scenarios with biomass provision for bioenergy or biorefinery in the framework of the circular bioeconomy. As a synthesis of such an integrated vision on European forest biodiversity, Figure 3 shows an overview of the most important habitat and landscape structures that support biodiversity in European forests; all of which need to be harnessed by exploiting their potential and addressing what threatens them. Forest management for biodiversity (including non-intervention) will need to be moulded around all these elements of biodiversity in a spatially optimised way (see Chapter 5).

Levels of ambition for biodiversity conservation

One can formulate at least three different levels of ambition for the conservation of biodiversity (Angelstam et al. 2004). The first level is to ensure the simple **presence** of individuals of a species in a remnant patch of a once widespread habitat. A second, higher level of ambition is to ensure the successful conservation of naturally occurring species in **minimum viable populations** over long periods (Reed et al. 2003). Because ecosystems are dynamic, the total area needed to ensure the persistence of each species in a viable population is higher in the long term than in the short term (Pickett and Thompson 1978). Consequently, the third and highest level of ambition is to ensure **ecosystem integrity** (Karr et al. 2021; Pimentel et al. 2000) as this allows self-sustaining interactions among species and processes (Bengtsson et al. 2003). This supports ecological resilience (Gunderson 2000), an area of increasing importance given the current environmental challenges the planet faces. This third level of ambition is included in more holistic concepts that try to capture the key ecological attributes meant to provide a framework describing the conditions required to create and preserve functioning ecosystems and their components (Schick et al. 2019). These different levels of ambition apply both to the visions of natural forests and cultural woodlands (Angelstam et al. 2021). Thus, when interpreting and debating what status and trends data points to regarding biodiversity, the level of ambition also needs to be specified.

Critical habitat degradation thresholds

The term **biotope** refers to an area of environmentally uniform land cover supporting the occurrence of a particular species. A **habitat** is defined as an area providing the requirements, such as feeding, cover and breeding, of a particular population of a species and often includes several biotopes (Udvardy 1959). Therefore, there is a need to identify and assess the quality, size and spatial distribution of biotopes that form habitats. However, habitats are more than just a combination of biotopes or types of land cover as predators, competitors as well as micro- and macroclimates also affect the functions of biotopes. A species population's ability to persist in a landscape or region depends on how much habitat there is, whether individuals or seeds can move between different patches of suitable habitat and how the habitat network is maintained over time. Land-use change is a major driver of habitat degradation (Fahrig 2013, Hanski 2011) as it is a significant contributor to area loss and fragmentation of habitats.

Habitat loss decreases the number of individuals that can be supported in that habitat. It is a major driver of species loss and there is both theoretical and empirical evidence for the existence of thresholds for local extinctions of a population as the amount of available habitat is reduced (Andrén 1994; Bender et al. 1998; Fahrig 2003). Thresholds refer to the fact that the risk for population extinction shifts from low to high within a limited range of further loss of habitat (Guénette and Villard 2004). This means that there are limits to how much of different forest habitats may disappear without threatening the viability of populations of all naturally occurring species. The existence of such thresholds forms the basis of the formulation of long-term evidence-based performance targets for how much of different forest habitats are needed (Angelstam et al. 2004; Svancara 2005; Tear et al. 2005).

Fragmentation increases the isolation of remaining areas of habitat, making it harder for individuals to move within the landscape and is a common reason why species disappear locally, regionally and, eventually, completely (Andrén 1994; Fahrig 2003). There are two types of key thresholds, or tipping points. The first key threshold is referred to as the **percolation threshold,** this occurs when contiguous habitat is broken up into patches, thus no longer permitting a flow of individuals or propagules of different species through an unfragmented habitat (With and Crist 1995; Bascompte and Soulé 1996). The second key threshold is when inter-patch distance increases, leading to **patch isolation**, which may result in unsustainable levels of inbreeding depression and loss of fitness within the isolated populations.

Habitat degradation thresholds have three important consequences for the conservation of species:

1. Given the vagaries of the natural world, even if a habitat is ideal, the occupancy of any given species is always less than 100%.

2. For species that can live in landscapes with fragmented habitats, the effects on the persistence of species are often non-linear. This means that when 10–30% of their habitat remains, the probability of persistence of the species drops rapidly. This is the evidence-based background for the CBD's and EU's targets of 10–30% protected areas (Dinerstein et al. 2019).

3. There may be considerable time lag between habitat loss and species loss. This applies in particular to long-lived species such as certain trees and is termed **extinction debt** (Hanski 2011).

Conservation of habitats forms the basis for the **EU Habitats Directive** and related **Natura 2000 network**. In fragmented landscapes, which is the context for many European forests, spatial planning toward representative ecological networks encompassing the integration of protection, management and landscape restoration (Mansourian et al. 2020) is needed. In turn, this requires regionally adapted approaches to collaborative landscape stewardship (Muñoz-Rojas et al. 2015). This argumentation is well reflected in the CBD's Aichi target 11 (CBD 2010) and the current first-order draft targets for effective and representative protected area networks (CBD 2021), and also in European policy about **Green Infrastructure** and biodiversity (EC 2020, 2021), which builds on established principles in landscape ecology and conservation biology for sustaining functional habitat networks (Opdam et al. 2001, Liquete et al. 2015).

2.4. People's value of biodiversity and ecosystem services

Humans gain significant benefits from biodiversity in natural ecosystems and cultural landscapes. The concept of ecosystem services captures this idea and structures value pathways between biodiversity conservation as a whole and benefits for humans into three overall categories. The first and most obvious category is **provisioning services**, which include directly marketed products such as timber and biomass but also includes many other provisioning services directly and indirectly derived from resource use. The second category is **regulating services**, such as the pollination of plants by insects, carbon sequestration and erosion control within ecosystems. The third category is **cultural services**, which include activities and values associated with recreation, aesthetics and so forth. In older versions of the ecosystem services framework, biodiversity and its ecosystems functions, such as photosynthesis, soil formation and the like, are sometimes considered as a fourth category of supporting services (MEA 2005).

While some species, structures and functions clearly underpin several of these ecosystem services, based on current knowledge, it is not obvious what role, if any, all the organisms that constitute Europe's forest biodiversity have to play in these services or if they simply exist for their own sake. Furthermore, across these categories, the benefits and experiences derived by human individuals and groups are quite different (Primmer et al. 2018). The subjective nature of human values and interests poses challenges if we want to measure and assess the sustainability of the benefits of biodiversity conservation and the costs of losing species, habitat structures and ecosystem functions.

Economists have long recognised that markets' fixation on financial interests threaten socially-optimal management regimes, especially when it comes to natural resources that are unique, finite or technically irreplaceable or where the true value derived from a resource cannot be readily realised by market activities (Krutilla 1967, Arrow and Fisher 1974, Spash 2021). They have also pointed out that not all values are easily measured despite the fact that over the past few decades numerous aspects of the economic **value of biodiversity** have been addressed (Helm and Hepburn 2014). While values associated with provisioning services, such as timber and hunting, that are derived from certain species and ecosystems and are traded on markets are fairly easy to calculate, at least in monetary terms, many other values are much more challenging (Hanley and Perrings 2019). For example, how does one quantify and qualify the value to home owners of landscapes with forest views, and what is the value to people making use of natural areas for recreation or health purposes. Even more confounding are **non-use values**, where a landscape may be held in high regard and/or enjoyed by people who do not undertake any particular behaviour or perceivable interaction with that landscape. These notably include what is called the **existence value**, that is the value people derive from knowing that a species exists or a naturally dynamic ecosystem is protected in ways that guarantee its long-term existence, which is somewhat similar to how unique cultural heritage and objects are perceived. Closely related to this is the **bequest value**, the value of knowing that species, ecosystems and biodiversity is preserved for future generations, as well as option values, the value of having the option to enjoy any of the other values at some point in the future. While much of the above is certainly difficult to quantify and qualify, it is not always impossible, and as explained by Hanley and Perrings (2019), methods have been developed and adapted to address these values and are increasingly accepted as a basis for policy evaluation.

Building on these concepts of value is the **natural capital approach**, which is designed to keep track of the various ecosystems' and resource stocks' ability to provide flows and hence retain their value. The realisation that natural dynamics and biodiversity underpin these values for humans is reflected in **nature-based solutions**. This is an approach that entails protecting and restoring nature to counter the challenges posed by environmental change. Each of these approaches has its own merits, however, putting all of them into practice to safeguard values requires both progress in monitoring, verification and reporting on the impacts of actions, as well as in the careful consideration of what policy instruments should be used to guide the choice of suitable actions.

3. Forest biodiversity monitoring: approaches and trends

Chapter highlights

Monitoring of biodiversity at different scales ranging from genes to ecosystems, continued development of novel monitoring technologies and harmonizing of used indicators between European countries and institutions, as well as their long-term implementation, all need further support and investment.

National Forest Inventories (NFI) are a great information source for biodiversity monitoring of managed and unmanaged forests, and enriching their sampling procedures with non-tree biodiversity information adds value to existing data sources.

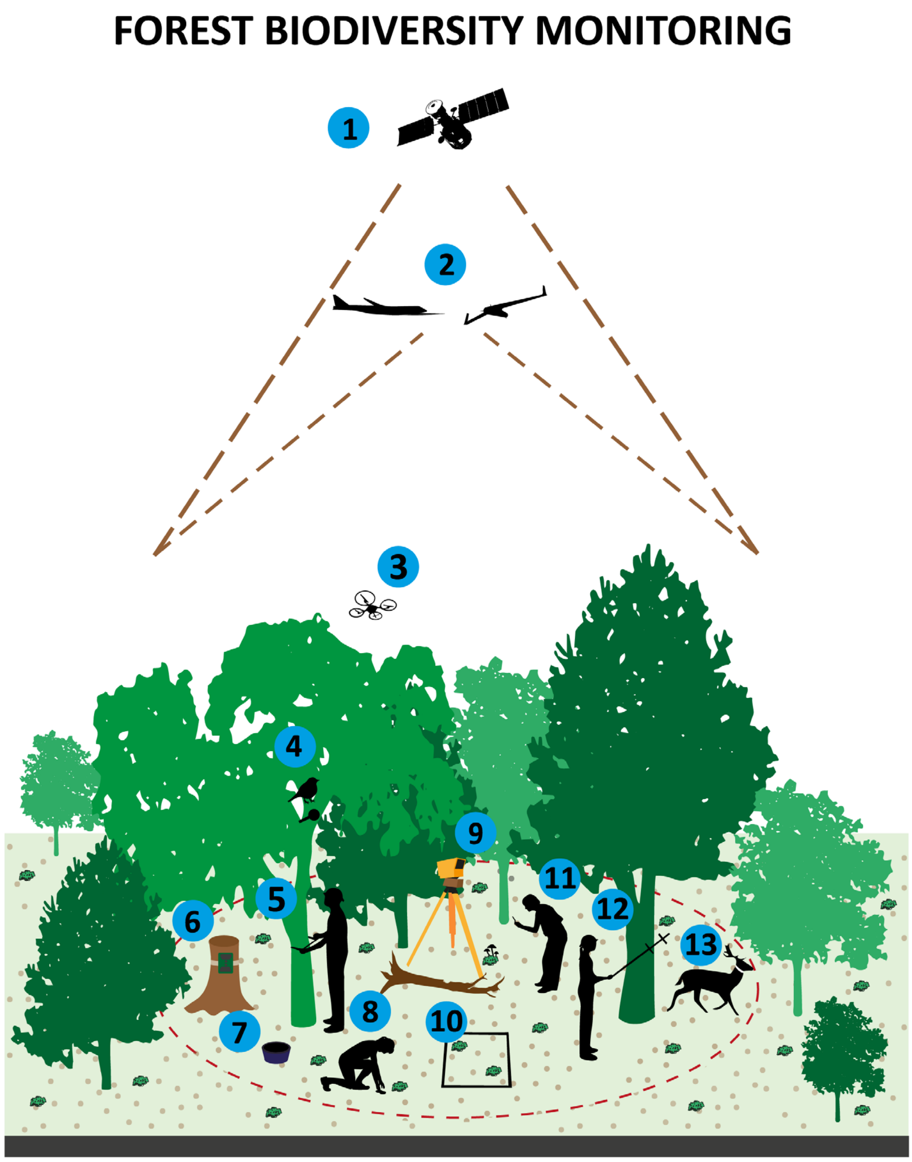
Biodiversity is currently being lost at an unprecedented scale. To guide and assess biodiversity policies and management strategies aiming at reversing this trend, regular, **reliable and standardised information on the state of biodiversity** is needed. Potentially, hundreds of variables can be measured to study, report on and manage biodiversity, however, many biodiversity assessments around the world still primarily focus only on species inventories. As discussed in Chapter 2, biodiversity refers to more than the number of species and population sizes as it also encompasses other components and dimensions. Without a standardised framework to deliver harmoniously measured, regular, timely data on the major and complementary components and dimensions of biodiversity change, it becomes difficult to identify the mechanisms driving biodiversity loss and reducing ecosystem services at multiple spatial scales, and then determine the best solutions to prevent such loss. Having said that, many tools are available to reliably monitor various dimensions of biodiversity at multiple scales, such as eDNA, camera traps, drones and satellite remote sensing. While the use of such modern technologies has expanded scientists' options drastically (Figure 4), as mentioned above, the difficulty comes in interpreting the data these tools provide.

This chapter focuses on European forest biodiversity and its monitoring, paying attention to the different components and dimensions of biodiversity and how these can be monitored. To organise such knowledge, the chapter is structured around the six classes of **essential biodiversity variables (EBVs)** globally recognised by the Group on Earth Observations Biodiversity Observation Network (GEO-BON), which allow quantification of the rate and direction of change in complementary aspects of the state of biodiversity over time and across space (Peirera et al. 2013). These classes are **(i) genetic composition, (ii) species populations, (iii) species traits, (iv) community composition, (v) ecosystem structure and (vi) ecosystem functioning** (Pereira et al. 2013). EBVs are expected to be critical for understanding and predicting changes in the most integrated and established global indicators of biodiversity. As such, they form an intermediate layer between raw data from in situ/remote sensing observations and international indicators used to communicate the current state of and trends in biodiversity (Table 1).

The first part of this chapter explores these approaches to monitoring, while the second part summarises the state of knowledge on biodiversity status and trends for European forests. The chapter then concludes with a discussion on existing knowledge gaps and perspectives.

Table 1. EBV classes and associated examples of EBVs.

EBV class	Examples of EBVs for this class
Genetic composition	Genetic diversity, effective population size, inbreeding
Species populations	Species distributions, species abundance
Species traits	Morphology, physiology, phenology
Community composition	Community abundance, phylogenetic diversity, trait diversity
Ecosystem structure	Ecosystem distribution, ecosystem vertical profile
Ecosystem functioning	Primary productivity, ecosystem disturbance regime



1. Satellite imagery
2. Airplane & UAV imagery
3. Drone imagery
4. Sound recorder
5. Tree inventory
6. Camera trap
7. Pitfall trapping
8. eDNA sampling
9. Ground-based LIDAR
10. Vegetation relevé
11. Citizen science with cell phone
12. Wildlife officer receiving antenna
13. Transmitting collar

Figure 4. A sample of state-of-the-art forest biodiversity monitoring from regional coverage using satellites to local ground coverage using high intensity biodiversity monitoring plots. Graphic inspired by Bennett and Tkacz 2008, Turner 2014, Hardisty et al. 2019, Pennisi 2021.

3.1 Forest biodiversity monitoring: from genes to ecosystems

Genetic composition

Genetic diversity is of paramount importance for forests' resilience to environmental change. Essentially, greater genetic diversity within a given species means that its populations are likely to be more apt to withstand variable environmental conditions. There are multiple variables that can capture information about the genetic composition in forest ecosystems, such as measures of allelic diversity within and between populations of a species.

The importance of monitoring genetic diversity has been discussed by the Food and Agriculture Organization's (FAO) Department for Forest Genetic Resources since 1970. Their proposal on the implementation of forest genetic monitoring was published in 1996 (Namkoong et al. 1996), however, their recommendations were not considered feasible due to the high costs and logistical constraints. Following the 2015 European Forest Genetic Resources Programme (EUFORGEN) recommendations for **genetic monitoring** of forest trees in Europe, the Manual for Forest Genetic Monitoring and associated guidelines was published in 2020 (Bajc et al. 2020). This manual provides concise descriptions of the scientific principles behind the proposed forest genetic monitoring system, descriptions and explanations of the indicators, verifiers and background information as well as a description of different levels of intensity of genetic monitoring. In addition, it provides step-by-step instructions on how to carry out all the activities necessary to implement and conduct forest genetic monitoring, and assess the costs of monitoring for each of the above-mentioned levels of intensity.

To date, **forest genetic monitoring** has been primarily confined to deriving information from the sequencing of tree samples. However, a number of studies have started to highlight how remote sensing technology may contribute to forest genetic monitoring in the future (Czyż et al. 2020). Furthermore, recent work has outlined the potential for **eDNA** to improve the collection of population genetic data (Adams et al. 2019). eDNA is DNA that is collected from a variety of environmental samples such as soil, seawater, snow or air, rather than directly sampled from an individual organism. Example sources of eDNA include, but are not limited to, microbes, spores, faeces, mucus, gametes, shed skin, carcasses and hair. eDNA **metabarcoding** is a novel method of assessing biodiversity wherein DNA is extracted from such samples, and then amplified to generate thousands, or even millions, of reads. From this data, species presence can be determined and genetic information gathered. These technological developments are gaining traction with scientists, especially given their cost effectiveness and ability to access standardised information across landscapes. Coupled with greater efforts to engage practitioners with the benefits of incorporating genetic monitoring in forest management, there could be a significant boost in the near future to our ability to track changes in the genetic composition of key forest species across Europe.

Species populations

Monitoring the conservation status of all species and habitats of 'community interest' is an obligation under the European Habitats Directive and every Member State is required to report on the progress made in implementing the Habitats Directive every six years. Information on species populations underpins, among other types of information, the assessment of the conservation status of species for the IUCN Red List of threatened species, the world's most comprehensive inventory of the global conservation status of species. These expert assessments bring together information on species distribution and abundance status as well as trends in these numbers to allow evaluation of their extinction risk, robustly highlighting which species require particular conservation attention.

The distribution and abundance of forest-dependent species are traditionally monitored using methodologies such as plot monitoring, line transects and sequential soil coring, often focusing on typical or easily identifiable **indicator species**. The emergence of new technologies, such as **remote sensing** (particularly camera traps, bioacoustics, drones, sensors onboard planes and satellite remote sensing), eDNA and **citizen science**, increasingly refine and complement these data. For example, hyperspectral sensors onboard drones, planes and satellites can be used to assess the distribution of specific tree species in given ecosystems (Asner and Martin 2016). Genetic material can be collected from soil, water bodies, watersheds and even from the air, to help monitor diversity and the abundance of target species or species groups (Crookes et al. 2020, Leempoel et al. 2020, Clare et al. 2021). A key strength of using eDNA to provide data is that it provides information about hard-to-sample species and is less reliant on imperfect indicator species. The emergence of rapidly improving **species recognition apps** employing machine learning has gone hand in hand with increased enthusiasm for species monitoring by citizen scientists and has boosted biodiversity mapping efforts. The amount of data collected by citizen scientists now represents a significant proportion of the information available to scientists for assessing and modelling changes in species distribution (Chandler et al. 2017), particularly when it comes to birds, butterflies, moths, plants and fungi. However, there are several challenges associated with the use of these technologies. For example, a broad system for the standardisation of eDNA workflows still needs to be developed (Harper et al. 2019), while poorly documented and varying sampling efforts can represent a major issue when making inferences based on citizen science data (Sicacha-Parada et al. 2021, Bradter et al. 2021). In general, the collection and interpretation of citizen-science sourced data is complex and requires careful calibration with consolidated identification and counting methods coupled with reviewing by expert panels.

Species composition and species traits

Forest composition monitoring refers to the assessment of species composition and richness in a given forest ecosystem. These assessments have traditionally relied on the quantification of **alpha, beta and gamma diversity** (see Chapter 2). Species-based metrics that are based on the number of distinct species in a defined area (i.e., species richness) remain the most common approach to look at ecological community composition. Such information is increasingly being gathered using remote sensing technology, citizen-science approaches and environmental DNA surveys. Rapid progress is, however, being made both in terms of defining best practices and accounting for bias using robust analytical solutions, meaning that the importance of remote sensing approaches to improve forest biodiversity monitoring in Europe is likely to increase in the near future.

Species **traits** can be broadly understood as those qualities common to all organisms of a species, such as body mass, length or basal metabolism. **Functional traits** are a specific type of trait that relates to the characteristics of an organism that are considered relevant to its response to the environment and/or its effects on ecosystem functioning. **Functional trait diversity** has been shown to underpin ecosystem functioning and ecosystem services delivery (Duncan et al. 2015). In many situations, species trait monitoring requires considerable on-the-ground monitoring efforts, such as home range size and lifespan, and is primarily monitored through medium to long-term projects. However, several traits, such as phenology or tree height, can be derived from satellite data, camera traps and aerial pictures. Systematic monitoring of species traits at large spatial scales remains rare, meaning there is still much to do to compile comprehensive, European-wide portfolios for most taxa.

Ecosystem extent and structure

Forest structure refers to an amount or arrangement of entities within a particular space. This structure can be broken down into several components, including structural components, trophic components, as well as vertical and horizontal arrangements.

- **Structural components** of forests typically include water bodies, deadwood and tree cover, the number and condition of which can be derived from ground-based or remotely-sensed information depending on the expertise, budget and logistical constraints of the monitoring entities.

- **Trophic structures** are formed by the arrangements of food webs. Understanding such webs requires comprehensive information on species occurrence in a given forest ecosystem as well as information on the local diets of each species present. These trophic structures can be challenging to assess as they require, in many situations, stomach content or DNA analyses to identify dietary composition.

- **Vertical structures** are formed by the vertical arrangement of canopies and tissues within a forest ecosystem and are a key component of habitat quality for many species. Typical variables capturing information about the complexity in the arrangement of vertical structures include the structural complexity, variability and distribution of a forest canopy as well as its root/shoot ratio. These variables can only be monitored at large spatial scales using either extrapolation models of ground-based information or a combination of remote sensing data collected by passive (multispectral) and active (LiDAR, RADAR) sensors.

- **Horizontal arrangement** typically refers to fragmentation metrics, such as the number of fragments, the distance between fragments and the size of ecosystem fragments or ecosystem components. Although long-term field experiments have been invaluable for investigating the outcomes of fragmentation on biodiversity, remote sensing is still considered the only viable approach to provide comprehensive data on horizontal ecosystem structure at larger spatial scales. The costs and technical challenges associated with horizontal arrangement assessments are relatively low, however, these challenges tend to increase as the spatial extent and desired resolution detail increase and if the considered landscape is complex.

The **National Forest Inventories (NFI)** of the different European countries form a huge network of permanent monitoring plots, often with a plot resampling interval of 10 years. This network was originally designed to monitor tree species composition, standing stocks, wood increment, and diameter class distribution in the context of various forest provisioning services. The available data is nevertheless increasingly used as a **broad-spectrum ecosystem assessment tool**, a purpose for which it was not originally intended. As such, while the currently gathered data remains useful, extending the measured variables to indicators for tree species, structural and functional diversity would largely benefit biodiversity conservation efforts.

Ecosystem functioning

Ecosystem functions are ecological processes that control the fluxes of energy, nutrients and organic matter through an environment. Examples of ecosystem functions include soil formation, soil retention, food production and biological control. They differ from ecosystem services, which are defined as the benefits human populations derive, directly or indirectly, from ecosystem functions (Pettorelli et al. 2018). The key distinction between ecosystem functions and services is thus that functions can have both intrinsic and potential anthropocentric values, while services are defined only in terms of their benefits to people.

A wealth of methods relying on the collection of field data is available to locally measure various ecosystem functions. For example, the functioning of European forests is currently assessed by the International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests). This network has been monitoring forest conditions since 1985 at two monitoring intensity levels. In the 5663 Level I plots, tree vitality is routinely assessed on an annual basis while the diversity of ground vegetation and general information on living trees and deadwood are assessed at irregular intervals. In the more than 600 Level II plots, vitality, growth, atmospheric deposition and soil condition are more intensively monitored each year.

However, to obtain large scale information on the functioning of ecosystems, local data needs to be integrated to produce standardised metrics. Because the measurement of the functions themselves is complicated, time-consuming and expensive, very few methods can realistically be scaled up to provide large-scale coverage on a regular (daily, weekly or even monthly) basis. Therefore, **proxies are often used** instead (Stephens et al. 2015), as is the case with the use of standing biomass to forecast productivity. Proxies derived from satellite imagery hold great potential for cost-effectively tracking changes in certain ecosystem functions at scale, such as productivity, although in many cases methodologies still need to be agreed upon (Pettorelli et al. 2018).

3.2 Overall trends in European forest biodiversity

Here, some results of European forest monitoring are presented and interpreted in an overview that is not intended to be exhaustive but gives important examples of positive and negative trends.

Species composition and population size

A total of 454 trees and other woody species are known to be native to Europe, yet European forests are relatively homogeneous when it comes to tree composition (Rivers et al. 2019). Around 30% of European forest area is dominated by single species, mostly conifers while 51% of the continent's forest area has only two to three tree species in it and only 5% has six or more tree species (Forest Europe 2011). Monitoring data has so far revealed that 58% of Europe's endemic trees are threatened, while 42% of all native species are threatened with regional extinction. Of the endemic tree species, 15% (66 species) have been assessed as critically endangered (Rivers et al. 2019). These numbers have to be interpreted with care though as they are strongly influenced by a large number of threatened micro-species from the genus *Sorbus,* which are confined to a few locations in the UK and the Carpathians.

According to recent reports by the EEA (2016), 27% of mammals, 10% of reptiles and 8% of amphibians linked to forest ecosystems are considered to be under threat of extinction within the EU. Trends in conservation status are well-established for birds, with 32% of forest bird species showing an improvement while the status of another 40% is stable. Interestingly, populations of common forest bird species have been relatively stable over the past 37 years, with only minor fluctuations observed in the common forest bird index (but see chapter 4 for recent climate change effects). Encouragingly, several mammal species have also increased their distribution range in Europe by approximately 30% since the mid-20th century (Deinet et al. 2013); some of these species (wolf, lynx and bear) have directly benefited from forest expansion (Cimatti et al. 2021). Finally, according to Natura 2000-reporting, forests exhibit the highest proportion of improving trends among the assessments, with 14% of the forest assessments having a good conservation status. However, 31% of forest assessments still show a bad conservation status and 54% a poor conservation status (EEA 2019). This suggests that European forests have not escaped the global trend of biodiversity loss, however, in contrast to other non-forest systems, they seem to show some improvement in parts of their indicator set.

Efforts to track forest species trends are, however, far from being evenly distributed across space and taxonomic groups, with a recent review reporting that arthropods, birds and plants are the most frequently used indicators for biodiversity in European managed forests and this does not paint a complete picture (Oettel and Lapin 2021). This means significant gaps remain in our ability to identify many forest species at risk as, for example, the vast majority of fungal and micro-organism species are yet to be assessed (Gonçalves et al. 2021).

Ecosystem extent and structure

Recent analyses have established that forest extent is increasing slowly in Europe. Forest area in the EU increased 0.2% annually between 2010 and 2020 (Pötzelsberger et al. 2021). However, differences between countries exist as some countries, particularly in northern and south-western Europe, are experiencing a decline in forested areas. Importantly, primary forests continue to be lost both in and outside the EU, particularly in Romania, Poland, Ukraine, Slovakia and the boreal North (Sabatini et al. 2021). Moreover, recent analyses carried out by the Joint Research Centre show a relatively constant degree of forest land cover fragmentation by country over the assessment period, which ran from 2000 to 2018. The UK, Ireland, Iceland and Denmark were found to have the highest proportion (> 30%) of isolated forest fragments. The lowest proportions of fragmentation (~ 1%) are found in Finland and Sweden, while the EU average is approximately 8%. The highest fragmentation increase was found in Spain, which grew from 9.7% in 2000 to 12.9% in 2018 (Vogt 2019). However, it is important to remember here that a low proportion of forest cover fragmentation does not necessarily guarantee a good conservation status, as forest patches with high biodiversity value can still be highly fragmented within a forest matrix (Angelstam et al. 2020).

Large trees, **snags** (standing dead trees), lying deadwood, open spaces and diverse understories are important structural components of forests (McElhinny et al. 2005) and are monitored in some of the NFIs (Vidal et al. 2016). The overall amount of available deadwood in European forests increased between 1990 and 2015 despite an opposite trend in Central and East Europe during this period. This increase could be explained by more frequent disturbances related to climate change and more nature-oriented forest management practices (Pötzelsberger et al. 2021, see further in Chapter 5). For what was then the EU-28, average deadwood amounts were 12 m^3/ha in 2015 (7% of the growing stock), which is lower than the 20-50 m^3/ha proposed as minimum deadwood volumes (Müller and Bütler 2010) and considerably lower than what is found in old-growth forests (Stokland et al. 2012). However, there is a large variation among regions, with total deadwood volume ranging between 2.3 m^3/ha in Portugal and 28.0 m^3/ha in Slovakia (Pötzelsberger et al. 2021). To become a more relevant indicator, the total amount of deadwood needs to be complemented with information about its spread in terms of size and decay stages (Stokland 2001) as well as its spatial distribution (Ódor et al. 2006). For example, while hard deadwood has increased in Sweden due to storm events, decayed deadwood is stable or declining depending on which region is considered (Jonsson et al. 2016).

Ecosystem functioning

Long-term changes in forest ecosystem functions are rarely assessed at the continental scale, leading to significant gaps in our ability to understand and predict changes in ecosystem services delivery. An increase in the frequency of forest disturbances, often linked to augmented defoliation, wind and insect outbreaks as well as the wider occurrence of forest fires, has the potential to erode Europe's carbon storage potential. Opportunities to study the **resilience of forests** and their functioning are possible through the long-term annual monitoring of tree crown condition of almost 110,000 trees across Europe in the **ICP Forests network** (see Chapter 3.1). The overall mean defoliation for all species was 23.3% in 2019 and defoliation increased on 19% and decreased on 9% of the network plots between 2010 to 2018 (ICP Forests 2020). Insects were the predominant cause of damage and accounted for 26% of all recorded damage symptoms, with other prominent causes being fungi (11%) and drought (9%). Although trends in disturbance size were highly variable, disturbance frequency consistently increased while disturbance severity decreased in the 1986-2016 period (Forest Europe 2020; Senf and Seidl 2021a). An interesting point to note in this context is that mixed forests showed stronger resilience to recent droughts than monoculture forests (Sousa-Silva et al. 2018).

3.3 Data gaps and perspectives

Direct information on the different components and dimensions of forest biodiversity in Europe can be difficult to obtain, meaning that in many situations forester managers, governments, statistical offices, NGOs and researchers, rely on other proxies to draw conclusions about biodiversity, often with a strong bias towards tree-specific information. Such proxies include metrics capturing information on the conservation status of tree species as well as metrics considered to correlate with forest management sustainability, such as forest growing stock, inter- and intra-stand age structure and/or tree diameter distribution in forests and other wooded land and the proportion of an area managed for conservation.

Many biodiversity-monitoring efforts are focused on informing policies at the national and international levels. In particular, the urgency of undertaking large-scale mapping of high conservation value forests, which is an important step for policy formulation ensuring their conservation, can be highlighted here. However, there is also a pressing **need to monitor biodiversity at the forest management unit (FMU) level** within the framework of forest management planning and evaluation. The principles and approaches to monitoring biodiversity at the FMU level are similar to those undertaken at the national and international levels but, in general, forest managers have less knowledge and funding to engage in state-of-the-art monitoring and often continue to simply measure standard forest inventory variables. Attempts have been made to compose **indicator baskets** targeting the most **cost-effective way to evaluate biodiversity at the stand or FMU levels** (Angelstam and Dönz-Breuss 2004; Maes et al. 2011), but more work is needed to standardise and promote these among forest managers. There is huge potential for training and collaborative learning, which may turn stewards and managers of forests into more effective sources of forest biodiversity information.

The number of forest biodiversity indicators being suggested as valuable by scientists, practitioners and policy-makers has been growing steadily over the past few decades leading, in many cases, to situations where agencies are having difficulties in identifying which of the large number of proposed indicators they will employ in their data collection efforts. This steady increase in indicators has also rendered prioritising funding for large scale data collection and analysis more difficult. A relatively recent evaluation of the strength of evidence for many of the indicators showed that the validity of most indicators on which monitoring and conservation planning is based are scientifically weak, highlighting the dangers associated with decision-making based on indirect proxies of forest biodiversity (Gao et al. 2015). Additionally, forest biodiversity **monitoring efforts are currently not sufficiently aligned with EBVs**, showing different priorities and methodologies among countries. Uniform pan-European gridded datasets to inform researchers, policy-makers and conservationists on the state of forest biodiversity would vastly improve our ability to track changes in our natural capital and predict how it may fare in the future. Recently, 29 countries started an integrative harmonisation process through the foundation of the European National Forest Inventory Network (ENFIN). Initiatives such as these are important steps towards data standardisation, which is required to improve forest biodiversity monitoring in Europe.

4. Main threats to European forest biodiversity

Chapter highlights

The need to implement policy processes to reduce external threats, such as climate change, atmospheric pollution and invasive species, all of which negatively impact European forest biodiversity, remains urgent.

Several forest management practices are harmful to biodiversity, but the forestry sector has the potential to reduce these impacts.

In this chapter we position the well-known quintet of major causes of biodiversity loss at global level, **namely pollution, climate change, biological invasions, land-use change and overexploitation** (Brook et al. 2008) in a European forest context. To make these threats even more relevant to forest stakeholders and actors when assessing their negative effects on forest biodiversity, we distinguish between **'external' threats** that are beyond the direct influence of forest owners and managers and **'internal' threats** that are directly related to forest management practices (Figure 5).

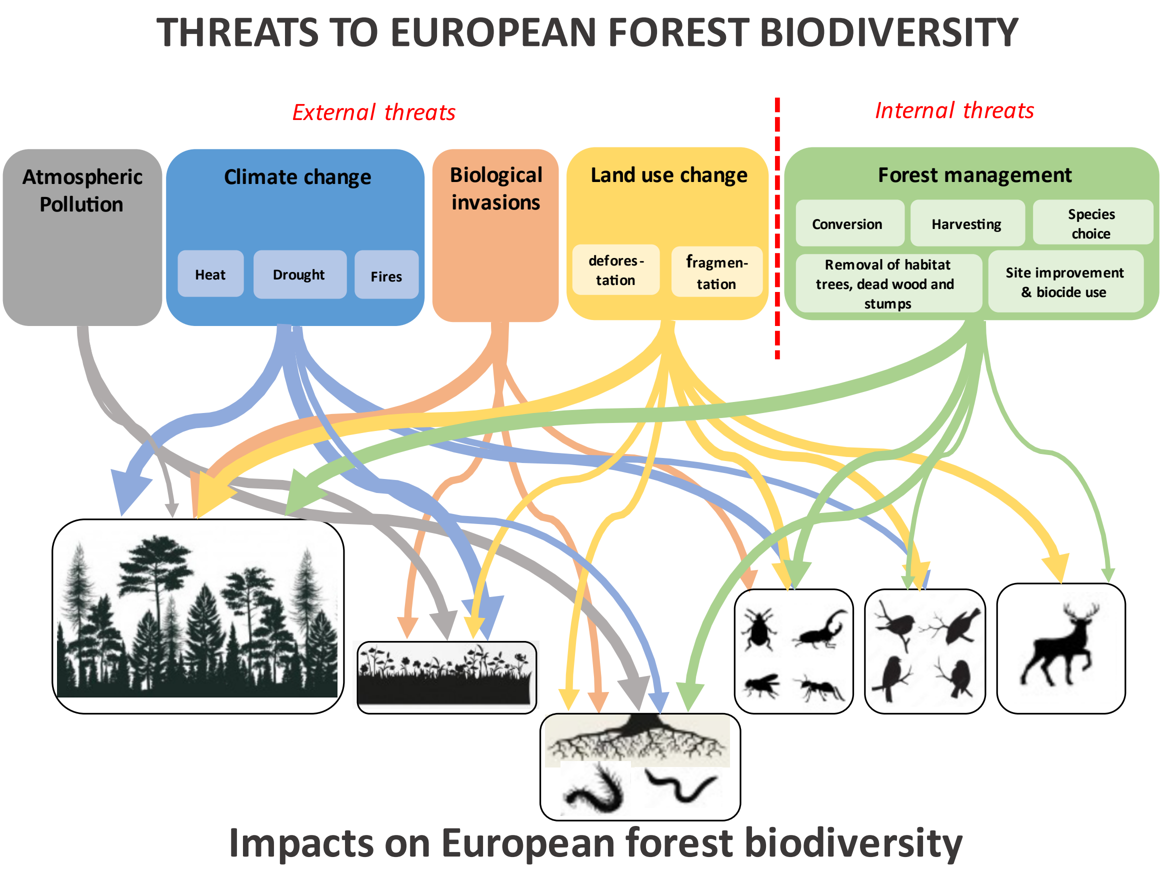


Figure 5. The relationships between major threats to biodiversity in European forests and particular groups of species (from left to right: trees, understory vegetation, soil organisms, insects, birds, mammals). The thickness of the arrows indicates the estimated magnitude of effects based on expert opinion. Indirect effects, such as changes in forest stand structure, are not represented.

4.1. External threats

Pollution

Decades of high inputs of atmospheric pollutants, such as **sulphur dioxide, nitrogen oxides and ammonia**, have increased the acidity and nitrogen content of forest soils over large parts of Europe (Arnolds 1991). The resulting **acidification** and **eutrophication** have caused shifts in the structure and functioning of forest soils, including mycorrhizal community structure and diversity (Kraigher and Al Sayegh Petkovšek 2011). They are also responsible for changes in the composition of plant communities (van Dobben and de Vries 2017), with a decline in the diversity of vascular plants and specific species adapted to poor soils, such as heathers and lichens and an increase in nitrogen demanding plants, such as some grass species (Strengbom et al. 2001, Eichenberg et al. 2021). These changes can cause cascading effects in forest food webs where, for example, an altered plant composition might result in decreased snail populations, and the lower consumption of snails by birds reduces their calcium intake, leading to lower breeding success due to more fragile eggshells (Graveland et al. 1994). The gradual decrease in nitrogen and acid deposition currently taking place across Europe has not yet translated into clear improvements in forest ecosystem functioning (Schmitz et al. 2019) or species recovery (e.g., Esseen et al. 2022) because input levels continue to exceed critical loads in many regions. Also the secondary pollutant **tropospheric ozone** resulting from atmospheric processes in a polluted environment continues to cause damage to the health of trees and other forest dwellers (Lin et al. 2020). The proximity of forests to agricultural fields can also lead to pollution exposure caused by the drift of applied biocides (including insecticides, fungicides and herbicides etc.). These **biocide residues** can, for example, have a deleterious effect on forest-dwelling insects via a reduction in their abundance and diversity (Seibold et al. 2019; Brühl et al. 2021).

Climate change

Anthropogenic greenhouse gas emissions are rapidly and drastically altering climatic conditions worldwide, leading to **elevated temperatures** and an increase in the frequency and intensity of **extreme weather events** such as heatwaves, heavy precipitation and **droughts** (Lange et al. 2020). The effects of such changes on forest biodiversity are complex, particularly because they are interacting with other pressures like land-use change or biological invasions, thereby e.g., increasing the destructive power of **wildfires** or facilitating invasive species expansion. By 2070, approximately 45% of all protected areas in temperate deciduous and mixed forests are expected to experience climatic conditions beyond the limits of historical variability, relative to the 20th century (Hoffmann et al. 2019). This will cause **range shifts** for many species which will often not be possible within the currently protected area boundaries (Araújo et al. 2011). Forest dwelling species with low dispersal ability will face a fragmented European forest landscape and be hindered to respond to changing habitat. More generally, extreme climatic events may have direct deleterious effects on the survival of individuals and the dynamics of populations, generating increased risks of local extinctions. In the following paragraphs, the effect of temperature rise, extreme droughts and forest fires are discussed in more detail.

Temperature rises and heat waves

The response of forest biodiversity to temperature rises and heat waves is mainly observed through changes in the spatial distribution of species. A global review of reported climate change effects on biodiversity mentions shifts in the distribution of tree species along with the associated forest-dwelling species biomes (Scheffers et al. 2016). European-scale climate projections confirmed major **poleward shifts** of dominant tree species, which could result in **local extinctions** or at least a continent-wide reduction in their distribution range (Dyderski et al. 2018, Benito-Garzón et al. 2019).

While studies have predicted range reductions for many tree species, herbivorous forest insects are expected to generally benefit from such future changes in climatic conditions, with warming conditions leading to shorter generation time, higher fecundity and survival rates coupled with an expansion of their ranges. This will, in turn, lead to a potentially higher number and severity of **pest outbreaks** (primarily bark beetles) and, as a result, increased tree mortality (Jactel et al. 2019). However, temperature extremes such as heat waves are also known to be a potential danger to the survival of insects given their sensitivity to thermal shocks and the higher frequency of lethal temperatures, which is often around 40°C (Jactel et al. 2019). In addition, it has been shown that climate warming reduces the amount of snow cover in winter and increases litter decomposition, which has a detrimental effect on the diversity of insects requiring thick litter layers to overwinter (Harris et al. 2019). Such loss in forest floor mass and associated organisms has already been observed in different parts of Europe (Wellbrock et al. 2017).

A decline in bird diversity has already been recorded for birds in boreal forests and Europe's higher mountain ranges, with further range contractions predicted as global warming continues (Virkkala 2016). In addition, changes in tree phenology (e.g., the start date of leaf flush in Spring) may lead to a **mismatch with herbivorous insect phenology** (e.g., egg hatching) resulting in a lower abundance of prey for insectivorous birds (Renner and Zohner 2018).

Extreme droughts

An increase in the duration, frequency and intensity of droughts caused by climate change will lead to a significant increase in tree mortality worldwide and particularly in Europe (Allen et al. 2010; Anderegg et al. 2015a). Heatwaves combined with the increased water stress associated with ever-hotter droughts can increase **forest microclimate alteration and forest mortality** (Allen et al. 2015; De Frenne et al. 2019; Senf et al. 2020, Schuldt et al. 2020). The main physiological mechanism explaining drought-induced tree mortality is xylem **hydraulic failure**, namely the risk of cavitation in the stem vessels (Adams et al. 2017). Many European tree species, such as poplars, willows, alders and maples, are considered at high risk of mortality from extreme droughts, whereas junipers, cypresses, yews and box trees are more drought resistant (Choat et al. 2012). Water-stressed trees can also be more vulnerable to forest pests and diseases, such as the aforementioned **bark beetle outbreaks** (Jactel et al. 2012; Anderegg et al. 2015b; Seidl et al. 2017). The resulting rates of tree mortality will threaten the persistence of all organisms that depend on the resources and habitats provided by living trees. Of particular concern is the fact that many of Europe's large and dominant tree species that are particularly important for forest biodiversity (Bütler et al. 2013) seem to be the most vulnerable to drought (Taccoen et al. 2021). Decreased precipitation, which leads to the drying of forest soils and ponds, has been found to result in a significant **reduction of amphibian diversity** (Walls et al. 2013). However, the **increased abundance of dead trees** is expected to benefit deadwood dependent species such as saproxylic beetles and fungi, which are highly-diverse species groups (Lassauce et al. 2011) and of high conservation value because they are comprised mostly of forest-habitat specialists (Ranius et al. 2011).

Forest fires

Forest fires are a **natural disturbance** to which biodiversity in parts of Europe is well-adapted and, for some species, is even required (e.g., some Mediterranean pines). However, rising temperatures coupled with more frequent and severe droughts, in combination with land abandonment and associated biomass accumulation, have all contributed to **an increase in the occurrence and extent of large forest fires** in Europe (Senf and Seidl 2021b). Fires have obvious direct adverse effects on forest species by burning vegetation and killing any animals not able to escape the flames (Ward et al. 2020). This is particularly critical when fires become **hotter and more intense** or occur in forest ecosystem types that have not evolved in a manner that allows them to recover from recurrent forest fires. Small, remnant habitat patches (e.g., bogs) that cannot easily be recolonised are one example of a particularly at-risk ecosystem. Forest fires can also have significant adverse belowground effects as they affect soil microbes and ectomycorrhizal fungi (Dove and Hart 2017). Fire frequency and intensity is key: while occasional fires can encourage the regeneration of certain tree species (Bobiec et al. 2019), overly recurrent fires may prevent natural forest regeneration and significantly impact biodiversity (Adámek et al. 2016).

Although the abiotic hazards caused by climate change each have their own effects on biodiversity, it should be remembered that they also often have cumulative or interactive effects (Seidl et al. 2017), and that certainly holds true in a forest fire context. A striking example of this phenomenon is the outbreak of bark beetles, particularly the spruce bark beetle (*Ips typographus).* Windthrow created by storms serves as an ideal breeding substrate and, if the temperatures are warm enough, beetles can have one extra breeding cycle per year, increasing their population to epidemic levels. This leads to them attacking trees in the infected area, which are often already under water stress, in overwhelming numbers and resulting in a very large increase in the number of dead trees that, in turn, provide **abundant fuel for forest fires**.

Biological invasions

Non-native species may arrive in forests by planned introduction, accidental release and migration. The increase in the volume and speed of global trade and travel has heightened the **risk of invasion by non-native species** worldwide (Chapman et al. 2017), including in Europe, meaning that invasions are exponentially increasing for all taxonomic groups (Seebens et al. 2017). For example, around 450 species of non-native insect species are currently known to have colonised European forests, a number that increases by an average of 6 each year (Roques et al. 2020). About 400 non-native plant species have also been detected in European forests (Wagner et al. 2017), although to date only a small share of them are potentially invasive. However, several of the at least 150 non-native woody plants introduced to European forests are known for their potential invasiveness (Pötzelsberger et al. 2020). These invasive species include acacias and mimosas, black locust, black cherry, tree of heaven, red oak and box elder. Their impact on European flora is mainly seen in their **use of water, nutrient and light resources, leading to a competitive struggle with native trees** occupying the same ecological niche and subsequently leading to a reduction of native natural regeneration (Langmaier and Lapin 2020).

The direct mortality effects of **invasive herbivorous insects and fungal pathogens on native trees** are much more extensive (Lapin et al. 2021). Most European elm species are almost extinct because of Dutch elm disease, which is caused by a fungus transmitted by bark beetles (Santini and Battisti 2019). Ash trees are suffering the same fate due to ash dieback, also caused by a fungus. The pinewood nematode is severely threatening pine species in southern Europe, while the box tree moth can kill 100% of the box trees in warm areas. All of these tree species' losses lead to extinction cascades for the organisms that depend on them. For example, 43 fungi and 18 invertebrate species are exclusively found on box trees and are thus endangered by the box tree moth invasion (Mitchell et al. 2018). Similarly, the 97 lichen species specifically found on elm trees in the British Isles have been severely threatened by the Dutch elm disease (Watson et al. 1988) and 115 species closely associated with European ash trees in Sweden are at high risk of regional extinction (Hultberg et al. 2020).

Several **exotic bird and mammal species**, such as ring-necked parakeets (Strubbe and Mathysen 2007) and raccoon dogs (Dahl and Åhlén 2019) that were kept as pets or for material provisioning (fur, game meat etc) have escaped captivity and established viable populations in European forests, often at the expense of native species' habitats and population densities.

Land-use change and forest fragmentation

The forest area of the European continent has strongly increased over the last three decades, gaining approximately 28,000 km² net forest cover per year (Song et al. 2018). This has come about partially as a consequence of active afforestation but primarily as a result of agricultural land abandonment (Buitenwerf et al. 2019, Hampe et al. 2020). The biodiversity outcomes of these new forests are time and context dependent, with both species gains and losses being reported. In the first phase following a deforestation event, new forest patches form dense thickets of pioneer trees, pushing the species found in the preceding cultural landscape toward extinction, however, many forest species, especially those typical for ancient forests, can be extremely slow to reach and colonise these new forests, depending on the surrounding landscape (Andrés and Ojeda 2002, Martín-Forés et al. 2020). Moreover, the net increase in forest area may result in mainly homogenous wood-production oriented stands and masks the increasing loss of primary and old-growth areas that are being deforested at an average rate of 1000 km² per year (Nabuurs et al. 2022).

Deforestation and forest conversion often destroy primary, old-growth, mature and even sometimes ancient forests (Sabatini et al. 2018), leading to the increased fragmentation of forest landscapes and further biodiversity loss. In a pan-European context (CORINE land cover maps), 8% of forest landscapes are severely fragmented (Vogt 2019) and within the forest matrix, the remaining patches of primary, old growth and ancient forest are very fragmented. Both forest habitat loss and fragmentation at the landscape level can have profound detrimental effects on biodiversity. Smaller and more isolated patches of forest habitats have lower species richness of fungi (Abrego and Salcedo 2014), shade plants (Bähner et al. 2020), birds (Enoksson et al. 1995; Bełcik et al. 2020), forest specialist ground beetles (Niemelä 2001) and saproxylic beetles (Brunet and Isacsson 2009). Additionally, woody plant species face increased risk from a loss of genetic variation and fitness following habitat fragmentation (Vranckx et al. 2012). By way of a final point to this section, forest fragmentation also interacts with climate change as fragmentation leads to smaller populations and thus lower genetic variation which, in turn, results in lower adaptive capacity when organisms are confronted by other negative environmental drivers such as climate change.

4.2. Internal threats related to forest management

Anthropogenic disturbances connected with our use of forests very often have a much stronger impact than most natural disturbances. While natural disturbances leave so-called biological legacies in the form of residual surviving trees and deadwood, they also promote regeneration and leave patches of undisturbed stands, whereas human activities, such as wood harvesting, very often remove most of the biomass and do not leave any undisturbed older stands in a landscape. Forest management with a focus on wood production leads to a dominance of young and middle-aged homogeneous stands of medium to high density across Europe (see Figure 2). Meanwhile, clearcutting which entails the removal of all trees followed by replanting has been practised on a significant part of Europe's forest land over the last 150 years, and has led to landscape homogenisation and caused a significant decline in forest biodiversity. In contrast to traditional forms of forest use, such as coppice, coppice with standards and wooded pastures which result in high structural diversity within and between stands, even-aged silvicultural systems often produce less stand-level heterogeneity and harm many forest species (Heidrich et al. 2020). In addition, the impact of wood harvesting on biodiversity is exacerbated by the continued export of biomass, which would otherwise remain available to support natural food webs (Lindeijer 2000). The decisions of forest managers and the implementation of these silvicultural operations can, therefore, have major impacts on forest biodiversity. When considered holistically, and despite the increase in protected and voluntarily set-aside areas in recent years, the intensification of forest management has led to the deterioration of habitat network connectivity (Svensson et al. 2019, Angelstam and Manton 2021).

Conversion of old-growth and primary forests

The removal of woody biomass from Europe's last intact primary and old-growth forests (Barredo et al. 2021) is ongoing (Svensson et al. 2019, Angelstam and Manton 2021), sometimes for economic reasons, sometimes in the context of sanitary logging after insect infestations. From a biodiversity conservation point of view, the transformation of these pristine forests into secondary forests equates to severe forest habitat degradation, even if the tree canopy cover does not change (Angelstam et al. 2021). The main negative impacts on such forests involve their specialist species, in particular those dependent on large, old trees or deadwood as well as those needing forest connectivity and temporal continuity (Paillet et al. 2010; Roleček and Řepka 2020; Angelstam et al. 2020).

Other cutting and harvesting interventions

Tree cutting is a crucial intervention in forestry, not only because it provides wood to human society, but because it redistributes light and soil resources that promote either the growth of individual trees (stand tending, including thinning) or enables tree regeneration (regeneration cuttings in silvicultural systems). **Thinning operations** are implemented throughout the forest rotation cycle to support the growth of selected trees by reducing competition with their neighbours. The resulting decrease in stand density induces changes in light, temperature, water and nutrient regimes, all of which have various effects on habitat conditions. While strong thinnings promote the growth of light and warmth demanding plant species, they have negative impacts on shade-demanding understory plant species and ectomycorrhizal community composition (Buée et al. 2005). Thinning from below, including the removal of suppressed and poorly formed trees, reduces the amount and diversity of tree-microhabitats (Courbaud et al. 2022). Regeneration cuttings, including clearcut, strip cut, shelterwood cut and patch cut systems, are generally more intensive and have therefore a larger impact on biodiversity than thinnings. Altering the age distribution of trees in a forest ecosystem toward younger classes has negative effects on biodiversity (Eggers et al. 2020; see Figure 2). Reviews comparing different **harvesting regimes** conclude that more negative effects on biodiversity occur when clearcuts are applied, although the scale and severity of these effects vary among different taxonomic groups (Rosenvald and Lohmus 2008; Fedrowitz et al. 2014; Chaudhary et al. 2016). In general, the abundance and species richness of specialist species strictly related to forest habitats, such as some birds, saproxylic insects and fungi, is decreased by clearcuts while the diversity of species preferring open habitats, such as vascular plants of grasslands and forest edges as well as many forest insects adapted to gaps and disturbances, may increase. Experimental research showed that shelterwood cuts were harmful to the genetic diversity of pioneer tree species while clearcuts, retention tree cuts and patch cuts eroded the genetic diversity of late-successional tree species (El-Kassaby et al. 2003). An important shortcoming of many studies focused on harvesting's effects on biodiversity is that they only look at selected species groups rather than the full range of forest dwelling species (Oettel et al. 2021). The effects of different harvesting methods on diversity at the landscape scale have also barely been assessed (however, see Schall et al. 2018, Angelstam et al. 2018, Naumov et al. 2018, Eggers et al. 2020). A final point of concern here is that the use of heavy machinery in harvesting operations can lead to soil compaction, which can result in a significant decrease in ectomycorrhizal fungal abundance and diversity (Wiensczyk et al. 2002).

Removal of habitat trees, deadwood and stumps

Cavity-bearing habitat trees (Figure 6) are among the most important habitats for specialized forest wildlife, including forest bats and birds while several highly threatened forest beetles also depend on old cavities with organic matter. Such trees are a typical element of old-growth forests, although the density of large habitat trees is often less than one per hectare in managed forests (Bütler and Lachat 2009, Larrieu and Cabanettes 2012; Larrieu et al. 2018). However, in managed forests and cultural landscapes, they are still routinely cut for safety reasons or economic return.

Sanitation cutting, salvage logging and the **removal of deadwood** is a common practice in many European countries to maintain either the traditional ideal of a clean and tidy forest or required by law for perceived reasons of forest health or human safety. However, deadwood provides a habitat for a myriad of saproxylic species of which many species are characterised by a narrow host range (Stokland et al. 2012). For salvage logging in the context of bark beetle we refer to Hlásny et al. (2019) for a critical review of management options. Although there might be economic and operational reasons for salvage logging, there are also many instances where these gains would be minimal, while the biodiversity gain is high. In some intensive forestry regimes, clearcutting may also be accompanied by **stump removal** that can have negative effects on soil structure and soil fauna (Kataja-aho et al. 2011; Taylor and Victorsson 2016) while simultaneously removing valuable habitats for many saproxylic insects and fungi (Hjältén et al. 2010; Brin et al. 2013; Victorsson et al. 2013).

Tree species selection for afforestation and reforestation

Threats to biodiversity also arise from the **cultivation of non-native tree species**. The latter have not shared a co-evolutionary path with other non-tree species native to Europe and are therefore less likely to provide these species with suitable habitats (Felton et al. 2013, Gossner et al. 2009). While **tree species choice** will influence canopy closure and light transmission for the understorey vegetation or the diversity of tree-related microhabitats (Larrieu et al. 2018), it also affects leaf litter composition (nutrient rich litter versus nutrient

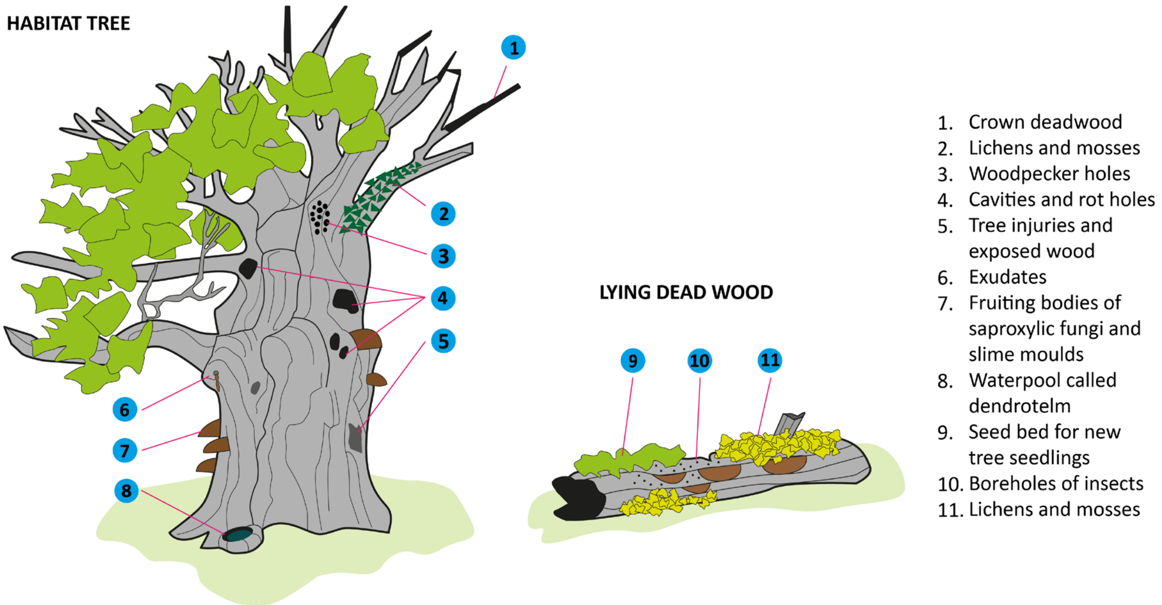


Figure 6. Habitat trees and dead wood offer a large variety of microhabitats supporting biodiversity, and may be threatened by harvesting in managed forests (inspired by Bütler et al. 2020, Humphrey and Bailey 2012).

poor litter, Desie et al. 2020a) and, in turn, the interplay between soil decomposer communities, the humus type and soil acidity (Desie et al. 2020b). In this context, conversion from ancient broadleaved forests to exotic conifer stands leads to strong, largely irreversible changes in their belowground communities and herb layer (Desie et al. 2019).

In addition to which specific tree species are present in an established tree community, the diversity of those species is also important. At the stand scale, there is ample evidence that homogeneous stands support lower species richness than **mixed stands** (Ampoorter et al. 2020) for many taxonomic groups, such as understory plants (Gong et al. 2021), birds (Castano-Villa et al. 2019) and predatory arthropods (Stemmelen et al. 2022), while other taxonomic groups, such as certain ground beetles (Barsoum et al. 2014; Kriegel et al. 2021) are less impacted. On the landscape scale, a **patch-wise variation of stands** with different tree species results in a higher heterogeneity and may provide higher overall species richness (e.g., Heinrichs et al. 2019).

Site improvement and biocide use

In intensively managed forests, shredding of harvest residues, ploughing and tillage operations to prepare forest soils for planting, all affect biodiversity, as can be seen, for example, by the negative effects **soil work** has on ground dwelling beetles and spiders (Kosewska et al. 2018). Moreover, the use of **nitrogen fertiliser** to stimulate tree growth negatively affects the species richness and diversity of understory plants (Strengbom and Nordin 2008) as well as the abundance of mosses, lichens (Sullivan et al. 2018), mycorrhiza, ground beetles (Rodríguez et al. 2021), amphibians and ungulates (Sullivan et al. 2018). More common in European forests are soil amendments using **limestone, dolomite or rock dust as a soil restoration** measure to compensate for historical acidification and nutrient deficiencies. Depending on the product used and dosage applied, these amendments may cause an abrupt change in soil pH and nutrient content which can have a negative impact on the composition of understory vegetation (Becker et al. 1992). Although the use of **herbicides** in European forestry is rather limited, it is sometimes used for weeding before reforestation or to promote conifer over broadleaved regeneration, which affects both plant diversity and has indirect adverse impacts on wildlife, such a reduced abundance of songbirds and mammals (Guynn et al. 2004). The use of insecticides and fungicides is even more uncommon in European forestry, so that its impacts are not detailed here. In summary, when it comes to **biocides**, most forest management systems have a much lower intensity and frequency of use than in intensive agricultural systems (Freer-Smith et al. 2019), however, when it is used, there are inherent negative consequences.

4.3 Interaction between threats

Measures to mitigate threats to forest biodiversity are primarily a part of the broader efforts to mitigate climate change, curb acidifying and eutrophicating atmospheric deposition and control invasive exotics. However, significant untapped potential to mitigate various threats remains within the reach of forest management itself, not only by answering the crucial question of whether **to manage or not to manage**, but also the question of **how to manage**. The possible approaches and context dependencies of forest management favouring biodiversity are discussed in Chapter 5.

The diverse range of external and internal threats to forest ecosystems do not occur and act in isolation as evidenced by the fact that many such threats often present themselves simultaneously. The interaction of different threats can increase the vulnerability of organisms, populations and communities to individual stressors. It has, for example, been shown that many tree species are more susceptible to pests such as bark beetles, as a result of temperature extremes, drought events, and air-borne heavy metal disposition (Harvey et al. 2020, Canelles et al. 2021). Internal threats can also lower the resilience of forests to certain threats where, for example, the erosion of genetic diversity is known to limit populations and species ability to adapt to changing conditions (Koskela et al. 2013) while the loss of functional and species diversity impairs the stability of forests in response to disturbances (Jing et al. 2021, Jactel et al. 2021). Importantly, the impact of **threat interactions** is likely to increase given that climate-related disturbances are predicted to be more frequent and severe in the future (Canelles et al. 2021).

As an endnote to this chapter, it is not only important to monitor forest biodiversity but also to **monitor the threats** to that biodiversity (Oettel and Lapin 2021). There are still serious gaps in our understanding of and ability to monitor such threats (Joppa et al. 2016). Attempts were made to include **land-use impact** on biodiversity in commonly used desk-top methods for environmental assessment of products, such as the Life Cycle Assessment (LCA; Wagendorp et al. 2006, Myllyviita et al. 2019), however, no ISO standard has been agreed upon, unlike other impact categories in LCA.

5. Forest management promoting biodiversity

Chapter highlights

The biodiversity of primary, old-growth and ancient forests is both unique and vulnerable and, therefore, urgently requires more complete and careful mapping coupled with conservation measures undertaken at the landscape scale.

Hands-on nature-positive management is possible in every forest managed for wood production, including plantation forests. Doing so begins with increasing tree species diversity, the retention of more deadwood and habitat trees coupled with the use of disturbance patches, all of which are within the means of every forest manager.

Triad management combines segregative and integrative approaches to biodiversity conservation. It is a promising option, but good policy solutions to implement them in mixed ownership landscapes need further development.

As previously noted, the majority of forests in Europe have been managed for a long time. Forest management traditionally focused on wood production but has now evolved to include multiple ecosystem services. This chapter focuses specifically on forest management approaches and techniques that serve to promote biodiversity conservation. These approaches may entail decreased management intensity, or even no management, but not in all cases as, for example, biodiversity associated to cultural landscapes, such as coppice forests and agro-silvo-pastoral systems, will continue to need recurrent management interventions (Bernes et al. 2015). In the first part of this chapter, we present a range of measures that are known to positively influence different aspects of forest biodiversity. Following on from this, the second section combines these measures and discusses them in the context of different forest management systems that range from pure conservation to multipurpose and production forestry.

5.1 The portfolio of measures for managing forest biodiversity

A broad portfolio of possible measures is at the disposal of forest managers to promote and capitalise on biodiversity (see e.g., Agra et al. 2016). These measures have been developed in harmony with the boundary conditions for forest biodiversity in unmanaged natural and managed cultural forests introduced in Chapter 2 as well as by our understanding of management interventions that are harmful to biodiversity, as identified in Chapter 4. The presented measures will sequentially address key components (genes, species and ecosystems) and dimensions (compositional, structural and functional diversity) of forest biodiversity as identified in Chapter 2.

Managing for genetic diversity

The lower hierarchical level of biodiversity to consider in forest management is the genetic diversity of single species and its populations. Genetic diversity is fundamental to a species' ability to adapt and evolve to a changing environment. Adaptative potential depends on complex interactions between population growth, migration, genetic exchange with other populations and loss of genetic variation (Fady et al. 2016). In the following sections we concentrate on the fitness of (a) tree populations and (b) other species.

a) Tree population viability

Studies on the genetic diversity of trees in forests have chiefly targeted economically important tree species. There exist several measures to introduce and maintain high genetic diversity throughout the production chain of forest reproductive material (seeds, seedlings and cuttings). Of great importance here is the use of **recommended registered sources of basic material,** that is the various provenances, stands and seed orchards providing the seeds for afforestation/reforestation, that are available in each country. Other practice-relevant information, such as guidelines on minimum effective population size for natural regeneration and seed harvest guaranteeing sufficient genetic variation, are provided by the EUFORGEN programme (EUFORGEN 2021). The latter provides management recommendations for maintaining high genetic diversity in tree populations and is being expanded to include data on rare and vulnerable tree species, like Sicilian fir, Taxus or Common Ash. The data provided by EUFORGEN is also extremely useful for more common species, such as beech, where irregular shelterwood showed minimal impact on tree genetic diversity in comparison to unmanaged old growth (Westergren et al. 2015), while the high pollen mobility connects otherwise isolated populations and compensates for the negative genetic consequences of small stands and low tree densities (Vranckx et al. 2014). Crucial for the genetic conservation of all tree species is to take advantage of full mast years with massive seed production, both for silvicultural interventions aiming at natural regeneration as well as for seed collection for seedling production in nurseries. If populations are small or genetic diversity is low, natural regeneration can be combined with enrichment planting. In the face of climate change, **climate-smart seed sourcing** (Prober et al. 2015) is often recommended as it consists of seed collection in full mast years to maintain high genetic diversity and **assisted migration** by expert-based mixing that draws on more southern provenances in existing populations (Fady et al. 2016), which is often legally restricted. For further knowledge gain, it is important to ensure the traceability of used forest reproductive material and the implementation of long-term monitoring of the effectiveness of the abovementioned measures, as proposed by EUFORGEN (Aravanopoulos et al. 2015) and detailed in the manual for forest genetic monitoring (Bajc et al. 2020).

b) Other species population viability:

Similar to trees, other species have varying potential to persist, adapt and evolve at a given site depending on the conditions. Gathering data on the genetic diversity and genetic structure of populations informs management in two important ways:

- First, reduced genetic diversity serves as an early warning signal of the genetic impoverishment of regional populations and can be the basis on which targeted population support can be implemented. This approach has been successfully used on the Eurasian lynx (Ratkiewicz et al. 2012), Ural owl (Hausknecht et al. 2014) and Western capercaillie (Bajc et al. 2011), among others.

- Secondly, genetic structure provides important information on the connectivity between subpopulations of endangered forest species as well as their dispersal potential, as has been shown in studies involving Ural owls (Hausknecht et al. 2014), Western capercaillies (Braunisch et al. 2010) and various deadwood insects and wood-dwelling fungi (Komonen and Müller 2018). This information can then be used to guide management interventions such as captive breeding and release, as was done for the European bison (Gippoliti 2004). Providing adapted nesting facilities for birds is widely undertaken in gardens and forests throughout Europe, although this practice may also benefit non-native species (Bailey et al. 2020).

While broadly seen as good practice, information on genetic structure and diversity is only available for a limited number of keystone species, meaning more work needs to be done in this area.

Managing species and habitat diversity at the stand scale

To promote a diversity of habitats and their associated species at the stand scale, forest managers employ approaches that attempt to increase heterogeneity which is predominantly pursued along three major axes: (i) the composition and diversity of trees and shrubs; (ii) variation in the stands' horizontal and vertical structure; and (iii) the amount and variation of deadwood and habitat trees of different kinds. All three axes are significantly reduced in production forests when compared to natural forests (e.g., Angelstam and Dönz-Breuss 2004, Müller and Bütler 2010). Although relevant at the stand scale, the management of game populations is discussed in the next section about the landscape scale.

Enhancing tree species diversity

As explained in Chapter 3, the current diversity of tree species in European forest stands is predominantly low (Forest Europe 2020). Given the positive relationship between tree diversity and the diversity of other forest-dwelling species (e.g., Ampoorter et al. 2020; Barbaro et al. 2019) on the one hand and the need to facilitate the adaptation of Europe's forests to climate change on the other, it appears to be a matter of urgency to **accelerate the diversification of tree species** within forests.

As an initial step, future forest management should aim for a **diversity of site-adapted tree species** that support native biodiversity. Mixtures can be optimised to include different functional types with respect to stress tolerance and ability to promote high species diversity to strengthen forest resilience in the face of rapid climate change (Baeten et al. 2019, Messier et al. 2019). The adaptive potential of genetically-robust native tree species, including previously rare species, should be exploited first when adapting tree species composition to climate change.

A subsequent step can be to include other European tree species that are not native to a given region but are closely related to the indigenous tree species. This may involve, for example, introducing a more drought resistant oak species that currently has a more southern distribution but will most likely support a relatively large proportion of the biodiversity normally associated with indigenous oaks (Vogel et al. 2021). Site-adapted exotic species, such as Douglas fir, for which long and positive cultivation experience exists, can also be used to reduce the risk of invasiveness (Bindewald et al. 2021b). Critical proportions for mixing such non-natives into a matrix of native trees, above which the viability of populations of native forest dwelling species are at risk, are still largely unknown and should, therefore, be carefully investigated.

Beyond the active planting of additional tree species, an increase in the variation of canopy opening (see below) and the suppression of browsing ungulates are the two most effective management strategies to foster tree diversity (Bauhus et al. 2017, Boulanger et al. 2018, Hothorn and Müller 2010). Hence, active management that promotes the survival and growth of rare, light-demanding and palatable (for ungulates) species, for example through early thinning or the creation of mono-specific patches of these species to eliminate competition by other species, can greatly support tree diversity in the long term.

Enhancing forest structural heterogeneity

High **vertical heterogeneity** within stands locally promotes the diversity of different species groups including birds, insects and fungi (MacArthur and MacArthur 1961, Heidrich et al. 2020). Large trees form the most pronounced three-dimensional structure in forests and numerous species are adapted to the habitats provided by this late-successional structural element (Moning and Müller 2009). In silvicultural systems dominated by clearcuts, as is the case in boreal forests, the retention of living trees within the logged area for one or more additional rotations is an important tool to enhance the structural heterogeneity and related biodiversity (Fedrowitz et al. 2014, Thorn et al. 2020). Given that without proper management the survival of retention trees can be low (Rosenvald et al. 2019), science-based intervention guidelines have been developed (Gustafsson et al. 2020) and, interestingly, these guidelines can be quite different from retention levels negotiated in forest certification standards (Angelstam et al. 2013, Kuuluvainen et al. 2019).

It sounds counter-intuitive, but even multi-layered forests, such as selection forests in Central European mountains, can have negative effects on the overall landscape-level forest biodiversity. This is because their complex structure at the stand level is repeated over larger areas which causes unnatural homogenisation at the landscape scale and low beta diversity (Schall et al. 2018). Therefore, besides vertical structural variation in managed forests, significant **horizontal variation** in the density of canopy cover, ranging from very dense stands without regeneration to those with gaps offering areas of full sunlight, is a key requirement to promote forest biodiversity at the landscape scale. The occurrence of disturbance events should be seen as an opportunity for the development of early seral, species-rich forests that retain biological legacies, such as standing dead trees and lying deadwood, meaning active planning of disturbance-like interventions at the landscape scale should be a central element of modern ecological forestry (Aszalós et al. 2022; see more in Chapter 5.2).

Enhancing deadwood and habitat trees

The most prominent effect of the removal of woody biomass through harvesting is the decline in the number of large old trees as well as standing and lying deadwood in different stages of decay (Stokland et al. 2012). Therefore, the conservation and restoration of near-natural forests, dispersed large habitat trees and deadwood should always be part of the management plan in biodiversity-oriented forestry. Practically, this not only keeps living trees, but also snags, lying deadwood, charred trees, or disturbed forest patches (Gustafsson et al. 2012). Based on meta-analysis of the literature on this topic, it can be concluded that for specialist organisms dependent on deadwood, diversity in the tree species providing deadwood and trees with old-growth features is more critical than their amount, with the amount more important than its spatial arrangement. The diversity of deadwood is formed by both different tree species and by deadwood types in terms of diameter and successional stage. This information opens opportunities for economically optimised deadwood enrichment strategies (Seibold et al. 2016). To foster biodiversity related to microhabitats (see Chapter 2), actively preserving habitat trees during thinning operations, ringbarking trees to create standing deadwood and even actively creating microhabitats (e.g., carving wounds or cavities in living trees) are increasingly used (Gustafsson et al. 2020, Sandström et al. 2019).

Managing species and ecosystem diversity at the landscape scale

To ensure the spatial continuity of all features important for biodiversity, it is also necessary to consider forest management at the landscape scale. Landscapes with forests in Europe range from those heavily dominated by forested areas to those that are predominantly agricultural with only few forest fragments.

In **forest-dominated landscapes** (Figure 7A), forest cover does not impose harsh boundaries beyond which forest-related species cannot travel (Busse et al. 2022, Fahrig 2013, Seibold et al. 2017), although this may not be true for specialist species. Across naturally dynamic landscapes, the variability in abiotic conditions, such as local climate, soils and topographic conditions, is reflected in the existence of heterogeneous and dynamic mosaics of different forest types. It is important to understand this variability in time and space and to maintain or restore it through proper management. In this context, **identifying and preserving specific and very often marginal habitats** within forests, such as water springs, watercourses, wetlands, peatlands, rocky sites and topographically extreme sites with lower tree density, provide excellent opportunities to maintain and promote biodiversity. If the landscape under consideration still contains ancient forest areas (see Chapter 2), soil disturbing management, conversion to conifer monocultures, ploughing, soil scarification, various fertilisation regimes and other activities known to harm a variety of organisms endemic to ancient forests, are to be avoided.

Understanding and emulating natural disturbances is a key aspect of forest management practices promoting biodiversity. These concepts have been well used and documented in the **ecological forestry** approach that seeks to maintain and restore structural complexity in forest stands based on patterns found in stands and landscapes shaped by natural disturbances (Palik and D'Amato 2017). Management practice employing a combination of silviculture systems, harvest rates coupled with appropriate extent, rotation period and return intervals that best emulate the natural disturbance regime would be a major improvement (see also Chapter 4.2). Maintaining sufficient areas of mature and old-growth stands across a landscape and securing their connectivity is also crucial tool for biodiversity promotion as leaving and retaining sufficient amounts of such groups of trees or small stands without management is a way to positively influence the age structure of the landscape (Angelstam et al. 2020).

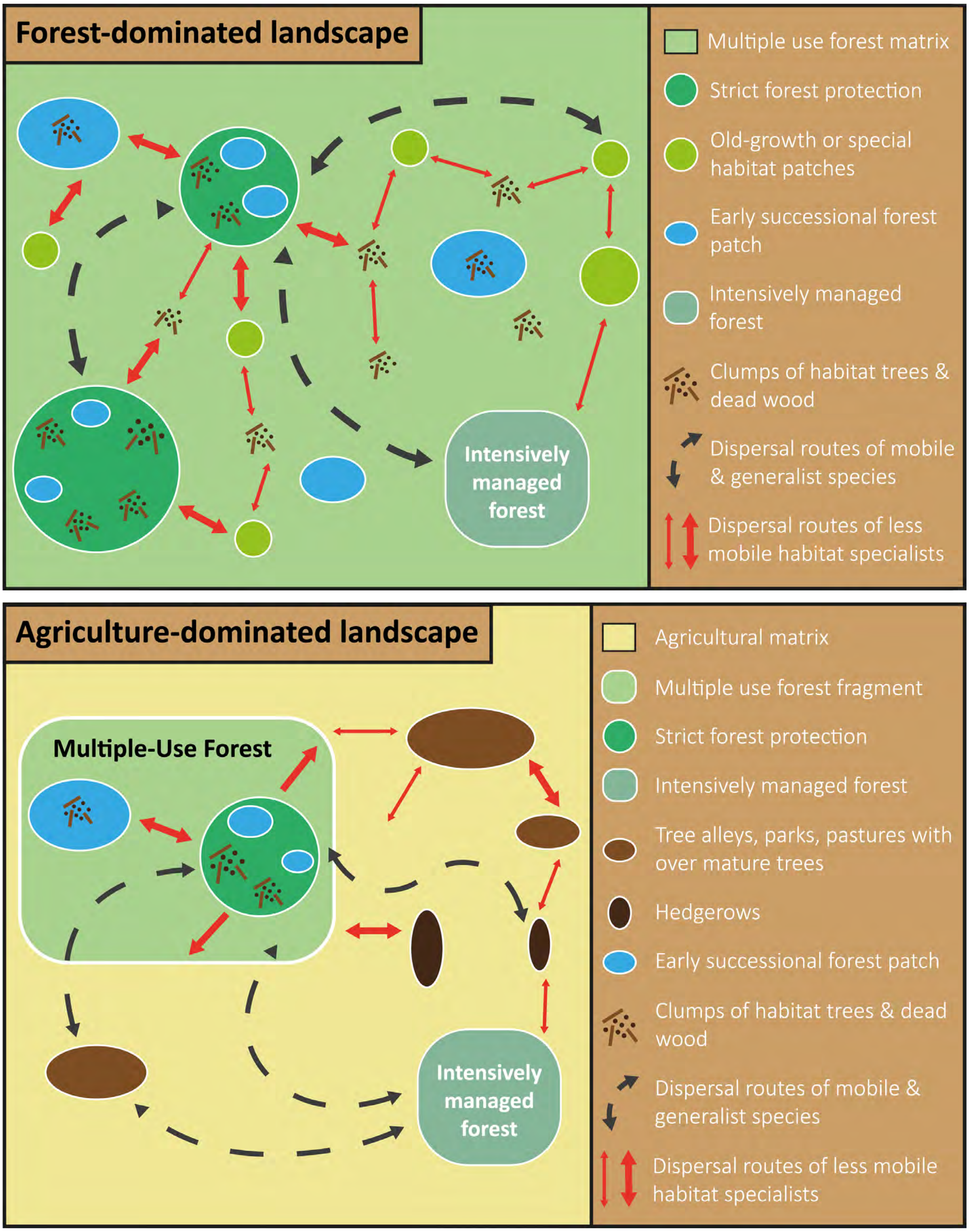


Figure 7. Schematic representation of the integration of biodiversity into forest management at the landscape scale using two abstract European landscapes. A. (upper pane) Forest-dominated landscape; the matrix consists of multiple-use forest with wood production, containing patches of habitat trees and deadwood and also individual habitat trees; small dark green patches represent old-growth stands or special biotopes (1–10 ha); B. (lower pane) Agriculture-dominated landscape; the matrix consists of agricultural land with remnant trees and hedges functioning as stepping stones, but mainly for mobile species dispersal.

To mimic structural heterogeneity found in natural forest landscapes, a variety of management interventions must be applied across different spatial scales (Palik and D'Amato 2017). Silvicultural systems, such as group selection and irregular shelterwood, maintain the **multi-cohort stand structure** of the late-successional stage. However, to maintain heterogeneity at the landscape scale, this is not sufficient by itself and must be combined with silvicultural systems that (i) entail higher canopy removal and (ii) incorporate natural disturbances into the regeneration process. **Spatial heterogeneity in harvesting levels**, in terms of intensity and cutting area, can have a direct bearing on promoting heterogeneity within traditional silviculture systems, particularly when spatially-diversified harvest patterns are used to promote the size and connectivity of habitats.

In **agriculture-dominated landscapes** (Figure 7B), the same guiding principles for forest-dominated landscapes apply within the remaining forest fragments, especially when it comes to abiotic variability, rare biotopes, successional stages and areas with low or zero management. The main difference with forest-dominated landscapes is that the population size of forest-related species is limited to smaller forest fragments and exchanges between populations is impeded by an agricultural matrix unsuitable for forest species (Decocq et al. 2016). In such a context, biodiversity-oriented management should focus on **enlarging the remaining forest fragments and maximising their structural and functional connectivity**. Enlarging existing fragments is the preferable option over creating new ones because, among other things, it protects the fragment's core area from weedy plant invasion (Honnay et al. 2002) and from influxes of the fertilisers and biocides commonly employed in agricultural activities. Enlarged fragments will also support larger populations of limited-mobility species, which is especially important if the fragments are ancient forests (see Chapter 2). Maximising connectivity can be realised by conserving and restoring **tree-based landscape elements**, such as remnant habitat trees, hedgerows and tree-lined thoroughfares and/or by creating new fragments as stepping stones.

An important specific issue regarding biodiversity management at the landscape scale is **game management**. Game populations, especially ungulates such as roe deer, deer, moose and wild boar, have increased dramatically in European forests in recent decades (Angelstam et al. 2017a and 2017b, Carpio et al. 2021). This trend is caused by a complex interplay of factors, including ineffective game management, milder winters, abundant food in agricultural land adjoining their habitats and the lack of natural predators. When population levels are moderate and not overly dense, ungulate herbivory has positive effects on herbaceous plant diversity by mediating the competition between woody and non-wood plants (Bernes et al. 2018, Boulanger et al. 2018, Fløjgaard et al. 2018). However, **booming ungulate populations** means they have become a matter of concern for biodiversity conservation in some regions. Young trees, primarily broadleaved but also some native conifer species, can be heavily damaged through browsing and antler fraying, which has obvious impacts on their survival rates. This has a cascading effect on the species dependent on these trees as they mature, including lichens, fungi, pollinator insects and bird species (Angelstam et al. 2017a; Vázquez and Simberloff 2004) and may limit adaptation options to climate change in affected areas (e.g., Kunz et al. 2018). A range of management approaches is available, ranging from culling animals to protecting plants using barriers such as fences and grow tubes (Carpio et al. 2021).

5.2 Forest management systems supporting biodiversity

Possible measures to support biodiversity from the genetic to the landscape level as outlined in Chapter 5.1 primarily intend to reduce or eliminate negative impacts of forest management on forest structure and composition. The next key question is how they can be implemented in existing forest management systems across Europe. For example, the **retention forestry** approach, which promotes the long-term retention of living and dead trees and small areas of mature forest at the time of harvest, can be implemented in most types of silvicultural systems and forests (Gustafsson et al. 2012). Its application from clear-felling to single-tree selection silvicultural systems supports biodiversity and ecosystem functions at stand and landscape scale by promoting variation in forest structure, composition and complexity. Likewise, the

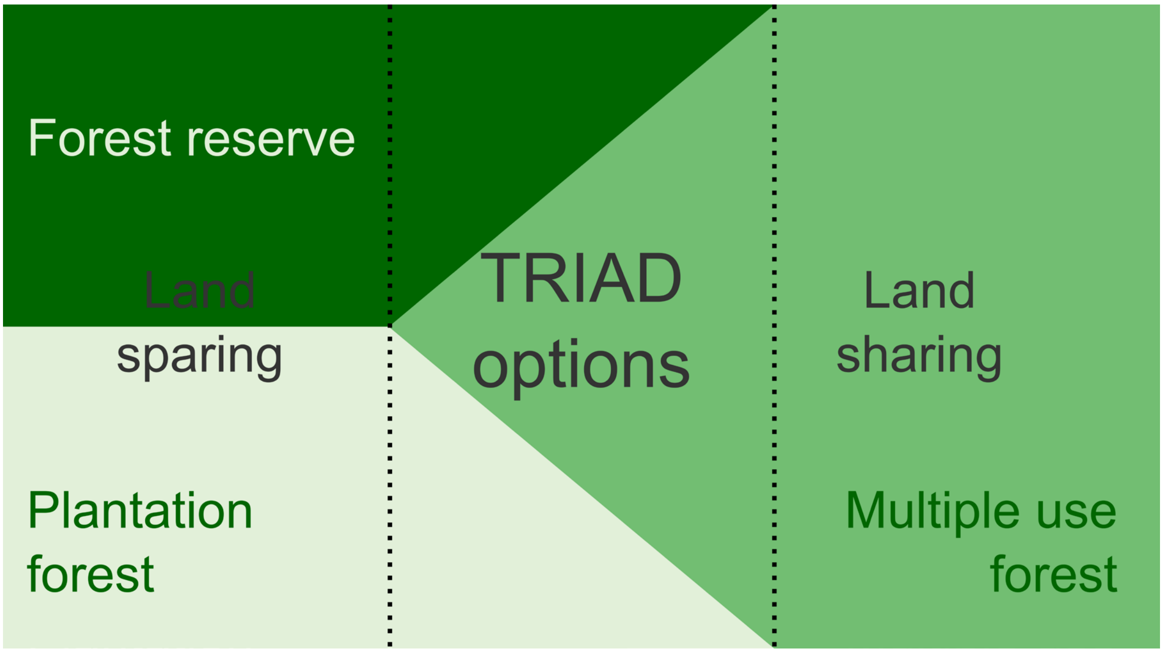


Figure 8. Forest management options at the landscape scale. Along the horizontal axis, the options vary from pure land sparing without multiple-use forest (left) over triad options (centre) to pure land-sharing without forest reserves and plantation forests (right) (modified after Larsen et al. 2022). By way of example, a triad forest landscape could be composed of 30% unmanaged, 60% multiple-use and 10% intensively managed for wood production The range of proportions of the different management options in a triad approach is flexible.

maintenance of **genetic diversity of populations** of trees and other forest-dwelling organisms can be integrated into many different silvicultural systems.

Besides biodiversity conservation, European citizens and governments expect forests to provide other ecosystem services like wood production and cultural services. Different forest management systems vary in the **level of spatial integration** of management for biodiversity with the other goals of forest management (McDonald and Lane 2004). There are different options (Figure 8), which may result in very distinct forest landscapes:

- **Land sparing** is a landscape management approach that involves strict spatial **segregation** of conservation and wood production areas. This includes forest areas with no management or conservation-oriented regime in place and other areas that are intensively managed, such as plantations. This approach is well-suited to preserving sensitive species that can either disappear even with low levels of human disturbance or need large undisturbed areas to flourish (Betts et al. 2021). Under this approach, intensively managed plantation areas can contribute to the conservation of biodiversity, under the condition that the efficiency of timber production from plantations leads to a reduction of timber extraction from the remaining forest areas (Bauhus et al. 2010). Meeting this condition requires strong forest governance to achieve a regional balance of forest management objectives and related systems, which is not easy to realise in forest landscapes with a large proportion of private ownership (Bollmann and Braunisch 2013).

- **Land sharing** is an integrated landscape management approach that promotes the multiple-use of areas for conservation, wood production and other ecosystem services. This approach results from the idea that in forests managed for wood production, which covers the majority of forest land in Europe (Sabatini et al. 2020), the recognised deficits in forest structural diversity and old-growth structural elements can only be addressed by maintaining and restoring these elements in managed area, unless the area of segregated elements (i.e., reserves) is significantly enlarged.

- **Triad management** is a hybrid approach combining land sparing and land sharing. Because both strategies have strengths and weaknesses (Nagel et al. 2017), triad management has been proposed as a best-of-both-worlds system of sustainable management of forest landscapes (Seymour and Hunter 1992, Côté et al. 2010). In triad landscapes, protected areas and intensive forest-use systems account for a part of the landscape, with the remainder occupied by integrated management systems, for example, continuous cover and close-to-nature forest management. The latter provides a suitable matrix for improving connectivity between protected areas and acts as a buffer against intensively managed stands. This approach encourages flexibility to find a compromise between the desire to increase protected area, implement multiple-use forest management systems and simultaneously expand wood production for the bioeconomy. As a starting point, triad schemes for forest landscapes can be optimised based on predetermined wood production goals and conservation targets (Côté et al. 2010).

The functionality of the three elements in triad management, namely strict conservation areas, intensively managed areas and integrated multiple-use areas are further elaborated in the next paragraphs.

Management of high-value protected areas

National parks, forest reserves and other types of protected areas are essential governance structures where forest management measures strongly focused on promoting biodiversity (see 5.1) should be rolled out. Large protected areas allow natural processes such as fires, insect outbreaks and wind fall to occur and run their course on large scales, which is typically impossible in forested landscapes managed for wood production. Therefore, the occurrence of natural disturbances in such locations should not necessarily be followed by sanitary and salvage logging, which removes the newly-created unique habitats. Using landscape-level planning, protected areas and their buffer zones should be designed in a way that permits natural disturbances without having them spill over into surrounding managed forests and vice versa. While smaller protected areas create important biodiversity islands in the matrix of managed forests, potentially hosting source populations of many rare species (see Figure 7), large protected areas allow for the re-introduction and/or maintenance of viable populations of large mammals which are, in the case of herbivores, often important ecosystem engineers. However, the distribution of such reserves at the landscape level requires careful planning to create functional networks of representative habitats.

In many protected areas across Europe, it is common practice to perform sanitary and salvage logging, especially after a disturbance. While natural disturbances have positive effects on species diversity, negative effects of salvage and sanitary logging on species diversity have been well documented (Thorn et al. 2018). This contrast between available evidence and common practices underlines the need for greater acceptance of natural disturbances in the management of the Natura 2000 network.

Existing primary and old-growth forests in Europe are of paramount importance for conserving biodiversity. Despite their relatively small share of Europe's total forest cover, entire regions with relatively large areas of primary forests still persist. Surprisingly, there are significant knowledge gaps about the location, extent and current status of many remaining primary and old-growth stands (Sabatini et al. 2020). Since only half of the known primary and old-growth forests are strictly protected, and a significant number are threatened by timber harvesting, immediate action at the national and EU levels is required to prevent the loss of these critically important forests.

Close-to-nature and other forms of integrative forest management

In a context of strong fragmentation and widespread but small-scale private ownership, conservation measures in European forests, both inside and outside of the Natura 2000 area, are often integrated into forest management at the stand scale. This integration requires a reduction in forest harvesting levels to retain habitat trees, build up desired deadwood stocks and set aside patches of unharvested forest with high conservation value. Integrative forest management approaches have been advocated to support biodiversity conservation, while balancing with the many other demands placed on forests (Krumm et al. 2020).

A widespread integrative approach is **close-to-nature forest management** (Pommerening and Murphy 2004). It has traditionally included many elements of conservation management discussed under Chapter 5.1, such as maintaining tree species and structurally diverse forest stands, a preference for the presence of site-adapted native tree species, a reliance on non-anthropogenic processes such as natural regeneration and ungulate-population control as well as using long production cycles (Bauhus et al. 2013). After the World IUFRO Congress in Ljubljana (1986) the international **Pro Silva** movement was founded and further spread these ideas among forest managers. In recent decades, the close-to-nature forest management approach has been widened to incorporate the concepts and measures of retention forestry (Gustafsson et al. 2020), which are better suited for lowland and boreal forests without many shade-tolerant tree species.

From the perspective of biodiversity conservation, some issues with the close-to-nature forest management approach have been identified: a lack of retained forest structures, a high frequency of harvesting and a lack of consideration of landscape-level forest heterogeneity (e.g., Schall et al. 2018, Bässler et al. 2014, Gossner et al. 2013). There are many ways to address these deficits using flexible and regionally adapted approaches of **integrative forest management** (e.g., Krumm et al. 2020). Higher landscape-level structural variation can be achieved through applying a wider range of management intensities as well as retention and maintenance (or reactivation) of historical forest management systems, such as **coppice** and coppice with standards. These concepts have been embraced by the **ecological forestry** approach that seeks to maintain and restore structural complexity in forest stands based on patterns to be found in stands and landscapes shaped by natural disturbances (Palik and D'Amato 2017). Owing to the lack of landscape-scale management in many of Europe's forested regions, for the reasons outlined above, these ideas have not yet been widely incorporated into forest management (however, see Roth et al. 2019).

Management of cultural landscapes

The origin of Europe's cultural landscapes and their biodiversity was described in Chapter 2 where it was noted that historically, forest land use included a wide variety of techniques such as coppicing with or without standards, low-intensity forest grazing, low-intensity burning and charcoal production, partial tree biomass removal and others. These types of historical land-uses were often intertwined with similarly diverse agricultural and agroforestry practices that created areas of fine-grained and heterogeneous mosaics of open and closed habitats, mimicking a higher variety of succession stages and with a significant presence of habitat trees. From a modern forestry point of view, these are low-value forests with low tree density, low canopy closure and biomass as well as low-quality timber. For this reason, they have been converted into high forest of often conifers, or are abandoned and colonised by pioneer trees. However, from a biodiversity point of view, these are highly valuable forest landscapes as they host many species that have become rare and increasingly endangered. In some regions of Europe, these cultural landscapes and their biodiversity are still largely intact or have returned to traditional management techniques as part of their local conservation efforts. Often, traditional management techniques including coppicing or pollarding of trees and mowing or livestock grazing of ground vegetation are paramount to keep on interrupting the forest succession, keep the canopy cover low and maintain the high cover and diversity of forbs in these systems (Baeten et al. 2009, Douda et al. 2017). In general, when it comes to effectively managing forests in a cultural landscape, involving actors that actively search for win-wins to manage the trade-offs is seemingly the most viable approach to keep such forests economically profitable while simultaneously safeguarding the biodiversity found in these landscapes. Policy and subsidy adjustments through the European CAP and Farm to Fork programmes have immense potential to support biodiversity conservation efforts in these landscapes (Varela et al. 2018).

Contributions of plantation forests to biodiversity conservation

Plantation forests are typically characterised by even-aged stands with one dominant and fast-growing tree species which, in many cases, is of exotic or regionally non-native origin. They are usually managed in rather short rotation cycles with regeneration after clear-felling that leaves virtually no living or dead biomass in situ between production cycles. As a result, this practice supports little biodiversity typical for natural forests and compares poorly to less intensively managed forests (López-Bedoya et al. 2021). However, plantations are sometimes far from being biological deserts (Stephens and Wagner 2007) and can also contribute to the preservation of the biodiversity in neighbouring natural forests in shared landscapes (Brockerhoff et al. 2008) by realising more wood provisioning on less land (Freer-Smith et al. 2019). The establishment of new plantations may induce either a loss or a gain in biodiversity, depending on the intensity and conservation value of the previous land use. When established in former intensive agricultural land, a gain in biodiversity can be expected because the reduced pesticide use and soil disturbance, as well as the created forest cover promote the establishment of understory vegetation, providing new habitats for forest fauna. In contrast, replacing natural forests or native grasslands with plantations will lead to a net biodiversity loss. In addition, there are a range of **management alternatives to improve biodiversity within forest plantations** (e.g., Hartley 2002, Flaspohler et al. 2011) while maintaining their commercial efficacy. These include a preference for native tree species, the use of species mixtures (Messier et al. 2021), avoiding or reducing tillage and weeding, retention of logging residues as well as the protection of the understory during thinning and harvesting while also retaining some of the structural elements (Bauhus and Schmerbeck 2010).

At the landscape scale, plantations can provide surrogate habitats to forest species, although these will primarily be generalist species. They can also improve connectivity between forest fragments in a non-forest matrix by functioning as stepping stones (Brockerhoff et al. 2008). Importantly, plantation forests should not be composed of invasive species or harbour pests that spread to nearby natural forests. Consistent with the segregation approach, landscape-level measures, such as limiting the size of planted stands and precluding the spread of harvesting operations over time, can moderate possible negative effects (Flaspohler et al. 2011). The potentially most important contribution to biodiversity conservation of productive plantations is their concentration of wood production into a small area, which can facilitate the reduction of harvesting pressure on other forests or even in the set-aside of reserves in other parts of the landscape through land sparing (Bauhus et al. 2010). Increasing the land area used for plantation forestry could thus facilitate an increase in the size and/or the number of strict conservation areas in segregation and triad approaches (Figure 8). Hence, future afforestation efforts in Europe may benefit biodiversity through restoring valuable habitats on the one hand and providing highly productive forests that compensate for biodiversity conservation-related reductions in harvesting intensity of other forests on the other. However, this requires careful landscape planning across multiple scales.

In conclusion, finding a well-selected combination of different management and non-management approaches, well-integrated at the landscape level into joint land sharing and land sparing approaches, that are well-adapted to local legacies and potential is the key challenge. In this regard, there are opportunities for the new **closer-to-nature forest management** concept proposed in the new EU Forest Strategy (EC 2021) that are yet to be fully seized upon. A range of principles for the effective deployment of closer-to-nature forest management have already been proposed in Larsen et al. (2022).

6. Forest biodiversity policies and incentives

Chapter highlights

Policies should always be evaluated considering their (potential) impact, effectiveness, efficiency and legitimacy as well as the coherence of the policy mix.

A mix of tailor-made financial and other instruments needs to be expanded to successfully secure Europe's exceptional biodiversity heritage. Market-driven instruments, such as reverse auctions and biodiversity offsets are still underdeveloped and require further exploration and discussion.

Multi-sectorial collaboration is needed for efficient conservation and use of biodiversity in natural and cultural landscapes, combining agriculture, forestry, and human-cultural heritage.

This chapter describes the logic of different policy instruments and governance mechanisms that are aimed at steering and supporting the conservation of forest biodiversity and shows how context and history-dependent policy is, as well as how different policy instruments function together and influence each other. The chapter starts by describing how policy is often justified and designed to generate positive impacts and then draws attention to the context and history that condition new policies, including the influence exerted by stakeholder interests and conflicts. The chapter ends with a description of different instrument types available to policymakers, including regulations, economic instruments, information, collaborative governance and sustainable financing.

6.1 The role of public policy

Biodiversity underpins the provision of ecosystem services and their resilience, which also has other values for society, as described in Chapters 1 and 2. The need for effective policy in this area arises from biodiversity loss not generating obvious direct costs or providing direct feedback to landowners and managers. For these reasons, the responsibility and costs of conservation fall on society in general and, consequently, biodiversity conservation is primarily steered by public policy.

Broadly speaking, **public policy steers societal activities by setting goals and identifying means**, and it is designed in government-led processes (Howlett and Cashore 2014). Public policy relating to forest biodiversity conservation seeks to reduce threats to biodiversity and improve the conservation status of forest ecosystems by steering the actions of forest owners and managers while supporting the structures that they rely on. Because biodiversity and forest ecosystem functions depend on and constitute complex ecological processes, even at the landscape level, conservation efforts need to pay attention to dynamics that do not fall within the nominal administrative and legal boundaries of individual properties, municipalities and countries.

The success of a public policy can be evaluated in varied ways, such as by analysing processes as well as the activities and impacts on the ground, that is, whether the inputs are generating the desired outputs (Mickwitz 2003). In biodiversity policy, as with environmental policy in general, the emphasis is often on the (expected) ecological impacts and the (expected) effectiveness of policies in achieving those results (Turnhout et al. 2016). In other words, much attention is paid to how effectively biodiversity is conserved when measured using certain criteria. Despite the importance of the ecological effectiveness of policy and specific policy instruments, it is also important to consider other aspects, such as social and economic impacts as well as the institutional feasibility of a given policy. All of these can condition the impacts and effectiveness of policy by influencing the criteria used to assess it.

Historically, the focus on conserving forest biodiversity has been pursued with both goods (fuel, food and feed) and protective functions regarding wilderness, heritage and aesthetic aspects. Today, biodiversity conservation is also driven by existence and bequest values, charismatic species as well as areas and organisms that are rare, unique or endangered. The increasing intensity of forest use has resulted in the ascendance of segregated approaches, as described in Chapters 2 and 5, in Europe's efforts to conserve biodiversity. Unfortunately, this approach has resulted in the segregation of administrative efforts on this issue as well. Protection and protected areas are governed by environmental administration but because they can fall under the auspices of one or more policies, the specifics of how such policy is applied can vary. When an administration's focus is on ecosystem services, the multiple functions of forest ecosystems or specific ecosystem features, such as old trees or structural diversity, forest biodiversity conservation is chiefly governed and pursued through forest policy. This integrative approach is often framed as multi-functional forest management or sustainable forest management and its implementation is integrated into national forest law and planning systems steered with targeted instruments that are described in detail later in this chapter. A key notion to keep in mind at this point is that the triad approach that makes use of the range of preservation and sustainable, multifunctional forest management, as described in Chapter 5, is constrained by the fragmented ownership structure in many European countries.

6.2. Public good characteristics and externalities justify policy

The benefits derived from biodiverse forest ecosystems are numerous and many of them can be enjoyed beyond administrative boundaries. While landowners cannot exclude anyone from those benefits that are public goods, they often hold rights to influence the structure and function of the forest ecosystem affecting biodiversity. **The trade-offs between landowners' property rights and society's conservation interests are a cornerstone of what forest biodiversity policies seek to address.**

In economic terms, **biodiversity loss is a negative externality** resulting from degradation caused by harvesting, less than ideal management and land-use change. As previously stated, there is often no direct cost feedback to the agents driving degradation, except if degradation reduces aspects such as productivity or resilience in observable ways. The feedback to the consumers of the marketed forest goods is even weaker. Indeed, the negative consequences of degradation are experienced by others, which is why this is considered an external cost. This is the core economic argument for regulating forest management, to better account for the overall true value of biodiversity (Krutilla 1967). In a similar vein, but going in the other direction, increased biodiversity resulting from conservation-sensitive forest management would be a positive externality.

The commodification approach to forests has emphasised the easily measurable economic benefits of their exploitation but resulted in the inherent value of the ecosystem itself, along with its numerous non-economic benefits, simply being ignored (Sagoff 2008; Gómez-Baggethun 2010). Method developments in environmental valuation have allowed us to assign economic value to several ecosystem services, but not all (Dasgupta 2021), and the integration of values into decision-making remains a practical challenge (Primmer et al. 2018). Thus, the problem of unsatisfactorily addressed externalities remains. Having said that, a balancing proviso should be kept in mind regarding the problem of externalities. While often key actors in biodiversity loss, some landowners protect biodiversity without compensation for their economic losses (Primmer et al. 2014), even though their objectives and opportunity costs of providing this protection differ (Vedel et al. 2015).

Conservation measures are often carried out or compensated for by the public sector, whereas environmental regulations prescribing forest management restrictions can generate costs that are incurred solely by landowners. This has only recently begun to change with new initiatives entailing payments for ecosystem services and ecological compensations placing the costs of conservation on private-sector actors. As many threats to forest biodiversity come from forestry, any policy that hopes to be even marginally effective must address these threats. With property rights generally providing broad leeway regarding forest use and management, any new policies that moderate these rights will need to be well justified given the sensitivity of the topic.

6.3 Policy depends on context and history

Regional and local differentiation

European forests vary in their ecological features, protection, management histories, economic use, ownership structure and governance. Forest structure and biodiversity, as well as **approaches to conservation also vary across the continent**. Some countries, such as Finland and Sweden, have put more emphasis on land sparing approaches with relatively stricter protection and no interventions in protected areas, while other countries, such as Germany and Austria, have pursued conservation by integrating it into regular forest management. Further to this, in densely inhabited countries and countries with a long history of land-use change, for example, Denmark, the Netherlands, Italy and Spain, much of the emphasis on biodiversity conservation involves land sharing that includes the restoration and maintenance of traditionally managed landscapes. The third group of European countries have clearly opted for mixed approaches combining strict preservation with restoration and integrated conservation. In Romania, for example, strict nature protection of the last remaining primary forests is complemented with multi-functional forestry in managed areas that pursues forest regeneration using local species, small clear-cut areas and rotation lengths over 120–140 years.

The economic importance of forests and forestry is concentrated in rural areas and is, therefore, also socially important. Some European countries have a strong forest sector, for example, close to 20% of Finland's export income comes from its forest industry, while in many other countries, their forest industries are purely regional or only locally significant. While many European countries have rural areas that still use wood for fuel and others use wood for bioenergy that may play a significant role at an industrial scale, the relative economic importance of the forest sector across the continent is shrinking when measured in terms of its share of GDP. In addition, employment in the forest sector has decreased by approximately 33% in Europe since 2000 (Forest Europe 2020). Nevertheless, forests remain important in many rural areas, albeit in a socially and culturally important capacity, where their key economic relevance is increasing because of their real estate value (Angelstam et al. 2022). The broader implications of all of this for biodiversity conservation remain unclear.

Expectations for a broadening income base for forestry, such as by establishing new value chains that support rural areas, are now centred on the bioeconomy (Kleinschmit et al. 2014), social innovation (Nijnik et al. 2019), governing innovations for ecosystem service provision (Primmer et al. 2021) and a circular economy (D'Amato and Korhonen 2021). **New income generation opportunities from biodiversity conservation are expected at the policy level,** such as in the latest EU Forest Strategy (2021).

Land ownership and regulation

In the EU, approximately 60% of the forest area is privately owned, by non-industrial actors that are either individuals, associations, forest communities, religious organisations or small-scale local enterprises (Weiss et al. 2021). Although there is great variation in the size of privately-held forest lands, from larger private estates in Northern Europe, where some 70% of forests are privately owned, to much smaller private estates in South-East Europe, where around 90% of the forests are public (Pulla et al. 2013; Weiss et al. 2019), almost 90% of Europe's 16 million private forest owners hold less than 10 hectares each. **This diversity in forest ownership can be a challenge for policy design and implementation** but could also result in a mosaic of management approaches that could ideally support biodiversity conservation at the landscape level (Mölder et al. 2021), particularly if integrated or triad approaches are employed.

For decades, forest owners' rights and responsibilities have mostly been regulated through forest legislation and policies, with a primary focus on sustaining high yields so that, for example, growth exceeds harvest. Following the increasing recognition of the need to conserve biodiversity as a whole rather than just trees, environmental regulation is having increasing impacts on property rights (Nichiforel et al. 2020).

Interests and stakeholders

Biodiversity conservation is often positioned in opposition to the economic uses of forests that are based on maximum sustained yields. Its claims are considered to bypass the claims from rural livelihoods, large numbers of private forest owners and the economically significant forest industries operating in parts of Europe (Edwards and Kleinschmit 2013; Winkel and Sotirov 2016). The conflict is not novel in Europe as can be seen, for example, with the establishment of the Natura 2000 network that has resulted in heated debates and conflicts in many EU countries (Niemelä et al. 2005; Blondet et al. 2017), a pattern that echoed similar preceding conflicts (Hellström 2001). This frequent positioning of wood production and biodiversity conservation as opposing and incompatible goals has politicised biodiversity conservation, particularly when EU-level forest biodiversity policies are being formulated.

Governance architecture at national and EU levels

Overall, forestry and forest management in the EU are governed under national mandates and, in some European countries, also under regional mandates. In addition, many European countries have national forest strategies that vary in their emphases on forest functions (Primmer et al. 2021). Since the beginning of the 1990s, European countries have formulated forest- and biodiversity-related policy goals and voluntary policy instruments in the context of the pan-European Forest Europe process and different certification schemes.

The EU holds no forestry competence *per se* but formulates forest-related policy instruments largely based on its competences in agriculture, environment and trade. To coordinate these forest-related policies more tightly, the European Commission has compiled a portfolio of **forest strategies and action plans**: While the first EU Forest Strategy primarily summarised EU forest-related policies, the second strategy, produced in 2013, provided guidance to enhance EU-level coordination of forest-related policies (Aggestam and Pülzl 2018, Wolfslehner et al. 2020) and took a more holistic view of forest ecosystems and their related ecosystem services than other EU-level policies at the time (Bouwma et al. 2018). The latest EU Forest Strategy (EC 2021) has put a decided emphasis on biodiversity conservation with protected area targets and closer-to-nature forest management ideas (Larsen et al. 2022).

Environmental protection falls under the category of being a shared EU competence. This means that the European Commission can establish legal instruments for biodiversity conservation to meet international commitments, among other things, but these need to be implemented by Member States. The central instruments in this regard to date are two EU Directives (the Birds Directive and the Habitats Directive) and the EU Biodiversity Strategy which has a strong overall-strategy emphasis at the EU level. **This governance architecture targets biodiversity preservation by protecting threatened species and habitats but its implementation in many Member States also allows for its integration into forest management and wood harvesting practices.** Furthermore, the latest EU Biodiversity Strategy (EC 2020) embeds protected areas at the heart of conservation activities and mandates protecting the EU's remaining primary and old-growth forests as well as better managing protected areas to secure their biodiversity values.

Biodiversity conservation is now systematically recognised alongside wood production and other functions in national forest strategies (Primmer et al. 2021), as evidenced by some Member States' forest laws now including goals for biodiversity conservation. National biodiversity governance typically builds on nature protection legislation, implementing the EU Directives, and a national biodiversity strategy that addresses biodiversity conservation even beyond protected areas. In general, formal biodiversity conservation policies emphasise protection and preservation, reflecting a segregated approach to conservation and management.

6.4 Public policy instruments, private initiatives and instrument mixes

The societal goal of conserving biodiversity has resulted in numerous policy instruments that go beyond the pale of formal biodiversity conservation governance architecture and regulation. Steering with regulations is complemented with, and connected to, other mechanisms that trigger behavioural changes and support collective action to promote biodiversity. For example, payments through agro-environment schemes can make forest owners and managers become ecosystem service providers and encourage them to consider biodiversity-friendly ways of managing their forests. Importantly, knowledge exchange and collaboration have been shown to support commitment and allow co-learning as well as generate new experimental and innovative ways of enhancing biodiversity conservation (Rauschmeyer et al. 2009; Sterling et al. 2017). There are also several different voluntary private initiatives that are complementary to or align with public policies.

In practice, policy instruments are always applied together and are designed to support each other. In economic policy design, it is commonplace to suggest that every distinct goal requires its own policy instrument as very rarely are multiple diverse goals sufficiently met by one instrument. As such, at the design phase when a new regulation is being proposed, attention is often focused only on this one specific instrument, ignoring the supporting information and network instruments as well as the payments and transfers that come along with that new law. Similarly, newly established payments are generally based on specific laws and information instruments also support economic instruments, at least in their implementation and monitoring. **An effective policy instrument mix is often the result of policies being successively developed** to supplement existing policy-instrument mixes, bringing some nuanced improvement to address changes in real-world circumstances.

In the following sections, we describe the policy instruments that have been deployed to govern forest biodiversity conservation. We outline aspects of each policy instrument, the ways in which they are assumed to generate impacts and what enables and constrains such impacts. Some of these impacts are reasonably easy to evaluate, however, many policies, especially forest-related policies, are evaluated based on the perceived success of their implementation activities because their impacts on forest ecosystems require considerable time to become manifest, which can provide an inaccurate assessment of a policy's efficacy.

Legal instruments

Regulations require society and landowners to comply with a set of standards and practices while also allowing or requiring certain monitoring and enforcement procedures to be established. Regulations primarily impact organisations, structures, agencies and procedures but can also empower a public authority to ensure compliance and evaluate whether the guiding regulations are effective. Such evaluations are often conducted when planning changes in regulations that are often required due to new knowledge or altered societal demands making such changes prudent.

Biodiversity conservation via regulations can be and is done in many ways, ranging from politically-processed protected area programmes to more targeted and smaller-scale authority decisions. Practices depend on context, national forest management rules as well as on landowners' property rights, as discussed earlier in this chapter.

Regulation is always an 'entitlement'. When established, regulations may allow for a variety of interventions into forest use and can heavily favour producing timber, conserving biodiversity or fall somewhere on a spectrum between these extremes. When enhancing biodiversity conservation, a so-called 'property rule' is applied, entitling a landowner to claim compensation from the public budget for undertaking those actions required to maintain biodiversity. Alternatively, if a liability rule is applied, a landowner or manager is deemed as being responsible for biodiversity conservation and carries the associated costs of doing so according to the 'polluter pays principle'. This reflects the fact that regulation, and the issues it seeks to address, are always costly.

As a policy instrument, regulation in the forest sector may, in theory, be the tool most certain to bring about positive biodiversity-conservation impacts. In practice, both the efficiency and effectiveness of such regulations are conditional on information for targeting, well-functioning and funded enforcement mechanisms as well as the processes for managing their social legitimacy. The preparation and implementation of regulations should also deal with the associated uncertainties and risks for key actors and incorporate a background formulation process based on interest negotiations. This would help address some of the dangers of poorly construed regulations where, for example, protecting a specific area results in leakage, namely, a transfer of harvests to other areas that also have biodiversity value or, arguably worse, a new regulation can result in pushback or even non-compliance.

A challenge with regulation-based conservation is providing it with political and societal legitimacy. In contexts where landowners' property rights have a strong status, new conservation initiatives have raised opposition and resulted in conflicts. In highly restricted property rights contexts, such conflicts seem to be less evident, however, failures in implementation still occur (Bouriaud and Marzano 2014; Knorn et al. 2012; Knorn et al. 2013).

Financial instruments

Biodiversity conservation is often governed by payments and subsidies. Several EU countries have specific forest biodiversity payment schemes that rely on voluntary conservation contracts (Miljand et al. 2021). In addition, through the implementation of the CAP and some national programmes, for example, in Finland, Denmark and Belgium, subsidies and payment schemes are used to support and encourage landowners to take measures above and beyond what regulation requires (e.g., in Natura 2000 areas or other sites of interest).

Due to World Trade Organization (WTO) compliant EU regulations, public subsidies and payments should be set to compensate no more than the likely cost of the measures to landowners in terms of direct costs plus foregone net income at relevant discount rates (Jacobsen et al. 2013). In practice, this reduces the potential for increasing market-like features in government payments, often resulting in flat-rate payment schemes with limited efficiency. Difficulty in changing practices can also contribute to authorities seeking to treat landowners similarly, paying them similar amounts irrespective of their contribution to biodiversity and conservation efforts (Primmer et al. 2014). This can reduce the attractiveness of the payment scheme for some landowners and result in its ineffectiveness.

Sometimes, the design of payment-type policy instruments ignores the **motivational drivers of landowners** as some are reaping benefits that are not captured by the payment scheme which, if these benefits were captured, would result in them needing a smaller payment for their conservation activities. Importantly, landowners have shown a particular **willingness to accept payments** when they feel in control of the process and contract (Miljand et al. 2021). Furthermore, if an unexpectedly large number of landowners accept payments or subsidies and the associated management requirements, a scheme's annual budget could be prematurely exhausted. In such a scenario, this is both problematic and may indicate that the policy may be effective but not necessarily efficient, as a sufficient number of landowners may have accepted the measures for lower subsidy and payment amounts (Ferraro 2008).

Payments are often evaluated and justified based on their efficiency rather than their effectiveness or impact. The impacts of payments depend much on the design of the payment scheme and payments are generally based on expected impacts or even just the desired activity, for example, the restoration of or refraining from harvesting a parcel of forested land. As subsidies and payments are conditional on the required activity being implemented over time, some kind of monitoring system is usually put in place to follow the impacts of the scheme. Sanctions, in the form of repayment plus penalty interests or fees, are also sometimes employed and necessary in settings where local norms clash with a scheme's terms and conditions (Engel et al. 2008).

Market-based instruments

Reverse auctions, or conservation tenders, are instruments that allow a buyer (often a government or city, although sometimes a developer) to efficiently use a market-like setting in which landowners compete for contracts on biodiversity conservation by setting a price for specific actions. There are examples of tenders with different designs, particularly in the US and Australia, but they remain largely underemployed in Europe. Evaluations show that tenders may enhance cost-effectiveness but transaction costs can be high and these instruments also differ from the regular practices of the implementing authorities (Latacz-Lohman and Schilizzi 2005; Rolfe et al. 2017). In the EU, contract-based measures for biodiversity conservation were considered appropriate instruments for biodiversity protection in farmland if they primarily targeted the maintenance of threatened habitats and species (Herzon et al. 2018) and there are only a few such examples implemented in forest conservation (Demant et al. 2020).

Biodiversity offsets have been applied in various European contexts, however, they are still considered an emerging instrument (Wende et al. 2018). **Offsets are a way for the biodiversity degrading actor to compensate for the degradation caused by paying for the conservation or restoration activities** of others, a process that often operates in a market-like setting relying on brokers or banks to reduce costs and improve coordination and where the offset provider could be another forest owner (Bull et al. 2013). In forest biodiversity terms, the damaging actor with a compensatory responsibility and who thus needs to buy the offset is easiest to identify in situations where a forest is converted to other land use, such as for residential or mining purposes. For forestry actors participating in multifunctional forest management, the compensation and offsetting roles remain unclear because of the difficulty in measuring biodiversity loss and gain as well as in allocating the changes in rights and responsibilities between the harvesting, silviculture and nature management entities. Offsets have generally developed as a result of **legal compensation responsibilities** and follow a so-called **mitigation hierarchy**, seeking to incentivise actors to avoid or minimise damage to biodiversity by enforcing compensation and restoration to ensure no net loss of biodiversity occurs (McKenney and Kiesecker 2010). This is an interesting point to note as Natura 2000 has a compensation mechanism built into it.

Because of the context-dependency and complexity of biodiversity, **no-net-loss metrics** are hard to establish and their conservation effectiveness has been criticised (Bull and Strange 2018). There are also examples of offsets happening entirely voluntarily, as is the case with companies wanting to advance their reputation by appearing to be more socially and ecologically responsible, or citizens wanting to pay for their footprint, often copying the ideas from voluntary carbon offsetting. **The net impact of voluntary offsetting is hard to measure but seemingly remains a marginal positive to date.**

Information instruments

Information and planning have always played a central role in forest biodiversity conservation. While regulations and economic instruments are backed up with guidelines and extension services, local extension and advice services can turn information into real-world practice. Experience drawn from the Natura 2000 implementation has shown how **communication and information play a crucial role in improving the acceptance of conservation measures among landowners and reducing local conflicts**. However, it is often the information and planning stage that triggers debates at levels ranging from local through to the EU (Young et al. 2010; Edwards and Kleinschmit 2013).

The idea that information, evidence and learning may support biodiversity conservation has been repeatedly addressed in the literature on integrated landscape-level approaches, including adaptive management, conservation planning and many applications of ecosystem services analyses. At the same time, implementation analyses show that integration of different knowledge bases and access to local knowledge require significant simplification of measurement standards as well as engagement and acknowledgement of the framings of local stakeholders (Primmer et al. 2015; Saarikoski et al. 2018; Hodgson et al. 2019). In other words, information and advice alone do not generate impacts, but they are essential for supporting the implementation of other policy instruments.

Network governance

Forest biodiversity policy and its implementation are often supported by collaborative networks of actors that take many forms, from government-driven policy networks to project and operational networks

(Primmer 2011). As planning and regulation are considered centrally driven and, therefore, distanced from local social and ecological settings, local networks and conservation initiatives often emerge as counter-movements. For example, the Finnish Metso Programme employs voluntary conservation contracting and emerged in the aftermath of the Natura 2000 programme. Indeed, networks can generate new ideas and practices relevant to biodiversity conservation and their positive role is broadly recognised in both European and national forest and biodiversity strategies (Primmer et al. 2021).

Network governance simultaneously challenges and complements government-led policy initiatives by sharing resources, knowledge and learning while also increasing accountability (Rhodes 2007). Network governance is meaningful in biodiversity conservation and in complex environmental policy in general where many governance levels and actors have active roles (Jordan 1999; Loft et al. 2015). Although networks can take conservation initiatives and generate biodiversity impact on the ground, they are mostly valued for their capacity to engage with and support learning and commitment.

Market governance

Public concerns for biodiversity and environmental protection have induced different interest organisations and major market actors to seek to drive change through voluntary market measures and product differentiation. The most visible and prominent of these are forest management **certification standards** such as FSC and PEFC, which are open for enrolment from any owner (public or private) willing to adapt their forest management to the certification schemes' standards. Currently, about 52% of EU forests are certified, among which 80 million hectares are PEFC certified, and 52 million hectares are FSC certified, which includes different measures for biodiversity protection and enhancement (Forest Europe 2020). This certified area continues to grow, but the variation in adoption across countries and areas remains large.

From the forest owner's perspective, motivations include market access and potential price premiums on roundwood. While the first is easy to trace because there are large buyers that do not buy non-certified wood, evidence suggests that price premiums remain limited, likely in the low single digit percentages. This is despite consumer surveys suggesting a significant willingness to pay a premium for certified products, and the absence of high premiums in the market suggests that owners joining certification schemes do not change their practices in ways that would force prices up.

Evaluating the impacts of certification on forest management is methodologically challenging because of the observational nature of the data available. Recent major reviews suggest that evidence of positive effects is both scant and weak (e.g., Cubbage and Sills 2020). With little change in practices, certification generates little biodiversity impact in the EU, although when considered globally, certification can be meaningful in controlling the worst of the ecological impacts of certified timber value chains.

Sustainable finance

There are calls for the finance sector to address biodiversity conservation through standards and specific instruments. Examples include investors pushing private and public pension funds to shift to green investments and private companies investing in nature restoration and protection projects (Ferraro and Kiss 2002; Löfqvist and Ghazoul 2019). Sustainable finance has received growing attention in the last decade and it is increasingly called for in both UN and EU policies. At the same time, many companies, especially those for whom ecological issues are a sensitive issue, have been seeking to raise funds for their green investments in, for example, emission-reducing infrastructure. Funds for such investments may be raised by issuing dedicated **green bonds** that may help to reduce borrowing costs and include conditions for **Corporate Social Responsibility** (CSR) and **Environmental, Social and Governance** (ESG) actions. Another specific example can be found in the **insurance** industry that could influence their landowner customers with sustainable management requirements or invest back in restoration that increases ecosystem resilience (Paavola and Primmer 2019).

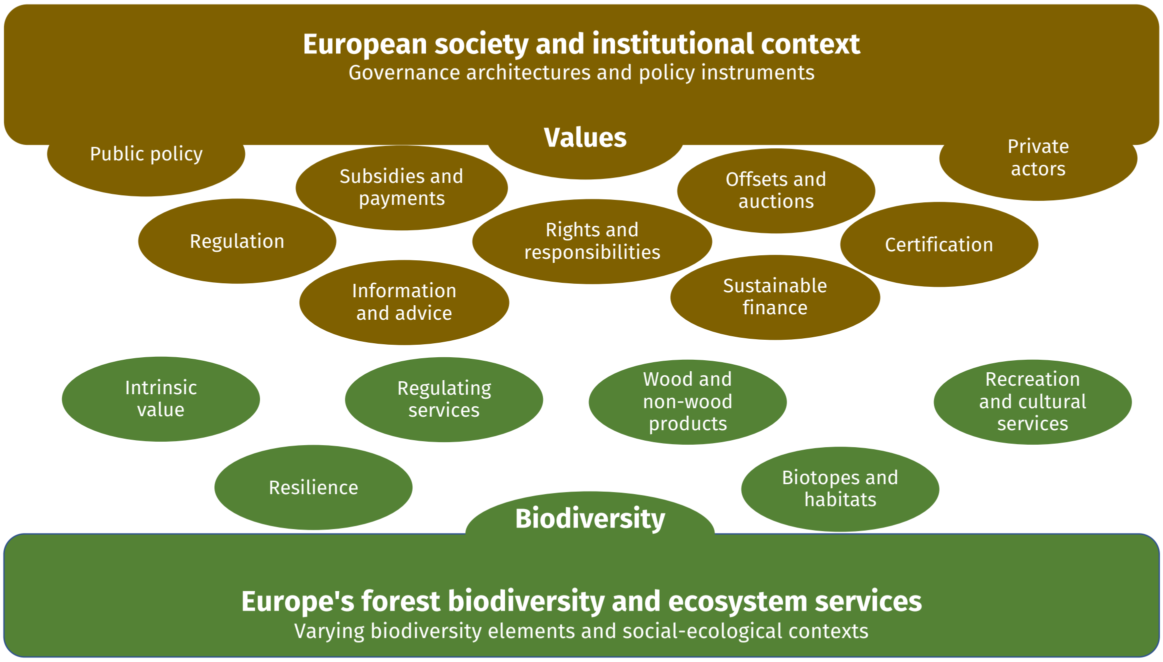


Figure 9. European forest biodiversity governance as an iterative multiscale process of fitting governance architecture and policy instruments (top) to the protection of biodiversity and the optimisation of the ecosystem services it supports (bottom).

Despite increasing expectations that the finance sector will start to invest in ecosystems and biodiversity, public funding still dominates such investment globally (OECD 2020). There are examples of **public-private partnerships** pursuing sustainable investments as many countries now require private foundations to give away part of their earnings for philanthropic purposes (Reich 2018). The EU has developed sustainable financing principles to be applied across the public and private domains under the so-called '**taxonomy for sustainable finance'**. The taxonomy criteria will define sustainability advancing and harming activities while seeking to influence investments and stimulate new finance sector initiatives that align with the criteria while supporting biodiversity conservation and nature restoration efforts. It is expected that this taxonomy will develop, and perhaps even tighten, its standards going forward to match the increasing ambition for sustainability.

Policy design should make full use of this entire **mix of policy instruments**, always adjusting to specific contexts and backgrounds as circumstances and local contexts require (Figure 9).

7 Policy and practice implications

Forests and other wooded lands are more than just trees. During the past 300 years, the dominant framing of forests in Europe has been as a source of material resources such as wood, fuel, fibre and, to a lesser extent, as natural capital providing multiple benefits. Biodiversity is the core feature of this natural capital onto which value chains of wood, non-wood forest products, amenity values and cultural values are built. Communities' needs and citizens' views of forests and other wooded lands are changing as societies increasingly appreciate the regulating and immaterial values of forests in addition to their direct provisioning value. Rural jobs in forestry and agriculture are declining while business opportunities in urban service sectors related to forest recreation, tourism and regulating services are growing, a process supported in some EU countries by specific governmental programmes on green jobs in forestry.

Our understanding of what biodiversity means, how it can be monitored accurately based on evidence-based performance targets, and, as well as how this status can be maintained or improved through conservation, management and restoration, is more robust than ever before and continues to evolve. The challenge is to encourage collaborative learning on which tools to use for the given biophysical, socio-economic and cultural European context being considered. Forest biodiversity is not the same if defined through a forestry production or a nature conservation lens, and it can be managed with different levels of ambition. Whatever views are considered, both traditional and novel sectoral interests are at stake, and there is no 'one-size-fits-all' solution. Certain measures may be beneficial for some aspects of biodiversity, and negative for others, therefore, evidence-based and robust governance mechanisms are needed to address the complexity of societal trade-offs in satisfactory and sustainable ways.

New policies will not solve all of Europe's biodiversity challenges at once, it has been a long road to get to where we are. Biodiversity responses to new policies manifest themselves after a considerable time lag given the slow pace of forest development. Because a policy is feasible only if there is long-term commitment and societal legitimacy, policy instruments advancing forest biodiversity conservation should be coherent, ambitious and firmly embedded in the portfolio in which they appear. Our growing ambition to address ongoing biodiversity loss is justified, particularly given the ever-higher value placed on biodiversity by society. Coherent direction is needed to guide this ambition as it seeks, among other things, to innovate in the struggle to preserve biodiversity via research, programmes and instruments tailored to specific policy contexts. As noted above, many policy measures to enhance and protect biodiversity do not generate immediately visible impacts; biodiversity conservation is a long-term goal that requires long-term commitment and the impact pathway from forest management, including non-intervention, to positive biodiversity outcomes needs assessment and to be communicated to all stakeholders with an understanding of the delays and lags at play.

Governing forest biodiversity cannot be done with single policy instruments. Improving the effectiveness and legitimacy of forest biodiversity conservation requires an understanding of the entire range of existing and potential policy instruments. New instruments need to dovetail with existing governance frameworks and policy mixes at the global, EU and national levels to ensure that they are complementary rather than contradictory. Policy actions should be tailored to the specific contexts where they will be implemented but still sufficiently coordinated across different scales and spaces to retain coherence. The ongoing success of forest biodiversity policy throughout the EU depends to some degree on the institutional legacy and the history of forest management and biodiversity protection. For this to happen, the large variation in ecological contexts, as well as socio-economic, cultural and institutional settings that differ across forested areas in Europe, needs to be factored in at the embryonic stages of policy formation.

Curbing biodiversity loss needs support from all sectors and actors at multiple levels. Environmental, forest(ry) and rural development authorities, civil society, forest owners and managers, as well as entities from the whole length of value chains should engage in a common effort to develop tools supporting the realisation of forest biodiversity policies on the ground. This includes actors that may harm biodiversity and could carry more of the conservation burden by internalising the externalities of harm. Investment and subsidies should be steered away from harmful activities and towards conservation activities through the use of sustainability criteria when channelling funds. This should be supplemented and supported by effective coordination and knowledge exchange across sectors and amongst different actors.

Considering contextual and regional socio-ecological differences, it will be essential to either design tailor-made approaches or approaches that are sufficiently flexible to undertake appropriate measures for the settings in which they will be applied. It is neither possible nor the purpose of this paper to propose policy and management priorities for individual countries and regions; however, based on the presence or absence profile of the landscape elements listed in Figure 3, it is possible to sketch out a potential biodiversity-conservation framework for every regional context. In the following paragraph, we revisit these landscape elements one by one, ranked from those less-demanding elements within reach of every forest manager up to the more challenging elements requiring concerted action at the regional, national or even international level.


Tree species mixture

Restoring species composition to a natural state and promoting a high diversity of tree species in Europe's forests is a challenge. Also plantation forests used purely for production purposes need more tree diversity for the sake of ecological stability. As a policy measure, tree diversity could be a criterion for restoration, to access subsidies, insurance and calamity funds.


Genetic resources

Forests should contribute to the conservation of genetic resources by securing the viability of key populations with broad genetic variation. In the context of rapid climate change, the natural regeneration of local tree populations, in combination with the proliferation of more heat and drought tolerant (European) tree species or provenances are options to be cautiously considered.


Deadwood

The amount of deadwood in different decay stages, and its mosaic distribution in space, is much lower in managed forests than in natural conditions. Restoration can be easily initiated by leaving deadwood unharvested, however, it will take decades to develop the required thresholds for many deadwood dependent species, meaning that commitments to this must be long-term.


Habitat trees

Single big old trees and small patches of old-growth stands host many microhabitats and provide unique deadwood. Forests without such trees need to be restored by sustaining retention trees and small stands thereof during silvicultural interventions. In forests where they do occur, they should be protected. In wood production systems with long rotations, individual trees with lesser timber value could be left to age and decay naturally.


Microhabitats

Tree-related microhabitats on trees with conservation value appear in all forests, even at a young age. Training foresters and other land managers to recognise and retain trees with these features is important to safeguard them during thinning and logging operations.


Disturbances

Natural disturbances and the biological legacies they create form an essential part of forest ecosystem dynamics. This entails non-intervention in primary forests, securing habitats for disturbance intolerant species. It also entails the retention of disturbed patches or certain biological legacies in disturbed areas of managed forests to enhance the biodiversity connected with early seral stages of forest development. In some countries and regions, this may require a change in legislation.


Rare biotopes

Valuable biotopes, such as springs, water bodies, peatlands, rocks and rare forest types, should be mapped and protected. In many areas of intensive forest management, they can be relatively easily restored. Legislation, subsidies, reverse auctions and local action can contribute significantly to improving the status of these biodiversity hotspots.


Connectivity

In landscapes where forests with a high conservation value are strongly fragmented, connectivity is a concern for the functionality of habitat networks (green infrastructure). Afforestation and the restoration of appropriately designed ecological corridors are viable options to improve connectivity without creating ecological traps for dispersing species.


Cultural landscapes

Cultural landscapes, such as heathlands, woodlands, dehesas and coppice forests often contain biodiversity of high conservation value that require more open environments, including those with low tree density. These areas are often legally part of forest land; raising awareness of their value, providing special protection and specialised management will help keep their open aspect and retain diversity.


Ancient forests

Ancient forests are a somewhat overlooked but very valuable type of forest with unique biodiversity features, even though they are often fragmented and managed. They need to be better mapped, protected against deforestation and managed with care to avoid conversion to intensive forms of forest management.


Old growth forests

Old-growth forests are characterised by later successional phases, including large amounts of old and dead trees. These biodiversity elements are rare and need better protection through EU and national multi-instrument approaches. In addition, there is great potential to restore old-growth forests for which suitable incentives should be developed.


Primary forests

Primary forests have been without any significant human interference for a long time. They combine features of ancient and old-growth forests and are simultaneously the most unique and threatened element of European forest biodiversity. In execution of the EU Biodiversity Strategy and Forest Strategy, all remaining primary forests should therefore receive full protection, a measure that will require legal and financial support at the national and European levels.

To realise the multifunctional roles of European forests that are being ever more ambitiously sought after, much hope has been put on spatial integration approaches where biodiversity conservation and management for other purposes, such as wood production and recreation, go hand in hand at the stand level. Although very valuable, these systems have their limitations and often fail specialised forest biodiversity that needs late-successional phases and large forested areas. For this reason, approaches where multifunctional secondary forests, protected areas (natural reserves, national parks, high-value conservation areas with specific but generally low-intensity or zero intervention management) and intensively-managed forests and plantations coexist within a landscape, deserve more attention. Placing more land under strict protection, made feasible by the intensive management and harvesting of other parts of the forest landscape, is an attractive but largely unexplored option in European forests. Successfully implementing such an approach at scale in European landscapes will need an effective mix of attuned and flexible policy instruments capable of accommodating the complexity of the circumstances surrounding Europe's forests.

Finally, it is important to emphasise the strong context-dependency of forest biodiversity and, therefore, the regional specificity requirements of any policy and finance measures being considered. In Southern Europe, the cultural landscape that has resulted from the long history of human use and degradation of forests is now undergoing a period of rapid forest expansion as a result of high rates of land abandonment.

Although this will slowly lead to increased forest-related biodiversity, the current challenge is to mitigate biodiversity loss in non-forested areas of that cultural landscape. Rural-innovation projects that promote win-win scenarios involving megafire prevention, wood-based bioeconomy development, extensive livestock production and the promotion of cultural and natural heritage hold great promise. Furthermore, in Western and Central Europe as well as Southern Scandinavia, expansive agriculture has reduced and fragmented forested lands over the last centuries. Integrated management of the remaining ancient and other forests could be promoted. Large-scale ecosystem restoration of forests previously converted to conifer monocultures that are now destabilised by atmospheric pollution, insect outbreaks and climate change remains a challenge. Fast-growing plantation forests along the west coast of the continent could consider investments to increase functional diversity and prevent collapse. Despite the designation of Natura2000 areas, Northern Scandinavia and Eastern Europe have an urgent need to address the protection of their primary and old-growth forests. In addition, near-natural forests that have been harvested only once or twice in the recent past offer great opportunities to conserve and restore biological diversity in harmony with other ecosystem services through low-intensity management practices, rather than intensifying their harvesting levels. However, in all the above-mentioned regions, the day-to-day work of keeping and restoring biodiversity falls primarily on the shoulders of the numerous public and private forest owners and other actors in the landscape, together with local communities and civil society, which will need to be motivated, trained and well-supported to address Europe's biodiversity challenges. This will require that these actors have access to a mix of tailor-made financial and other instruments that can successfully restore Europe's exceptional biodiversity heritage.

References

Abrego, N., Salcedo, I. 2014. Response of wood-inhabiting fungal community to fragmentation in a beech forest landscape. Fungal Ecology, 8, 18–27. https://doi.org/10.1016/j.funeco.2013.12.007

Adámek, M., Hadincová, V., Wild, J. 2016. Long-term effect of wildfires on temperate Pinus sylvestris forests: Vegetation dynamics and ecosystem resilience. Forest Ecology and Management 380, 285–295. https://doi.org/10.1016/j.foreco.2016.08.051

Adams, C. I. M., Knapp, M., Gemmell, N. J., Jeunen, G. J., Bunce, M., Lamare, M. D., Taylor, H. R. 2019. Beyond biodiversity: Can environmental DNA (eDNA) cut it as a population genetics tool? Genes, 10(3), 192. https://doi.org/10.3390/genes10030192

Adams, H. D., Zeppel, M. J., Anderegg, W. R., Hartmann, H., Landhäusser, S. M., Tissue, D. T., ... & McDowell, N. G. 2017. A multi-species synthesis of physiological mechanisms in drought-induced tree mortality. Nature Ecology & Evolution, 1(9), 1285–1291. https://doi.org/10.1038/s41559-017-0248-x

Aggestam, F., Pülzl, H. 2018. Coordinating the uncoordinated: the EU Forest Strategy. Forests 9(3):125. https://doi.org/10.3390/f9030125

Agnoletti, M., Emanueli, F. (Eds.). 2016. Biocultural diversity in Europe. Cham, Switzerland: Springer International Publishing. https://link.springer.com/book/10.1007/978-3-319-26315-1#toc

Agra, H.E., Carmel, Y., Smith, R. and Ne'eman, G. 2016. Forest conservation: Global evidence for the effects of interventions. Synopses of Conservation Evidence Series. University of Cambridge, Cambridge, UK.

Alday, J., Martínez de Aragón, J., de-Miguel, S.,Bonet, J.A. 2017. Mushroom biomass and diversity are driven by different spatio-temporal scales along Mediterranean elevation gradients. Scientific reports, 7(1), 1–11. https://doi.org/10.1038/srep45824

Allen, C. D., Breshears, D. D., McDowell, N. G. 2015. On underestimation of global vulnerability to tree mortality and forest die-off from hotter drought in the Anthropocene. Ecosphere, 6(8), 1–55. http://dx.doi.org/10.1890/ES15-00203.1

Allen, C.D., Macalady, A.K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., ... & Cobb, N. H. 2010. A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. Forest Ecology and Management 259(4):660–684. https://doi.org/10.1016/j.foreco.2009.09.001.

Ampoorter E, Barbaro L, Jactel H et al. 2020. Tree diversity is key for promoting the diversity and abundance of forest-associated taxa in Europe. Oikos 129(2), 133–146. https://doi.org/10.1111/oik.06290

Anderegg, W.R.L., Hicke, J.A., Fisher, R.A., Allen, C.D., Aukema, J., Bentz, B., Hood, S., Lichstein, J.W., Macalady, A.K., McDowell, N., Pan, Y., Raffa, K., Sala, A., Shaw, J.D., Stephenson, N.L., Tague, C., Zeppel, M. 2015b. Tree mortality from drought, insects, and their interactions in a changing climate. New Phytologist 208, 674–683. https://doi.org/10.1111/nph.13477

Anderegg, W. R., Schwalm, C., Biondi, F., Camarero, J. J., Koch, G., Litvak, M., ... & Pacala, S., 2015a. Pervasive Tree mortality from drought legacies in forest ecosystems, insects, and their implications for carbon cycle models. Science, 349(6247), 528–532. https://doi.org/10.1126/science.aab1833

Andrén, H. 1994. Effects of habitat fragmentation on birds and mammals in landscapes with different proportions of suitable habitat: a review. Oikos, 71, 3, 355–366. https://doi.org/10.2307/3545823

Andrés, C., Ojeda, F. 2002. Effects of afforestation with pines on woody plant diversity of Mediterranean heathlands in southern Spain. Biodiversity and Conservation, 11(9), 1511–1520. https://doi.org/10.1023/A:1016850708890

Angelstam, P., Asplund, B., Bastian, O., Engelmark, O., Fedoriak, M., Grunewald, K., ... & Öster, L. 2022. Tradition as asset or burden for transitions from forests as cropping systems to multifunctional forest landscapes: Sweden as a case study. Forest Ecology and Management, 505, 119895. https://doi.org/10.1016/j.foreco.2021.119895

Angelstam, P., Boutin, S., Schmiegelow, F., Villard, M.-A., Drapeau, P., Host, G., Innes, J., Isachenko, G., Kuuluvainen, M., Mönkkönen, M., Niemelä, J., Niemi, G., Roberge, J.-M., Spence, J., Stone, D. 2004. Targets for boreal forest biodiversity conservation – a rationale for macroecological research and adaptive management. Ecological Bulletins 51, 487–509. https://www.jstor.org/stable/20113330

Angelstam, P., Dönz-Breuss, M. 2004. Measuring forest biodiversity at the stand scale: An evaluation of indicators in European forest history gradients. Ecological Bulletins, 51, 305–332. https://www.jstor.org/stable/20113319

Angelstam, P., Manton, M., 2021. Effects of Forestry Intensification and Conservation on Green Infrastructures: A Spatio-Temporal Evaluation in Sweden. Land, 10(5), 531. https://doi.org/10.3390/land10050531

Angelstam, P., Manton, M., Green, M., Jonsson, B.G., Mikusiński, G., Svensson, J.,Sabatini, F.M. 2020. Sweden does not meet agreed national and international forest biodiversity targets: A call for adaptive landscape planning. Landscape and Urban Planning, 202, 103838. https://doi.org/10.1016/j.landurbplan.2020.103838

Angelstam, P., Manton, M., Pedersen, S., Elbakidze, M. 2017a. Disrupted trophic interactions affect recruitment of boreal deciduous and coniferous trees in northern Europe. Ecological Applications 27(4): 1108–1123. https://www.jstor.org/stable/26294474

Angelstam, P., Manton, M., Yamelynets, T., Fedoriak, M., Albulescu, A-C., Bravo, F., Cruz, F., Jaroszewicz, B., Kavtarishvili, M., Muñoz-Rojas, J., Sijtsma, F., Washbourne, C., Agnoletti, M., Dobrynin, D., Izakovicova, Z., Jansson, N., Kanka, R., Kopperoinen, L., Lazdinis, M., Metzger, M., van der Moolen, B., Özut, D., Pavloska Gjorgieska, D., Stryamets, N., Tolunay, A., Turkoglu, T., Zagidullina, A. 2021. Maintaining natural and traditional cultural green infrastructures across Europe: Learning from historic and current landscape transformations. Landscape Ecology 36:637–663. https://doi.org/10.1007/s10980-020-01161-y

Angelstam, P., Naumov, V., Elbakidze, M., Manton, M., Priednieks, J., Rendenieks, Z. 2018. Wood production and biodiversity conservation are rival forestry objectives in Europe's Baltic Sea Region. Ecosphere, 9(3), e02119. https://doi.org/10.1002/ecs2.2119

Angelstam, P., Pedersen, S., Manton, M., Garrido, P., Naumov, V., Elbakidze, M. 2017b. Green infrastructure maintenance is more than land cover: large herbivores limit recruitment of key-stone tree species in Sweden. Landscape and Urban Planning 167: 368–377. https://doi.org/10.1016/j.landurbplan.2017.07.019

Angelstam, P., Roberge, J. M., Axelsson, R., Elbakidze, M., Bergman, K. O., Dahlberg, A., ... & Törnblom, J. 2013. Evidence-based knowledge versus negotiated indicators for assessment of ecological sustainability: The Swedish Forest Stewardship Council standard as a case study. Ambio, 42(2), 229–240. https://doi.org/10.1007/s13280-012-0377-z

Araújo, B.B., Oliveira-Santos, L. G. R., Lima-Ribeiro, M. S., Diniz-Filho, J. A. F., Fernandez, F. A. 2017. Bigger kill than chill: The uneven roles of humans and climate on late Quaternary megafaunal extinctions. Quaternary International, 431, 216–222. https://doi.org/10.1016/j.quaint.2015.10.045.

Araújo, M.B., Alagador, D., Cabeza, M., Nogués-Bravo, D., Thuiller, W. 2011. Climate change threatens European conservation areas. Ecology Letters, 14, 484–492. https://doi.org/10.1111/j.1461-0248.2011.01610.x

Aravanopoulos, F.A., Tollefsrud, M.M., Graudal, L., Koskela, J., Kätzel, R., Soto, A., Nagy, L., Pilipovic, A., Zhelev, P., Božic, G., Bozzano, M. 2015. Development of genetic monitoring methods for genetic conservation units of forest trees in Europe. European Forest Genetic Resources Programme (EUFORGEN), Bioversity International, Rome, Italy.

Arnolds, E. 1991. Decline of ectomycorrhizal fungi in Europe. Agriculture, Ecosystems & Environment, 35(2–3), 209–244. https://doi.org/10.1016/0167-8809(91)90052-Y

Arrow, J., Fisher, A.C., 1974. Environmental Preservation, Uncertainty, and Irreversibility. The Quarterly Journal of Economics, 88, 312–319. https://doi.org/10.2307/1883074

Asbeck, T., Großmann, J., Paillet, Y., Wininger, N., Bauhus, J. 2021. The use of tree-related microhabitats as forest biodiversity indicators and to guide integrated forest management. Current Forestry Reports 7, 59–68. https://doi.org/10.1007/s40725-020-00132-5

Asner, G.P., Martin, R.E. 2016. Spectranomics: Emerging science and conservation opportunities at the interface of biodiversity and remote sensing. Global Ecology and Conservation, 8, 212–219. https://doi.org/10.1016/j.gecco.2016.09.010

Aszalós, R., Thom, D., Aakala, T., Angelstam, P., Brūmelis, G., Gálhidy, L., Gratzer, G., Hlásny, T., Katzensteiner, K., Kovács, B. and Knoke, T., 2022. Natural disturbance regimes as a guide for sustainable forest management in Europe. Ecological Applications, p.e2596. https://doi.org/10.1002/eap.2596

Baeten, L., Bauwens, B., De Schrijver, A., De Keersmaeker, L., Van Calster, H., Vandekerkhove, K., Roelandt, B., Beeckman, H., Verheyen, K. 2009. Herb layer changes (1954-2000) related to the conversion of coppice-with-standards forest and soil acidification. Applied Vegetation Science, 12(2),187–197. https://doi.org/10.1111/j.1654-109X.2009.01013.x

Bähner, K. W., Tabarelli, M., Büdel, B., Wirth, R. 2020. Habitat fragmentation and forest management alter woody plant communities in a Central European beech forest landscape. Biodiversity and Conservation, 29(8), 2729–2747. https://doi.org/10.1007/s10531-020-01996-6

Bailey, R.L., Faulkner-Grant, H.A., Martin, V.Y., Phillips, T.B., Bonter, D.N. 2020. Nest usurpation by non-native birds and the role of people in nest box management. Conservation Science and Practice, 2(5), p.e185. https://doi.org/10.1111/csp2.185

Bajc M, Čas M, Ballian D, Kunovac S, Zubić G, Grubešić M, Zhelev P, Paule L, Grebenc T, Kraigher H. 2011. Genetic differentiation of the Western Capercaillie highlights the importance of South-Eastern Europe for understanding the species phylogeography. PloS one 6 (8), 15 p. https://doi.org/10.1371/journal.pone.0023602

Bajc, M., Aravanopoulos, F.A., Westergren, M., Fussi, B., Kavaliauskas, D., Alizoti, P., Kiourtsis, F., Kraigher, H. (eds.) 2020. Manual for forest genetic monitoring. Studia Forestalia Slovenica 167. Silva Slovenica Publishing Centre, 326 p.

Barbaro, L., Allan, E., Ampoorter, E., Castagneyrol, B., Charbonnier, Y., De Wandeler, H., Kerbiriou, C., Milligan, H.T., Vialatte, A., Carnol, M. and Deconchat, M., De Smedt, P., Jactel, H., Koricheva, J., Le Viol, I., Muys, B., Scherer-Lorenzen, M., Verheyen, K., van der Plas, F. 2019. Biotic predictors complement models of bat and bird responses to climate and tree diversity in European forests. Proceedings of the Royal Society B, 286(1894), p.20182193. https://doi.org/10.1098/rspb.2018.2193

Barsoum, N., Fuller, L., Ashwood, F., Reed, K., Bonnet-Lebrun, A. S., Leung, F. 2014. Ground-dwelling spider (Araneae) and carabid beetle (Coleoptera: Carabidae) community assemblages in mixed and monoculture stands of oak (*Quercus robur* L./*Quercus petraea* (Matt.) Liebl.) and Scots pine (*Pinus sylvestris* L.). Forest Ecology and Management, 321, 29–41. https://doi.org/10.1016/j.foreco.2013.08.063

Bascompte, J., Soulé, R. V. 1996. Habitat fragmentation and extinction thresholds in spatially explicit models. Journal of Animal Ecology,65, 465–47. https://doi.org/10.2307/5781

Bässler, C., Ernst, R., Cadotte, M., Heibl, C., Müller, J. 2014. Near-to-nature logging influences fungal community assembly processes in a temperate forest. Journal of Applied Ecology 51, 939–948. https://doi.org/10.1111/1365-2664.12267

Bauhus, J., Forrester, D., Pretzsch, H., Felton, A., Pyttel, P., Benneter, A. 2017. Silvicultural options for mixed-species stands. In: Pretzsch, H., Forrester, D.I., Bauhus, J. (Eds.) Mixed-Species Forests - Ecology and Management. Springer Verlag Germany, Heidelberg, pp. 433–501.

Bauhus, J., Puettmann, K.J., Kuehne, C. 2013. Close-to-nature forest management in Europe: does it support complexity and adaptability of forest ecosystems? In: Messier, C., Puettmann, K.J.,Coates, K.D. (eds.): Managing Forests as Complex Adaptive Systems: building resilience to the challenge of global change. Routledge, The Earthscan forest library, pp. 187–213.

Bauhus, J., Schmerbeck, J. 2010. Silvicultural options to enhance and use forest plantation biodiversity. In: Bauhus, J., van der Meer, P., Kanninen, M. 2010. Ecosystem Goods and Services from Plantation Forests. Earthscan London, pp 96–139.

Bauhus, J., van der Meer, P. and Kanninen, M. 2010. Ecosystem Goods and Services from Plantation Forests. Earthscan, London, 254 p.

Becker, M., Bonneau, M., Le Tacon, F. 1992. Long-term vegetation changes in an *Abies alba* forest: natural development compared with response to fertilization. Journal of Vegetation Science, 3(4), 467–474. https://doi.org/10.2307/3235803

Bełcik, M., Lenda, M., Amano, T., Skórka, P. 2020. Different response of the taxonomic, phylogenetic and functional diversity of birds to forest fragmentation. Scientific reports, 10(1), 1–11. https://doi.org/10.1038/s41598-020-76917-2

Bender, D.J., Contreras, T.A., Fahrig, L. 1998. Habitat loss and population decline: a meta-analysis of the patch size effect. Ecology 79(2), 517–533. https://doi.org/10.1890/0012-9658

Bengtsson, J., Angelstam, P., Elmqvist, T., Emanuelsson, U., Folke, C., Ihse, M., Moberg, F., Nyström, M. 2003. Reserves, resilience and dynamic landscapes. Ambio 32(6), 389–396. https://doi.org/10.1579/0044-7447-32.6.389

Benito Garzón, M., Robson, T. M., Hampe, A. 2019. ΔTrait SDMs: species distribution models that account for local adaptation and phenotypic plasticity. New Phytologist, 222(4), 1757–1765. https://doi.org/10.1111/nph.15716

Bennett, D.D., Tkacz, B.M. 2008. Forest health monitoring in the United States: a program overview. Australian Forestry, 71(3),223–228. https://doi.org/10.1080/00049158.2008.10675039

Bernes, C., Jonsson, B.G., Junninen, K. et al. 2015. What is the impact of active management on biodiversity in boreal and temperate forests set aside for conservation or restoration? A systematic map. Environmental Evidence 4, 25. https://doi.org/10.1186/s13750-015-0050-7

Bernes, C., Macura, B., Jonsson, B.G., Junninen, K., Müller, J., Sandström, J., Lõhmus, A. and Macdonald, E., 2018. Manipulating ungulate herbivory in temperate and boreal forests: effects on vegetation and invertebrates. A systematic review. Environmental Evidence, 7(1), 1–32. https://doi.org/10.1186/s13750-018-0125-3

Bertrand, R., Lenoir, J., Piedallu, C., Riofrio-Dillon, G., de Ruffray, P., Vidal, C., Pierrat, J.C. and Gégout, J.C. 2011. Changes in plant community composition lag behind climate warming in lowland forests. Nature, 479(7374), 517–520. https://doi.org/10.1038/nature10548

Betts, M.G., Phalan, B.T., Wolf, C., Baker, S.C., Messier, C., Puettmann, K.J., Green, R., Harris, S.H., Edwards, D.P., Lindenmayer, D.B.,Balmford, A. 2021. Producing wood at least cost to biodiversity: Integrating Triad and sharing–sparing approaches to inform forest landscape management. Biological Reviews 96, 1301–1317. https://doi.org/10.1111/brv.12703

Bindewald, A., Miocic, S., Wedler, A., Bauhus, J. 2021. Forest inventory-based assessments of the invasion risk of *Pseudotsuga menziesii* (Mirb.) Franco and *Quercus rubra* L. in Germany. European Journal of Forest Research 140, 883–899. https://doi.org/10.1007/s10342-021-01373-0

Blondet, M., de Koning, J., Borrass, L., Ferranti, F., Geitzenauer, M., Weiss, G., Turnhout, E., Winkel, G. 2017. Participation in the implementation of Natura 2000: A comparative study of six EU member states. Land Use Policy, 66, 346–355. https://doi.org/10.1016/j.landusepol.2017.04.004

Bobiec, A., Podlaski, R., Ortyl, B., Korol, M., Havryliuk, S., Öllerer, K., Ziobro, J.M. et al. 2019. Top-down segregated policies undermine the maintenance of traditional wooded landscapes: evidence from oaks at the European Union's eastern border. Landscape and Urban Planning 189: 247–259. https://doi.org/10.1016/j.landurbplan.2019.04.026

Boeraeve, M., Everts, T., Vandekerkhove, K., De Keersmaeker, L., Van de Kerckhove, P., Jacquemyn, H. 2021. Partner turnover and changes in ectomycorrhizal fungal communities during the early life stages of European beech (*Fagus sylvatica* L.). Mycorrhiza, 31(1), 43–53. https://doi.org/10.1007/s00572-020-00998-0

Bollmann, K., Braunisch, V. 2013. To integrate or to segregate: balancing commodity production and biodiversity conservation in European forests. In: Kraus, D., Krumm, F. (eds) Integrative approaches as an opportunity for the conservation of forest biodiversity. European Forest Institute, pp. 18–31.

Boulanger, V., Dupouey, J.L., Archaux, F. et al. 2018. Ungulates increase forest plant species richness to the benefit of non-forest specialists. Global Change Biology 24: 485–495. https://doi.org/10.1111/gcb.13899

Bouriaud, L.,Marzano, M. 2016. Conservation, extraction and corruption: Is sustainable forest management possible in Romania. In: Gilberthorpe, E., Hilson, G. (eds.). Natural resource extraction and indigenous livelihoods: Development challenges in an era of globalization. Ashgate Publishing. Bouwma, I., Schleyer, C., Primmer, E., Winkler, K.J., Berry, P., Young, J., Carmen, E., Špulerová, J., Bezák, P., Preda, E., Vadineanui, A. 2018. Adoption of the ecosystem services concept in EU policies. Ecosystem Service 29, 213–222. https://doi.org/10.1016/j.ecoser.2017.02.014

Bracy Knight, K., Seddon, E. S., Toombs, T. P. 2020. A framework for evaluating biodiversity mitigation metrics. Ambio, 49(6), 1232–1240. https://doi.org/10.1007/s13280-019-01266-y

Bradter, U., Ozgul, A., Griesser, M., Layton-Matthews, K., Eggers, J., Singer, A., Sandercock, B. K., Haverkamp, P.J., Snäll, T. 2021. Habitat suitability models based on opportunistic citizen science data – evaluating forecasts from alternative methods versus an individual-based model. Diversity and Distributions 27, 2397–2411. https://doi.org/10.1111/ddi.13409

Braunisch, V., Segelbacher, G.,Hirzel, A.H. 2010. Modelling functional landscape connectivity from genetic population structure: a new spatially explicit approach. Molecular Ecology, 19(17), 3664–3678. https://doi.org/10.1111/j.1365-294X.2010.04703.x

Brin, A., Bouget, C., Valladares, L., Brustel, H. 2013. Are stumps important for the conservation of saproxylic beetles in managed forests? – Insights from a comparison of assemblages on logs and stumps in oak-dominated forests and pine plantations. Insect conservation and diversity, 6(3), 255–264. https://doi.org/10.1111/j.1752-4598.2012.00209.x

Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., Sayer, J. 2008. Plantation forests and biodiversity: oxymoron or opportunity? Biodiversity and Conservation, 17(5), 925–951. https://doi.org/10.1007/s10531-008-9380-x

Brook, B.W., Sodhi, N.S., Bradshaw, C.J. 2008. Synergies among extinction drivers under global change. Trends in ecology & evolution 23(8), 453–60. https://doi.org/10.1016/j.tree.2008.03.011

Brühl, C. A., Bakanov, N., Köthe,Eichler, L., Sorg, M., Hörren, T., ... & Lehmann, G. U. 2021. Direct pesticide exposure of insects in nature conservation areas in Germany. Scientific reports, 11(1), 1–10. https://doi.org/10.1038/s41598-021-03366-w

Brumelis, G., Jonsson, B.G., Kouki, J., Kuuluvainen, T., Shorohova, E. 2011. Forest naturalness in northern Europe: perspectives on processes, structures and species diversity. Silva Fennica 45(5), 807–821. https://doi.org/10.14214/sf.446

Buée, M., Vairelles, D., Garbaye, J. 2005. Year-round monitoring of diversity and potential metabolic activity of the ectomycorrhizal community in a beech (*Fagus silvatica*) forest subjected to two thinning regimes. Mycorrhiza, 15(4), 235–245. https://doi.org/10.1007/s00572-004-0313-6

Buitenwerf, R., Sandel, B., Normand, S., Mimet, A., Svenning, J.C. 2019. Land-surface greening suggests vigorous woody regrowth throughout European semi-natural vegetation. Global Change Biology 24,5789–5801. https://doi.org/10.1111/gcb.14451

Bull, J., Suttle, K., Gordon, A., Singh, N., Milner-Gulland, E. 2013. Biodiversity offsets in theory and practice. Oryx, 47(3), 369–380. https://doi.org/10.1017/S003060531200172X

Bull, J.W., Strange, N., 2018.The global extent of biodiversity offset implementation under no net loss policies. Nature Sustainability 790, 790–798. https://doi.org/10.1038/s41893-018-0176-z

Bürger-Arndt, R., Welzholz, J.C. 2005. The history of protected forest areas in Europe-from holy groves to Natura 2000 sites. News of Forest History, (36/37 (1)), 40–54.

Burrascano, S., Chytrý, M., Kuemmerle, T., Giarrizzo, E., Luyssaert, S., Sabatini, F. M., & Blasi, C., 2016. Current European policies are unlikely to jointly foster carbon sequestration and protect biodiversity. Biological Conservation, 201, 370–376. https://doi.org/10.1016/j.biocon.2016.08.005

Busse, A., Cizek, L., Čížková, P., Drag, L., Dvorak, V., Foit, J., Heurich, M., Hubený, P., Kašák, J., Kittler, F., Kozel, P., Lettenmaier, L., Nigl, L., Procházka, J., Rothacher, J., Straubinger, C., Thorn, S., Müller, J.2022. Forest dieback in a protected area triggers the return of the primeval forest specialist *Peltis grossa* (Coleoptera, Trogossitidae). Conservation Science and Practice 4:e612. https://doi.org/10.1111/csp2.612

Bütler, R., Lachat, T. 2009. Wälder ohne Bewirtschaftung: eine Chance für die saproxylische Biodiversität. Schweizerische Zeitschrift für Forstwesen 160(11):324–333. https://doi.org/10.3188/szf.2009.0324

Bütler, R., Lachat, T., Krumm, F., Kraus, D., Larrieu, L. 2020. Field guide to tree-related microhabitats. Descriptions and size limits for their inventory. Birmensdorf: Swiss Federal Institute for Forest, Snow and Landscape Research WSL.

Bütler, R., Lachat, T., Larrieu, L., Paillet, Y., Kraus, D. et al. 2013. Habitat trees: key elements for forest biodiversity. In: Kraus, D. and Krumm, F.(eds.). In Focus – Managing Forests in Europe: 'Integrative approaches as an opportunity for the conservation of forest biodiversity'. European Forest Institute.

Campos, P., Huntsinger, L., Oviedo, J.L., Starrs, P.F., Díaz, M., Standiford, R.B., Montero, G. 2013. Mediterranean oak woodland working landscapes. Landscape Series, 16.

Canelles, Q., Aquilué, N., James, P., Lawler, J., Brotons, L. 2021. Global review on interactions between insect pests and other forest disturbances. Landscape Ecology, 36(4), 945–972. https://doi.org/10.1007/s10980-021-01209-7

Carpio, A.J., Apollonio, M., Acevedo, P. 2021. Wild ungulate overabundance in Europe: contexts, causes, monitoring and management recommendations. Mammal Review, 51(1), 95–108. https://doi.org/10.1111/mam.12221

Castano-Villa, G. J., Estevez, J. V., Guevara, G., Bohada-Murillo, M., Fonturbel, F. E. 2019. Differential effects of forestry plantations on bird diversity: a global assessment. Forest Ecology and Management, 440, 202–207. https://doi.org/10.1016/j.foreco.2019.03.025

CBD 2010. Decision X/2 The Strategic Plan for Biodiversity 2011–2020 and the Aichi Biodiversity Targets.

CBD 2020. Global Biodiversity Outlook 5, Bonn.

CBD 2021. Preparation of the post-2020 global biodiversity framework, CBD/WG2020/3/L2, Bonn.

Chandler, M., See, L., Copas, K., Bonde, A.M., López, B.C., Danielsen, F., Legind, J.K., Masinde, S., Miller-Rushing, A.J., Newman, G., Rosemartin, A., Turak, E. 2017. Contribution of citizen science towards international biodiversity monitoring. Biological conservation, 213, 280–294. https://doi.org/10.1016/j.biocon.2016.09.004

Chapman, D., Purse, B. V., Roy, H. E., & Bullock, J. M. 2017. Global trade networks determine the distribution of invasive non-native species. Global Ecology and Biogeography, 26(8), 907–917. https://doi.org/10.1111/geb.12599

Chaudhary, A., Burivalova, Z., Koh, L. P., Hellweg, S. 2016. Impact of forest management on species richness: global meta-analysis and economic trade-offs. Scientific reports, 6(1), 1–10. https://doi.org/10.1038/srep23954

Choat, B., Jansen, S., Brodribb, T. J., Cochard, H., Delzon, S., Bhaskar, R., ... & Zanne, A. E. 2012. Global convergence in the vulnerability of forests to drought. Nature, 491(7426), 752–755. https://doi.org/10.1038/nature11688

Cimatti, M., Ranc, N., Benítez-López, A., Maiorano, L., Boitani, L., Cagnacci, F., Čengić, M., Ciucci, P., Huijbregts, M.A., Krofel, M., López-Bao, J.V. 2021. Large carnivore expansion in Europe is associated with human population density and land cover changes. Diversity and Distributions, 27(4), .602–617. https://doi.org/10.1111/ddi.13219

Clare, E.L., Economou, C.K., Faulkes, C.G., Gilbert, J.D., Bennett, F., Drinkwater, R., Littlefair, J.E. 2021. eDNAir: proof of concept that animal DNA can be collected from air sampling. PeerJ, 9, e11030. https://doi.org/10.7717/peerj.11030

Côté, P., Tittler, R., Messier, C., Kneeshaw, D.D., Fall, A., Fortin, M.J. 2010. Comparing different forest zoning options for landscape-scale management of the boreal forest: possible benefits of the TRIAD. Forest Ecology and Management, 259 (3), 418–427. https://doi.org/10.1016/j.foreco.2009.10.038

Courbaud , B., Larrieu, L., Kozak, D., Kraus, D., Lachat, T., Ladet, S., Müller, J., Paillet, Y., Sagheb-Talebi, K., Schuck, A., Stillhard, J., Svoboda, M., Zudin, S. 2022. Factors influencing the formation rate of tree related microhabitats and implications for biodiversity conservation and forest management. Journal of Applied Ecology 59, 492– 503 https://doi.org/10.1111/1365-2664.14068

Crookes, S., Heer, T., Castañeda, R.A., Mandrak, N.E., Heath, D.D., Weyl, O.L., MacIsaac, H.J., Foxcroft, L.C., 2020. Monitoring the silver carp invasion in Africa: a case study using environmental DNA (eDNA) in dangerous watersheds. NeoBiota, 56, 31–47. https://doi.org/10.3897/neobiota.56.47475

Cubbage, F., Sills, E. 2020. Forest Certification and Forest Use: A Comprehensive Analysis. In W. Nikolakis, J. Innes (Eds.), The Wicked Problem of Forest Policy: A Multidisciplinary Approach to Sustainability in Forest Landscapes (pp. 59–107). Cambridge: Cambridge University Press.

Czyż, E.A, Escribà, C.G., Wulf H.,Tedder A. ,Schuman, M.C., Schneider, F.D., Schaepman, M.E. 2020. Intraspecific genetic variation of a *Fagus sylvatica* population in a temperate forest derived from airborne imaging spectroscopy time series. Ecology & Evolution 10, 7419–7430. https://doi.org/10.1002/ece3.6469

Dahl, F., Åhlén, P.A. 2019. Nest predation by raccoon dog *Nyctereutes procyonoides* in the archipelago of northern Sweden. Biological Invasions, 21(3), 743–755. https://doi.org/10.1007/s10530-018-1855-4

D'Amato, D., Korhonen, J. 2021. Integrating the green economy, circular economy and bioeconomy in a strategic sustainability framework. Ecological Economics, 188, 107143. https://doi.org/10.1016/j.ecolecon.2021.107143

Dasgupta, P. 2021. The Economics of Biodiversity: the Dasgupta Review. HM Treasury.

De Frenne, P., Zellweger, F., Rodríguez-Sánchez, F., Scheffers, B. R., Hylander, K., Luoto, M., Vellend, M., Verheyen, K., Lenoir, J. 2019. Global buffering of temperatures under forest canopies. Nature ecology & evolution, 3(5), 744–749. https://doi.org/10.1038/s41559-019-0842-1

Decocq, G., Andrieu, E., Brunet, J., Chabrerie, O., De Frenne, P., De Smedt, P., Deconchat, M., Diekmann, M., Ehrmann, S., Giffard, B., Mifsud, E.G. 2016. Ecosystem services from small forest patches in agricultural landscapes. Current Forestry Reports, 2(1),30–44. https://doi.org/10.1007/s40725-016-0028-x

Deinet, S., Ieronymidou, C., McRae, L., Burfield, I.J., Foppen, R.P., Collen, B., Böhm, M. 2013. Wildlife comeback in Europe. The recovery of selected mammal and bird species. Zoological Society of London, UK.

Demant, L., Bergmeier, E., Walentowski, H., Meyer, P. 2020. Suitability of contract-based nature conservation in privately-owned forests in Germany. Nature Conservation, 42, 89. https://doi.org/10.3897/natureconservation.42.58173

Desie, E., Van Meerbeek, K., De Wandeler, H., Bruelheide, H., Domisch, T., Jaroszewicz, B., Joly, F.X., Vancampenhout, K., Vesterdal, L., Muys, B. 2020b. Positive feedback loop between earthworms, humus form and soil pH reinforces earthworm abundance in European forests. Functional Ecology, 34(12), 2598–2610. https://doi.org/10.1111/1365-2435.13668

Desie, E., Vancampenhout, K., Heyens, K., Hlava, J., Verheyen, K., Muys, B. 2019. Forest conversion to conifers induces a regime shift in soil process domain affecting carbon stability. Soil Biology and Biochemistry, 136, 107540. https://doi.org/10.1016/j.soilbio.2019.107540

Desie, E., Vancampenhout, K., Nyssen, B., van den Berg, L., Weijters, M., van Duinen, G.J., den Ouden, J., Van Meerbeek, K., Muys, B. 2020a. Litter quality and the law of the most limiting: Opportunities for restoring nutrient cycles in acidified forest soils. Science of the Total Environment, 699, 134383. https://doi.org/10.1016/j.scitotenv.2019.134383

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z., Hill, R., Chan, K.M., Baste, I.A., Brauman, K.A, Polasky, S, et al. 2018. Assessing nature's contributions to people. Science, 359(6373), 270–272. https://doi.org/10.1126/science.aap8826

Díaz, S., Zafra-Calvo, N., Purvis, A., Verburg, P.H., Obura, D., Leadley, P., Chaplin-Kramer, R., De Meester, L., Dulloo, E., Martín-López, B., Shaw, M.R. et al. 2020. Set ambitious goals for biodiversity and sustainability. Science, 370(6515), 411–413. https://doi.org/10.1126/science.abe1530

Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., Mayorga, J., Olson, D., Asner, G. P., Baillie, J. E. M., Burgess, N. D., Burkart, K., Noss, R. F., Zhang, Y. P., Baccini, A., Birch, T., Hahn, N., Joppa, L. N., Wikramanayake, E.. 2019. A global deal for nature: guiding principles, milestones, and targets. Science advances, 5(4), p.eaaw2869. https://doi.org/10.1126/sciadv.aaw2869

Douda, J., Boublík, K., Doudová, J., Kyncl, M. 2017. Traditional forest management practices stop forest succession and bring back rare plant species. Journal of Applied Ecology, 54(3), 761–771. https://doi.org/10.1111/1365-2664.12801

Dove, N. C., Hart, S. C. 2017. Fire reduces fungal species richness and in situ mycorrhizal colonization: a meta-analysis. Fire Ecology, 13(2), 37–65. https://doi.org/10.4996/fireecology.130237746

Duncan, C., Thompson, J.R., Pettorelli, N. 2015. The quest for a mechanistic understanding of biodiversity–ecosystem services relationships. Proceedings of the Royal Society B: Biological Sciences, 282(1817), 20151348. https://doi.org/10.1098/rspb.2015.1348

Dyderski, M.K., Paź, S., Frelich, L.E., Jagodziński, A.M. 2018. How much does climate change threaten European forest tree species distributions? Global Change Biology, 24(3), 1150–1163. https://doi.org/10.1111/gcb.13925

EC 2020. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. EU Biodiversity Strategy for 2030: Bringing nature back into our lives. COM/2020/380 final.

EC 2021. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. New EU Forest Strategy for 2030. Brussels, 16.7.2021. COM (2021) 572 Final.

Edwards, P., Kleinschmit, D. 2013. Towards a European forest policy—conflicting courses. Forest Policy and Economics, 33, 87–93. https://doi.org/10.1016/j.forpol.2012.06.002

EEA 2016. European forest ecosystems: state and trends. European Environment Agency report 5, Copenhagen, Denmark.

EEA 2019. The European environment — state and outlook 2020. Knowledge for transition to a sustainable Europe. European Environment Agency, Copenhagen, Denmark.

Eggers, J., Räty, M., Öhman, K., Snäll, T. 2020. How well do stakeholder-defined forest management scenarios balance economic and ecological forest values? Forests, 11(1), 86. https://doi.org/10.3390/f11010086

Eichenberg, D., Bowler, D.E., Bonn, A., Bruelheide, H., Grescho, V., Harter, D., Jandt, U., May, R., Winter, M., Jansen, F. 2021. Widespread decline in Central European plant diversity across six decades. Global Change Biology, 27(5), 1097–1110. https://doi.org/10.1111/gcb.15447

El-Kassaby, Y.A., Dunsworth, B.G., Krakowski, J. 2003. Genetic evaluation of alternative silvicultural systems in coastal montane forests: western hemlock and amabilis fir. Theoretical and Applied Genetics, 107(4), 598–610. https://doi.org/10.1007/s00122-003-1291-3

Engel, S., Pagiola, S. Wunder, 2008. Designing payments for environmental services in theory and practice: An overview of the issues. Ecological Economics, 65, 663–674. https://doi.org/10.1016/j.ecolecon.2008.03.011

Enoksson, B., Angelstam, P., Larsson, K. 1995. Deciduous forest and resident birds: the problem of fragmentation within a coniferous forest landscape. Landscape Ecology, 10(5), 267–275. https://doi.org/10.1007/BF00128994

Esseen, P.-A., Ekström, M., Grafström, A., Jonsson, B. G., Palmqvist, K., Westerlund, B., Ståhl, G. 2022. Multiple drivers of large-scale lichen decline in boreal forest canopies. Global Change Biology, 00, 1– 17. https://doi.org/10.1111/gcb.16128

EUFORGEN 2021. Forest Genetic Resources Strategy For Europe. 2021, European Forest Institute.

Fady, B., Cottrell, J., Ackzell, L., Alía, R., Muys, B., Prada, A. and González-Martínez, S.C. 2016. Forests and global change: what can genetics contribute to the major forest management and policy challenges of the twenty-first century? Regional Environmental Change, 16(4), 927–939. https://doi.org/10.1007/s10113-015-0843-9

Fahrig, L. 2003. Effects of habitat fragmentation on biodiversity. Annual review of ecology, evolution, and systematics, 34(1), 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fahrig, L. 2013. Rethinking patch size and isolation effects: the habitat amount hypothesis. Journal of Biogeography 40:1649–1663. https://doi.org/10.1111/jbi.12130

FAO 2000. On definitions of forest and forest change. FRA Working Paper No. 33. Rome.

Fedrowitz, K., Koricheva, J., Baker, S.C., Lindenmayer, D.B., Palik, B., Rosenvald, R., Beese, W., Franklin, J.F., Kouki, J., Macdonald, E., Messier, C. 2014. Can retention forestry help conserve biodiversity? A meta-analysis. Journal of Applied Ecology, 51(6), 1669–1679. https://doi.org/10.1111/1365-2664.12289

Felton, A., Boberg, J., Björkman, C., Widenfalk, O. 2013. Identifying and managing the ecological risks of using introduced tree species in Sweden's production forestry. Forest Ecology and Management, 307, 165–177. https://doi.org/10.1016/j.foreco.2013.06.059

Ferraro, P.J., Kiss, A. 2002. Direct Payments to Conserve Biodiversity. Science 298: 1718–1719. https://doi.org/10.1126/science.1078104

Ferraro, P.J. 2008. Asymmetric information and contract design for payments for environmental services. Ecological economics, 65: 810–821. https://doi.org/10.1016/j.ecolecon.2007.07.029

Flaspohler, D. J., Webster, C. R. 2011. Plantations for bioenergy: principles for maintaining biodiversity in intensively managed forests. Forest Science, 57(6), 516–524. https://doi.org/10.1093/forestscience/57.6.516

Fløjgaard, C., Bruun, H.H., Hansen, M.D., Heilmann-Clausen, J., Svenning, J.C., Ejrnæs, R. 2018. Are ungulates in forests concerns or key species for conservation and biodiversity? Reply to Boulanger et al. Global change biology, 24(3), 869–871. https://doi.org/10.1111/gcb.14029

Forest Europe 2011. State of Europe's Forests. Status and trends in sustainable forest management in Europe. Ministerial Conference on the Protection of Forests in Europe, Forest Europe, Liaison Unit Oslo.

Forest Europe 2020. State of Europe's forests. Bratislava, Slovakia.

Franklin, J.F. 1988. Structural and functional diversity in temperate forests. In: Wilson, E.O. (ed.). Biodiversity. National Academy Press, Washington, D.C., pp 166–175.

Freer-Smith, P., Muys, B., Bozzano, M., Drössler, L., Farrelly, N., Jactel, H., Korhonen, J., Minotta, G., Nijnik, M., Orazio, C. 2019. Plantation forests in Europe: challenges and opportunities. From Science to Policy 9. European Forest Institute. https://doi.org/10.36333/fs09

Fritz Ö, Heilmann-Clausen J. 2010. Rot holes create key microhabitats for epiphytic lichens and bryophytes on beech (*Fagus sylvatica*). Biological Conservation 143:1008–1016. https://doi.org/10.1016/j.biocon.2010.01.016

Fritz, Ö., Gustafsson, L., Larsson, K. 2008. Does forest continuity matter in conservation? A study of epiphytic lichens and bryophytes in beech forests of southern Sweden. Biological conservation, 141(3), 655–668. https://doi.org/10.1016/j.biocon.2007.12.006

Fuller, L., Ashwood, F., Reed, K., Bonnet-Lebrun, A. S., Leung, F. 2014. Ground-dwelling spider (Araneae) and carabid beetle (Coleoptera: Carabidae) community assemblages in mixed and monoculture stands of oak (*Quercus robur* L./*Quercus petraea* (Matt.) Liebl.) and Scots pine (*Pinus sylvestris* L.). Forest Ecology and Management, 321, 29–41. https://doi.org/10.1016/j.foreco.2013.08.063

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M.C., Fröberg, M., Stendahl, J., Philipson, C.D., Mikusiński, G., 2013. Higher levels of multiple ecosystem services are found in forests with more tree species. Nature communications, 4(1), 1–8. https://doi.org/10.1038/ncomms2328

Gao, T., Nielsen, A. B., Hedblom, M. 2015. Reviewing the strength of evidence of biodiversity indicators for forest ecosystems in Europe. Ecological Indicators, 57, 420–434. https://doi.org/10.1016/j.ecolind.2015.05.028

Gippoliti S. 2004. Captive-breeding and conservation of the European mammal diversity. Hystrix, the Italian Journal of Mammalogy 15 (1), 35–53. https://doi.org/10.4404/hystrix-15.1-4324

Godinho, S., Guiomar, N., Machado, R., Santos, P., Sá-Sousa, P., Fernandes, J. P., ... & Pinto-Correia, T. 2016. Assessment of environment, land management, and spatial variables on recent changes in montado land cover in southern Portugal. Agroforestry systems, 90(1), 177–192. https://doi.org/10.1007/s10457-014-9757-7

Gómez-Baggethun, E., De Groot, R., Lomas, P. L., Montes, C. 2010. The history of ecosystem services in economic theory and practice: from early notions to markets and payment schemes. Ecological economics, 69(6), 1209–1218. https://doi.org/10.1016/j.ecolecon.2009.11.007

Gonçalves, S.C., Haelewaters, D., Furci, G., Mueller, G.M. 2021. Include all fungi in biodiversity goals. Science, 373(6553), 403–403. https://doi.org/10.1126/science.abk1312

Gong, C., Tan, Q., Liu, G., Xu, M. 2021. Impacts of tree mixtures on understory plant diversity in China. Forest Ecology and Management, 498, 119545. https://doi.org/10.1016/j.foreco.2021.119545

Gossner, M.M., Lachat, T., Brunet, J., Isacsson, G., Bouget, C., Brustel, H., Brandl, R., Weisser, W.W., Müller, J. 2013. Current "near-to-nature" forest management effects on functional trait composition of saproxylic beetles in beech forests. Conservation Biology, 27, 605–614. https://doi.org/10.1111/cobi.12023

Gossner, M.M., Lade, P., Rohland, A., Sichardt, N., Kahl, T., Bauhus, J., Weisser, W.W., Petermann, J.S. 2016. Effects of management on aquatic tree-hole communities in temperate forests are mediated by detritus amount and water chemistry. Journal of Animal Ecology 85:213–226. https://doi.org/10.1111/1365-2656.12437

Gossner, M.M., Chao, A., Bailey, R.I., Prinzing, A. 2009. Native fauna on exotic trees: phylogenetic conservatism and geographic contingency in two lineages of phytophages on two lineages of trees. The American Naturalist, 173(5), 599–614. https://doi.org/10.1086/597603

Graveland, J., Van Der Wal, R., Van Balen, J.H., Van Noordwijk, A.J. 1994. Poor reproduction in forest passerines from decline of snail abundance on acidified soils. Nature, 368(6470), 446–448. https://doi.org/10.1038/368446a0

Guénette, J. S., Villard, M. A. 2005. Thresholds in forest bird response to habitat alteration as quantitative targets for conservation. Conservation Biology, 19(4), 1168–1180. https://doi.org/10.1111/j.1523-1739.2005.00085.x

Gunderson, L. H. 2000. Ecological resilience – in theory and application. Annual review of ecology and systematics, 31(1), 425–439. https://doi.org/10.1146/annurev.ecolsys.31.1.425

Gustafsson, L., Baker, S.C., Bauhus, J., Beese, W.J., Brodie, A., Kouki, J., Lindenmayer, D. B., Lõhmus, A., Martínez Pastur, G., Messier C., Neyland, M., Palik, B., Sverdrup-Thygeson, A., Volney, W.J.A., Wayne, A., Franklin, J.F. 2012. Retention Forestry to Maintain Multifunctional Forests: a World Perspective. Bioscience 62, 7, 633–645. https://doi.org/10.1525/bio.2012.62.7.6

Gustafsson, L., Bauhus, J., Asbeck, T., Augustynczik, A.L.D., Basile, M., Frey, J., Gutzat, F., Hanewinkel, M., Helbach, J., Jonker, M., Knuff, A. et al. 2020. Retention as an integrated biodiversity conservation approach for continuous-cover forestry in Europe. Ambio, 49(1), 85–97. https://doi.org/10.1007/s13280-019-01190-1

Guynn Jr, D. C., Guynn, S. T., Wigley, T. B., Miller, D. A. 2004. Herbicides and forest biodiversity—what do we know and where do we go from here? Wildlife Society Bulletin, 32(4), 1085–1092. https://doi.org/10.2193/0091-7648(2004)032[1085:HAFBDW]2.0.CO;2

Halada, L., Evans, D., Romão, C., Petersen, J. E. et al. 2011. Which habitats of European importance depend on agricultural practices? Biodiversity and Conservation, 20(11), 2365–2378. https://doi.org/10.1007/s10531-011-9989-z

Hampe, A., Alfaro-Sánchez, R., Martín-Forés, I. 2020. Establishment of second-growth forests in human landscapes: ecological mechanisms and genetic consequences. Annals of Forest Science, 77(3), 1–5. https://doi.org/10.1007/s13595-020-00993-7

Hanley, N., Perrings, C. 2019. The Economic Value of Biodiversity. Annual Review of Resource Economics, 11, 355–375. https://doi.org/10.1146/annurev-resource-100518-093946

Hanski, I. 2011. Habitat Loss, the Dynamics of Biodiversity, and a Perspective on Conservation. Ambio 40: 248–255. https://doi.org/10.1007/s13280-011-0147-3

Hardisty, A.R., Michener, W.K., Agosti, D., García, E.A., Bastin, L., Belbin, L., Bowser, A., Buttigieg, P.L., Canhos, D.A., Egloff, W., De Giovanni, R. et al. 2019. The Bari Manifesto: An interoperability framework for essential biodiversity variables. Ecological informatics, 49, 22–31. https://doi.org/10.1016/j.ecoinf.2018.11.003

Harper, L.R., Buxton, A.S., Rees, H.C., Bruce, K., Brys, R., Halfmaerten, D., Read, D.S., Watson, H.V., Sayer, C.D., Jones, E.P., Priestley, V. et al. 2019. Prospects and challenges of environmental DNA (eDNA) monitoring in freshwater ponds. Hydrobiologia, 826(1), 25–41. https://doi.org/10.1007/s10750-018-3750-5

Harris, J. E., Rodenhouse, N. L., Holmes, R. T. 2019. Decline in beetle abundance and diversity in an intact temperate forest linked to climate warming. Biological Conservation, 240, 108219. https://doi.org/10.1016/j.biocon.2019.108219

Hartley, M. J. 2002. Rationale and methods for conserving biodiversity in plantation forests. Forest ecology and management, 155(1–3), 81–95. https://doi.org/10.1016/S0378-1127(01)00549-7

Harvey, J. A., Heinen, R., Gols, R., Thakur, M. P. 2020. Climate change-mediated temperature extremes and insects: From outbreaks to breakdowns. Global Change Biology, 26(12), 6685–6701. https://doi.org/10.1111/gcb.15377

Hausknecht, R., Jacobs, S., Müller, J., Zink, R., Frey, H., Solheim, R., Vrezec, A., Kristin, A., Mihok, J., Kergalve, I., Saurola, P. 2014. Phylogeographic analysis and genetic cluster recognition for the conservation of Ural Owls (*Strix uralensis*) in Europe. Journal of Ornithology, 155(1), 121–134. https://doi.org/10.1007/s10336-013-0994-8

Heidrich, L., Bae, S., Levick, S., Seibold, S., Weisser, W. W., Krzystek, P., ... & Müller, J. 2020. Heterogeneity–diversity relationships differ between and within trophic levels in temperate forests. Nature ecology & evolution, 4(9), 1204–1212. https://doi.org/10.1038/s41559-020-1245-z

Heinrichs, S., Ammer, C., Mund, M., Boch, S., Budde, S., Fischer, M., ... & Schall, P. 2019. Landscape-scale mixtures of tree species are more effective than stand-scale mixtures for biodiversity of vascular plants, bryophytes and lichens. Forests, 10(1), 73. https://doi.org/10.3390/f10010073

Hellström, E. 2001. Conflict cultures – Qualitative Comparative Analysis of environmental conflicts in forestry. Silva Fennica Monographs 2. 109 p.

Helm, D., Hepburn, C. (eds.) 2014. Nature in Balance: The Economics of Biodiversity. Oxford, UK: Oxford University Press, 448 pp.

Hermy, M., Honnay, O., Firbank, L. et al. 1999. An ecological comparison between ancient and other forest plant species of Europe, and the implications for forest conservation. Biological Conservation 91: 9–22. https://doi.org/10.1016/S0006-3207(99)00045-2

Herzon, I., Birge, T., Allen, B., Povellato, A., Vanni, F., Hart, K., Radley, G., Tucker, G., Keenleyside, C., Oppermann, R., Underwood, E. 2018. Time to look for evidence: Results-based approach to biodiversity conservation on farmland in Europe. Land Use Policy, 71, 347–354. https://doi.org/10.1016/j.landusepol.2017.12.011

Hilmers, T., Friess, N., Bässler, C., Heurich, M., Brandl, R., Pretzsch, H., Seidl, R., Müller, J. 2018. Biodiversity along temperate forest succession. Journal of Applied Ecology, 55(6), 2756–2766. https://doi.org/10.1111/1365-2664.13238

Hjältén, J., Stenbacka, F., Andersson, J. 2010. Saproxylic beetle assemblages on low stumps, high stumps and logs: Implications for environmental effects of stump harvesting. Forest Ecology and Management, 260(7), 1149–1155. https://doi.org/10.1016/j.foreco.2010.07.003

Hlásny, T., Krokene, P., Liebhold, A., Montagné-Huck, C., Müller, J., Qin, H., Raffa, K., Schelhaas, M-J., Seidl, R., Svoboda, M., Viiri, H. 2019. Living with bark beetles: impacts, outlook and management options. From Science to Policy 8. European Forest Institute. https://doi.org/10.36333/fs08

Hodgson, I. D., Redpath, S. M., Fischer, A., Young, J. 2019. Who knows best? Understanding the use of research-based knowledge in conservation conflicts. Journal of environmental management, 231, 1065–1075. https://doi.org/10.1016/j.jenvman.2018.09.023

Hoffmann, S., Irl, D.H., Beierkuhnlein, S. 2019. Predicted climate shifts within terrestrial protected areas worldwide. Nature Communications 10(1):1–10. https://doi.org/10.1038/s41467-019-12603-w

Honnay, O., Verheyen, K., Hermy, M. 2002. Permeability of ancient forest edges for weedy plant species invasion. Forest Ecology and Management, 161(1–3), 109–122. https://doi.org/10.1016/S0378-1127(01)00490-X

Hothorn, T., Müller, J. 2010. Large-scale reduction of ungulate browsing by managed sport hunting. Forest Ecology and Management, 260(9), 1416–1423. https://doi.org/10.1016/j.foreco.2010.07.019

Howlett, M., Cashore, B. 2014. Conceptualizing Public Policy. In: Engeli, I., Allison, C.R. (eds) Comparative Policy Studies. Research Methods Series. Palgrave Macmillan, London. https://doi.org/10.1057/9781137314154_2

Hultberg, T., Sandström, J., Felton, A., Öhman, K., Rönnberg, J., Witzell, J., Clearly, M. 2020. Ash dieback risks an extinction cascade. Biological Conservation 244, 108516. https://doi.org/10.1016/j.biocon.2020.108516

Humphrey, J., Bailey, S. 2012. Managing deadwood in forests and woodlands. Forestry Commission Practice Guide. Forestry Commission, Edinburgh.

ICP Forests 2020. Forest Condition in Europe – The 2020 Assessment. ICP Forests Technical Report under the UNECE Convention on Long-range Transboundary Air Pollution, 95p.

IPBES 2015. IPBES/4/INF/1: preliminary guide regarding diverse conceptualization of multiple values of nature and its benefits, including biodiversity and ecosystem functions and services. Report of the Fourth Session of the Plenary of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

IPBES 2018. Summary for policymakers of the regional assessment report on biodiversity and ecosystem services for Europe and Central Asia of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. M. Fischer, M. Rounsevell, A. Torre-Marin Rando, A. Mader, A. Church, M. Elbakidze, V. Elias, T. Hahn, P.A. Harrison, J. Hauck, B. Martín-López, I. Ring, C. Sandström, I. Sousa Pinto, P. Visconti, N.E. Zimmermann and M. Christie (eds.). IPBES secretariat, Bonn, Germany. 48 pages.

IPBES 2019. Summary for policymakers of the global assessment report on biodiversity and ecosystem services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. S. Díaz, J. Settele, E. S. Brondízio, H. T. Ngo, M. Guèze, J. Agard, A. Arneth, P. Balvanera, K. A. Brauman, S. H. M. Butchart, K. M. A. Chan, L. A. Garibaldi, K. Ichii, J. Liu, S. M. Subramanian, G. F. Midgley, P. Miloslavich, Z. Molnár, D. Obura, A. Pfaff, S. Polasky, A. Purvis, J. Razzaque, B. Reyers, R. Roy Chowdhury, Y. J. Shin, I. J. Visseren-Hamakers, K. J. Willis, and C. N. Zayas (eds.). IPBES secretariat, Bonn, Germany. 56 pages.

IUCN 2021. The IUCN Red List of Threatened Species. Version 2021-3. https://www.iucnredlist.org . Accessed on [2022-03-01].

Jacobsen, J.B., Vedel, S., Thorsen, B. J. 2013. Assessing costs of multifunctional NATURA2000 management restrictions in continuous cover beech forest management. Forestry, 86: 575–582. https://doi.org/10.1093/forestry/cpt023

Jactel, H., Bauhus, J., Boberg, J. et al. 2017. Tree Diversity Drives Forest Stand Resistance to Natural Disturbances. Current Forestry Reports 3, 223–243. https://doi.org/10.1007/s40725-017-0064-1

Jactel, H., Koricheva, J., Castagneyrol, B. 2019. Responses of forest insect pests to climate change: not so simple. Current Opinion in Insect Science, 35, 103–108. https://doi.org/10.1016/j.cois.2019.07.010

Jactel, H., Moreira, X., Castagneyrol, B. 2021. Tree diversity and forest resistance to insect pests: patterns, mechanisms, and prospects. Annual Review of Entomology, 66, 277–296. https://doi.org/10.1146/annurev-ento-041720-075234

Jactel, H., Petit, J., Desprez-Loustau, M. L., Delzon, S., Piou, D., Battisti, A., Koricheva, J. 2012. Drought effects on damage by forest insects and pathogens: a meta-analysis. Global Change Biology 18:267–276. https://doi.org/10.1111/j.1365-2486.2011.02512.x

Jing, X., Muys, B., Baeten, L., Bruelheide, H., De Wandeler, H., Desie, E., Hättenschwiler, S., Jactel, H., Jaroszewicz, B., Jucker, T., Kardol, P., Pollastrini, M., Ratcliffe, S., Scherer-Lorenzen, M., Selvi, F., Vancampenhout, K., van der Plas, F., Verheyen, K., Vesterdal, L., Zuo, J., Van Meerbeek, K. 2021. Climatic conditions, not above- and belowground resource availability and uptake capacity, mediate tree diversity effects on productivity and stability. Science of the Total Environment 812. https://doi.org/10.1016/j.scitotenv.2021.152560

Jonsson, B.G., Ekström, M., Esseen, P.A., Grafström, A., Ståhl, G., Westerlund, B. 2016. Dead wood availability in managed Swedish forests–Policy outcomes and implications for biodiversity. Forest Ecology and Management, 376, 174–182. https://doi.org/10.1016/j.foreco.2016.06.017

Joppa, L.N., O'Connor, B., Visconti, P., Smith, C., Geldmann, J., Hoffmann, M., Watson, J.E., Butchart, S.H., Virah-Sawmy, M., Halpern, B.S., Ahmed, S.E. 2016. Filling in biodiversity threat gaps. Science, 352(6284), 416–418. https://doi.org/10.1126/science.aaf3565

Jordan, A. 1999. Editorial introduction: the construction of a multilevel environmental governance system. Environment and Planning C: Government and Policy, 17(1), 1–17. https://doi.org/10.1068%2Fc170001

Jucker, T., Bouriaud, O., Avacaritei, D., Coomes, D.A. 2014. Stabilizing effects of diversity on aboveground wood production in forest ecosystems: linking patterns and processes. Ecology Letters, 17(12), 1560–1569. https://doi.org/10.1111/ele.12382

Karr, J.R., Larson, E.R., Chu, E.W. 2021. Ecological integrity is both real and valuable. Conservation Science and Practice, e583. https://doi.org/10.1111/csp2.583

Kataja-aho, S., Fritze, H., Haimi, J. 2011. Short-term responses of soil decomposer and plant communities to stump harvesting in boreal forests. Forest Ecology and Management, 262, 379–388. https://doi.org/10.1016/j.foreco.2011.04.002

Kennedy, C.M., Oakleaf, J.R., Theobald, D.M., Baruch-Mordo, S., Kiesecker, J., 2019. Managing the middle: A shift in conservation priorities based on the global human modification gradient. Global Change Biology, 25(3), 811–826. https://doi.org/10.1111/gcb.14549

Kleinschmit, D., Lindstad, B. H., Thorsen, B. J., Toppinen, A., Roos, A., Baardsen, S. 2014. Shades of green: a social scientific view on bioeconomy in the forest sector. Scandinavian journal of forest research, 29(4), 402–410. https://doi.org/10.1080/02827581.2014.921722

Knorn, J., Kuemmerle, T., Radeloff, V.C., Szabo, A., Mindrescu, M., Keeton, W.S., Abrudan, I., Griffiths, P., Gancz, V., Hostert, P. 2012. Forest restitution and protected area effectiveness in post-socialist Romania. Biological Conservation, 146(1), 204–212. https://doi.org/10.1016/j.biocon.2011.12.020

Knorn, J.A.N., Kuemmerle, T., Radeloff, V.C., Keeton, W.S., Gancz, V., Biriş, I.A., Svoboda, M., Griffiths, P., Hagatis, A., Hostert, P. 2013. Continued loss of temperate old-growth forests in the Romanian Carpathians despite an increasing protected area network. Environmental Conservation, 40(2), 182–193. https://doi.org/10.1017/S0376892912000355

Komonen, A., Müller, J. 2018. Dispersal ecology of deadwood organisms and connectivity conservation. Conservation Biology, 32(3), 535–545. https://doi.org/10.1111/cobi.13087

Kosewska, A., Topa, E., Nietupski, M., Kędzior, R. 2018. Assemblages of carabid beetles (Col. Carabidae) and ground-dwelling spiders (Araneae) in natural and artificial regeneration of pine forests. Community Ecology, 19(2), 156–167. https://doi.org/10.1556/168.2018.19.2.8

Koskela, J., Lefèvre, F., Schueler, S., Kraigher, H., Olrik, D.C., Hubert, J., Longauer R. et al. 2013. Translating conservation genetics into management: Pan-European minimum requirements for dynamic conservation units of forest tree genetic diversity. Biological Conservation 157, 39–49. https://do.org/10.1016/j.biocon.2012.07.023

Kraigher H., Al Sayegh-Petkovšek S. 2011. Mycobioindication of stress in forest ecosystems. In: Rai M., Varma A. (eds.). Diversity and biotechnology of ectomycorrhizae. Soil biology, vol. 25. Heidelberg; New York: Springer, pp 301–322. https://doi.org/10.1007/978-3-642-15196-5_13

Kraus D., Krumm F. (eds.) 2013. Integrative approaches as an opportunity for the conservation of forest biodiversity. European Forest Institute. https://efi.int/sites/default/files/files/publication-bank/projects/integrate_2013.pdf

Kriegel, P., Matevski, D., Schuldt, A. 2021. Monoculture and mixture-planting of non-native Douglas fir alters species composition, but promotes the diversity of ground beetles in a temperate forest system. Biodiversity and Conservation, 30(5), 1479–1499. https://doi.org/10.1007/s10531-021-02155-1

Krumm, F., Schuck, A., Rigling, A. (eds.) (2020). How to balance forestry and biodiversity conservation? – A view across Europe. 640 p. https://www.dora.lib4ri.ch/wsl/islandora/object/wsl%3A25581/ datastream/PDF/Krumm-2020-How_to_balance_forestry_and-%28published_version%29.pdf

Krutilla, J.V. 1967. Conservation Reconsidered. American Economic Review, 57, 777–786.

Kunz J., Löffler G., Bauhus, J. 2018. Minor European broadleaved tree species are more drought-tolerant than *Fagus sylvatica* but not more tolerant than *Quercus petraea*. Forest Ecology and Management 414, 15–27. https://doi.org/10.1016/j.foreco.2018.02.016

Kuuluvainen, T., Lindberg, H., Vanha-Majamaa, I., Keto-Tokoi, P., Punttila, P. 2019. Low-level retention forestry, certification, and biodiversity: Case Finland. Ecological Processes, 8(1), 47. https://doi.org/10.1186/s13717-019-0198-0

Lange, S., Volkholz, J., Geiger, T., Zhao, F., Vega, I., Veldkamp, T., … & Frieler, K.Reyer, C.P., Warszawski, L., Huber, V., Jägermeyr, J. and Schewe, J. (2020). Projecting exposure to extreme climate impact events across six event categories and three spatial scales. Earth's Future, 8(12). https://doi.org/10.1029/2020EF001616

Langmaier, M., Lapin, K. 2020. A systematic review of the impact of invasive alien plants on forest regeneration in European temperate forests. Frontiers in Plant Science, 1349. https://doi.org/10.3389/fpls.2020.524969

Lapin, K., Bacher, S., Cech, T., Damjanić, R. Essl, F., Georges, F. I., … & Stojnic, S. 2021. Comparing environmental impacts of alien plants, insects and pathogens in protected riparian forests. NeoBiota (69). https://doi.org/10.3897/neobiota.69.71651

Larrieu, L., Cabanettes, A. 2012. Species, live status, and diameter are important tree features for diversity and abundance of tree microhabitats in subnatural montane beech-fir forests. Canadian Journal of Forest Research 42(8), 1433–1445. https://doi.org/10.1139/x2012-077

Larrieu, L., Cabanettes, A., Gonin, P., Lachat, T., Paillet, Y., Winter, S., Bouget, C., Deconchat, M. 2014. Deadwood and tree microhabitat dynamics in unharvested temperate mountain mixed forests: a life-cycle approach to biodiversity monitoring. Forest Ecology and Management 334, 163–173. https://doi.org/10.1016/j.foreco.2014.09.007

Larrieu, L., Paillet, Y., Winter, S., Bütler, R., Kraus, D., Krumm, F., Lachat, T., Michel, A. K., Regnery, B., Vandekerkhove, K. 2018. Tree related microhabitats in temperate and Mediterranean European forests: A hierarchical typology for inventory standardization. Ecological Indicators 84: 194–207. https://doi.org/10.1016/j.ecolind.2017.08.051

Larsen, J.B., Angelstam, P., Bauhus, J., Carvalho, J.F., Diaci, J., Dobrowolska, D., Gazda, A., Gustafsson, L., Krumm, F., Knoke, T., Konczal, A., Kuuluvainen, T., Mason, B., Motta, R., Pötzelsberger, E., Rigling, A., Schuck, A., 2022. Closer-to-Nature Forest Management. From Science to Policy 12. European Forest Institute. https://doi.org/10.36333/fs12

Larsson, T.-B., Angelstam, P., Balent, G., Barbati, A., Bijlsma, R.-J., Boncina, A., Bradshaw, R., Bücking, W., Ciancio, O., Corona, P., Diaci, J., Dias, S., Ellenberg, H., Manuel Fernandes, F, Fernandez-Gonzalez, F., Ferris, R., Frank, G., Friis Møller, P., Giller, P.S., Gustafsson, L., Halbritter, K., Hall, S., Hansson, L., Innes, J., Jactel, H., Keannel Dobbertin, M., Klein, M., Marchetti, M., Mohren, F., Niemelä, P., O'Halloran, J., Rametsteiner, E., Rego, F., Scheidegger, C., Scotti, R., Sjöberg, K., Spanos, I., Spanos, K., Standovar, T., Svensson, L., Tømmerås, B.Å., Trakolis, D., Uuttera, J., Van Den Meerschaut, D., Vanderkerkhove, K., Walsh, P.M., Watt, A.D. 2001. Biodiversity evaluation tools for European forests. Ecological Bulletins 50. 236 p.

Lassauce, A., Paillet, Y., Jactel, H., Bouget, C. 2011. Deadwood as a surrogate for forest biodiversity: meta-analysis of correlations between deadwood volume and species richness of saproxylic organisms. Ecological Indicators, 11(5), 1027–1039. https://doi.org/10.1016/j.ecolind.2011.02.004

Latacz-Lohmann, U., Schilizzi, S. 2005. Auctions for conservation contracts: a review of the theoretical and empirical literature. Report to the Scottish Executive Environment and Rural Affairs Department. 101 p.

Leempoel, K., Hebert, T., Hadly, E.A. 2020. A comparison of eDNA to camera trapping for assessment of terrestrial mammal diversity. Proceedings of the Royal Society B, 287(1918), 20192353. https://doi.org/10.1098/rspb.2019.2353

Lin, M., Horowitz, L.W., Xie, Y., Paulot, F., Malyshev, S., Shevliakova, E., Finco, A., Gerosa, G., Kubistin, D. and Pilegaard, K. 2020. Vegetation feedbacks during drought exacerbate ozone air pollution extremes in Europe. Nature Climate Change, 10(5), 444–451 https://doi.org/10.1038/s41558-020-0743-y

Lindeijer, E. 2000. Review of land use impact methodologies. Journal of Cleaner Production, 8(4), 273–281. https://doi.org/10.1016/S0959-6526(00)00024-X

Liquete, C., Kleeschulte, S., Dige, G., Maes, J., Grizzetti, B., Olah, B., Zulian, G. 2015. Mapping green infrastructure based on ecosystem services and ecological networks: A Pan-European case study. Environmental Science & Policy, 54, 268–280. https://doi.org/10.1016/j.envsci.2015.07.009

Löfqvist, S., Ghazoul, J. 2019. Private funding is essential to leverage forest and landscape restoration at global scales. Nature ecology & evolution, 3(12), 1612–1615. https://doi.org/10.1038/s41559-019-1031-y

Loft, L., Mann, C., Hansjürgens, B. 2015. Challenges in ecosystem services governance: Multi-levels, multi-actors, multi-rationalities. Ecosystem Services, 16, 150–157. https://doi.org/10.1016/j.ecoser.2015.11.002

López-Bedoya, P. A., Magura, T., Edwards, F. A., Edwards, D. P., Rey-Benayas, J. M., Lövei, G. L., Noriega, J. A. 2021. What level of native beetle diversity can be supported by forestry plantations? A global synthesis. Insect Conservation and Diversity, 14(6), 736–747. https://doi.org/10.1111/icad.12518

Lovelock, J.E., Margulis, L. 1974. Atmospheric homeostasis by and for the biosphere: the Gaia hypothesis. Tellus, 26(1–2), 2–10. https://doi.org/10.3402/tellusa.v26i1-2.9731

MacArthur, R. H., MacArthur, J. W. 1961. On bird species diversity. Ecology, 42(3), 594–598. https://doi.org/10.2307/1932254

Maes, W.H., Fontaine, M., Rongé, K., Hermy, M., Muys, B. 2011. A quantitative indicator framework for stand level evaluation and monitoring of environmentally sustainable forest management. Ecological Indicators 11: 468–479. https://doi.org/10.1016/j.ecolind.2010.07.001

Martín-Forés, I., Magro, S., Bravo-Oviedo, A., Alfaro-Sánchez, R., Espelta, J. M., Frei, T., Valdés-Correcher, E., Rodríguez Fernández-Blanco, C., Winkel, G., Gerzabek, G., Hampe, A., Valladares, F. & González-Martínez, S. C. 2020. Spontaneous forest regrowth in South-West Europe: Consequences for nature's contributions to people. People and Nature. DOI: https://doi.org/10.1002/pan3.10161

Mansourian, S., Parrotta, J., Balaji, P., Bellwood-Howard, I., Bhasme, S., Bixler, R. P., ... & Yang, A. 2020. Putting the pieces together: integration for forest landscape restoration implementation. Land Degradation & Development, 31(4), 419–429. https://doi.org/10.1002/ldr.3448

Mauerhofer, V., Ichinose, T., Blackwell, B. R., Willig, M.R., Flint, C.G., Krause, M.S., Penker, M. 2018. Underuse of social-ecological systems: A research agenda for addressing challenges to biocultural diversity. Land Use Policy 72, 57–64. https://doi.org/10.1016/j.landusepol.2017.12.003

McDonald, G.T., Lane, M.B. 2004. Converging global indicators for sustainable forest management. Forest Policy and Economics 6, 63–70. https://doi.org/10.1016/S1389-9341(02)00101-6

McElhinny, C., Gibbons, P., Brack, C., Bauhus, J. 2005. Forest and woodland stand structural complexity: Its definition and measurement. Forest Ecology and Management 218, 1–24. https://doi.org/10.1016/j.foreco.2005.08.034

McKenney, B.A., Kiesecker, J.M. 2010. Policy Development for Biodiversity Offsets: A Review of Offset Frameworks. Environmental Management, 45, 165–176. https://doi.org/10.1007/s00267-009-9396-3

MEA 2005. Ecosystems and human well-being. Millennium Ecosystem Assessment, Island Press, Washington DC.

Messier C, Bauhus J, Sousa-Silva R, et al. (2021) For the sake of resilience and multifunctionality, let's diversify planted forests! Conservation Letters, 15:e12829. https://doi.org/10.1111/conl.12829

Mickwitz, P. 2003. A framework for evaluating environmental policy instruments: context and key concepts. Evaluation, 9(4), 415–436. https://doi.org/10.1177/135638900300900404

Miljand, M., Bjärstig, T., Eckerberg, K., Primmer, E., Sandström, C. 2021. Voluntary agreements to protect private forests–A realist review. Forest Policy and Economics, 128, 102457. https://doi.org/10.1016/j.forpol.2021.102457

Mitchell, R., Chitanava, S., Dbar, R., Kramarets, V., Lehtijärvi, A., Matchutadze, I., ... & Kenis, M. 2018. Identifying the ecological and societal consequences of a decline in Buxus forests in Europe and the Caucasus. Biological Invasions, 20(12), 3605–3620. https://doi.org/10.1007/s10530-018-1799-8

Mölder, A., Tiebel, M., Plieninger, T. 2021. On the interplay of ownership patterns, biodiversity, and conservation in past and present temperate forest landscapes of Europe and North America. Current Forestry Reports, 7, 195–213. https://doi.org/10.1007/s40725-021-00143-w

Moning, C., Müller, J. 2009. Critical forest age thresholds for the diversity of lichens, molluscs and birds in beech (*Fagus sylvatica* L.) dominated forests. Ecological indicators, 9(5), 922–932. https://doi.org/10.1016/j.ecolind.2008.11.002

Müller, J., Jarzabek-Müller, A., Bussler, H., Gossner, M.M. 2014. Hollow beech trees identified as keystone structures by analyses of functional and phylogenetic diversity of saproxylic beetles. Animal Conservation 17, 154–162. https://doi.org/10.1111/acv.12075

Müller, J., Bütler, R. 2010. A review of habitat thresholds for dead wood: a baseline for management recommendations in European forests. European Journal of Forest Research, 129(6), 981–992. https://doi.org/10.1007/s10342-010-0400-5

Muñoz-Rojas, J., Nijnik, M., González-Puente, M., Cortines-García, F. 2015. Synergies and conflicts in the use of policy and planning instruments for implementing forest and woodland corridors and networks; a case study in NE Scotland. Forest Policy and Economics, 57, 47–64. https://doi.org/10.1016/j.forpol.2015.05.002

Myllyviita, T., Sironen, S., Saikku, L., Holma, A., Leskinen, P., Palme, U. 2019. Assessing biodiversity impacts in life cycle assessment framework-Comparing approaches based on species richness and ecosystem indicators in the case of Finnish boreal forests. Journal of Cleaner Production, 236, 117641. https://doi.org/10.1016/j.jclepro.2019.117641

Nabuurs, G.J., Mrabet, R., Abu Hatab, A., Bustamante, M., Clark, H., Havlík, P., House, J., Mbow, C., Ninan, K. N., Popp, A., Roe, S., Sohngen, B., Towprayoon, S., Ayala-Niño, F., Emmet-Booth, J. 2022. Agriculture, Forestry and other Land Uses. Chapter 7 of the IPCC Working Group III Report. In: Skea, J., & Shukla, P. (Eds.) Mitigation. IPCC, Working Group III Report to the Sixth Assessment Cycle. Geneva. Cambridge University Press. 185 p.

Nagel, T.A., Firm, D., Mihelic, T., Hladnik, D., de Groot, M., Rozenbergar, D. 2017. Evaluating the influence of integrative forest management on old-growth habitat structures in a temperate forest region. Biological Conservation, 216, 101–107. https://doi.org/10.1016/j.biocon.2017.10.008

Namkoong, G., Boyle, T., Gregorious, H.R., Joly, H., Savolainen, O., Ratman, W. et al. 1996. Testing criteria and indicators for assessing the sustainability of forest management: genetic criteria and indicators. Centre for International Forestry Research (CIFOR) Working paper No. 10. Bogor, Indonesia.

Naumov, V., Manton, M., Elbakidze, M., Rendenieks, Z., Priednieks, J., Uhlianets, S., ... & Angelstam, P. 2018. How to reconcile wood production and biodiversity conservation? The Pan-European boreal forest history gradient as an "experiment". Journal of Environmental Management, 218, 1–13. https://doi.org/10.1016/j.jenvman.2018.03.095

Nichiforel, L., Deuffic, P., Thorsen, B. J., Weiss, G., Hujala, T., Keary, K., Lawrence, A., Avdibegovic, M., Dobšinská, Z., Feliciano, D., Mifsud, E.G., Hoogstra-Klein, M., Hrib, M. et al. 2020. Two decades of forest-related legislation changes in European countries analysed from a property rights perspective. Forest Policy and Economics, 115, 102146. https://doi.org/10.1016/j.forpol.2020.102146

Niemelä, J. 2001. Carabid beetles (Coleoptera: Carabidae) and habitat fragmentation: a review. European Journal of Entomology, 98(2), 127–132. https://doi.org/10.14411/eje.2001.023

Niemelä, J., Young, J., Alard, D., Askasibar, M., Henle, K., Johnson, R., ... & Watt, A. 2005. Identifying, managing and monitoring conflicts between forest biodiversity conservation and other human interests in Europe. Forest Policy and Economics, 7(6), 877–890. https://doi.org/10.1016/j.forpol.2004.04.005

Nijnik, M., Secco, L., Miller, D., Melnykovych, M. 2019. Can social innovation make a difference to forest-dependent communities? Forest Policy and Economics, 100, 207–213. https://doi.org/10.1016/j.forpol.2019.01.001

Nordén, B., Dahlberg, A., Brandrud, T.E., Fritz, Ö., Ejrnaes, R., Ovaskainen, O. 2014. Effects of ecological continuity on species richness and composition in forests and woodlands: a review. Ecoscience, 21(1), 34–45. https://doi.org/10.2980/21-1-3667

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. Conserv Biol 4(4):355–364. https://www.jstor.org/stable/2385928

Ódor, P., Heilmann-Clausen, J., Christensen, M., Aude, E., van Dort, K.W., Piltaver, A., Siller, I., Veerkamp, M.T., Walleyn, R., Standovár, T., van Hees, A.F.M., Kosec, J., Matočec, N., Kraigher, H., Grebenc, T. 2006. Diversity of dead wood inhabiting fungi and bryophytes in semi-natural beech forests in Europe. Biological Conservation 131: 58–71. https://doi.org/10.1016/j.biocon.2006.02.004

OECD 2020. A Comprehensive Overview of Global Biodiversity Finance.

Oettel, J., Lapin, K. 2021. Linking forest management and biodiversity indicators to strengthen sustainable forest management in Europe. Ecological Indicators, 122, 107275. https://doi.org/10.1016/j.ecolind.2020.107275

Opdam, P., Foppen, R., Vos, C. 2001. Bridging the gap between ecology and spatial planning in landscape ecology. Landscape ecology, 16(8), 767–779. https://doi.org/10.1023/A:1014475908949

Paavola, J., Primmer, E. 2019. Governing the provision of insurance value from ecosystems. Ecological Economics, 164, 106346. https://doi.org/10.1016/j.ecolecon.2019.06.001

Paillet, Y., Bergès, L., Hjältén, J., Ódor, P., Avon, C., Bernhardt-Römermann,... & Virtanen, R. 2010. Biodiversity differences between managed and unmanaged forests: Meta-analysis of species richness in Europe. Conservation Biology, 24(1), 101–112. https://doi.org/10.1111/j.1523-1739.2009.01399.x

Palahí, M., Pantsar, M., Costanza, R., Kubiszewski, I., Potočnik, J., Stuchtey, M., ... & Holmgren, J. L. B. 2020. Investing in Nature to Transform the Post COVID-19 Economy: A 10-point Action Plan to create a circular bioeconomy devoted to sustainable wellbeing. The Solutions Journal, 11(2). https://thesolutionsjournal.com/2020/05/14/investing-in-nature-to-transform-the-post-covid-19-economy-a-10-point-action-plan-to-create-a-circular-bioeconomy-devoted-to-sustainable-wellbeing/

Palik, B.J., D'Amato, A.W. 2017. Ecological forestry: much more than retention harvesting. Journal of Forestry, 115(1), 51–53. https://doi.org/10.5849/jof.16-057

Paul, C., Knoke, T. 2015. Between land sharing and land sparing – what role remains for forest management and conservation? International Forestry Review, 17(2), 210–230. https://doi.org/10.1505/146554815815500624

Pennisi, E., 2021. Getting the big picture of biodiversity. Science, 374(6570), 926–931. https://doi.org/10.1126/science.acx9637

Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H. G., Scholes, R. J., ... & Wegmann, M. 2013. Essential biodiversity variables. Science, 339(6117), 277–278. https://doi.org/10.1126/science.1229931

Peterken, G. F. 1996. Natural woodland: ecology and conservation in northern temperate regions. Cambridge University Press.

Pettorelli, N., Schulte to Bühne, H., Tulloch, A., Dubois, G., Macinnis-Ng, C., Queirós, A.M., Keith, D.A., Wegmann, M., Schrodt, F., Stellmes, M., Sonnenschein, R. et al. 2018. Satellite remote sensing of ecosystem functions: opportunities, challenges and way forward. Remote Sensing in Ecology and Conservation, 4(2), 71–93. https://doi.org/10.1002/rse2.59

Pickett, S. T., Thompson, J. N. 1978. Patch dynamics and the design of nature reserves. Biological conservation 13(1), 27–37. https://doi.org/10.1016/0006-3207(78)90016-2

Pignatti, S., Menegoni, P., Pietrosanti, S. 2005. Bioindicator values of vascular plants of the Flora of Italy. Braun-Blanquetia, 39, 3–95.

Pimentel, D., Westra, L., Noss, R. F. (ed.). 2000. Ecological integrity: Integrating environment, conservation, and health. Island Press.

Pommerening, A., Murphy, S.T. 2004. A review of the history, definitions and methods of continuous cover forestry with special attention to afforestation and restocking. Forestry, 77(1), 27-44. https://doi.org/10.1093/forestry/77.1.27

Pötzelsberger, E., Bauhus, J., Muys, B., Wunder, S., Bozzano, M, Farsakoglou, A.M., Schuck, A., Lindner, M., Lapin, K. 2021. Forest biodiversity in the spotlight. European Forest Institute. https://doi.org/10.36333/rs2

Pötzelsberger, E., Spiecker, H., Neophytou, C., Mohren, F., Gazda, A., Hasenauer, H. 2020. Growing Non-native Trees in European Forests Brings Benefits and Opportunities but Also Has Its Risks and Limits. Current Forestry Reports 6, 339–353. https://doi.org/10.1007/s40725-020-00129-0

Pretzsch, H., Schütze, G. 2009. Transgressive overyielding in mixed compared with pure stands of Norway spruce and European beech in Central Europe: evidence on stand level and explanation on individual tree level. European Journal of Forest Research, 128(2), 183–204. https://doi.org/10.1007/s10342-008-0215-9

Primicia, I., Camarero, J.J., de Aragón, J.M., de-Miguel, S. Bonet, J.A. 2016. Linkages between climate, seasonal wood formation and mycorrhizal mushroom yields. Agricultural and Forest Meteorology, 228, 339–348. https://doi.org/10.1016/j.agrformet.2016.07.013

Primmer, E. 2011. Policy, project and operational networks: channels and conduits for learning in forest biodiversity conservation. Forest Policy and Economics, 13(2), 132–142. https://doi.org/10.1016/j.forpol.2010.06.006

Primmer, E., Jokinen, P., Blicharska, M., Barton, D. N., Bugter, R., Potschin, M. 2015. Governance of ecosystem services: a framework for empirical analysis. Ecosystem services, 16, 158–166. https://doi.org/10.1016/j.ecoser.2015.05.002

Primmer, E., Paloniemi, R., Similä, J. Tainio, A. 2014. Forest owner perceptions of institutions and voluntary contracting for biodiversity conservation: not crowding out but staying out. Ecological Economics, 103, 1–10. https://doi.org/10.1016/j.ecolecon.2014.04.008

Primmer, E., Saarikoski, H., Vatn, A. 2018. An empirical analysis of institutional demand for valuation knowledge. Ecological Economics, 152, 152–160. https://doi.org/10.1016/j.ecolecon.2018.05.017

Primmer, E., Varumo, L., Krause, T., Orsi, F., Geneletti, D., Brogaard, S., Aukes, E., Ciolli, M., Grossmann, C., Hernández-Morcillo, M., Kister, J., Kluvánková, T., Loft, L., Maier, C., Meyer, C., Schleyer, C., Spacek, M., Mann, C. 2021. Mapping Europe's institutional landscape for forest ecosystem service provision, innovations and governance. Ecosystem Services, 47, 101225. https://doi.org/10.1016/j.ecoser.2020.101225

Prober, S.M., Byrne, M., McLean, E.H., Steane, D.A., Potts, B.M., Vaillancourt, R.E., Stock, W.D. 2015. Climate-adjusted provenancing: a strategy for climate-resilient ecological restoration. Frontiers in Ecology and Evolution 3, 65. https://doi.org/10.3389/fevo.2015.00065

Pulla, P., Schuck, A., Verkerk, P. J., Lasserre, B., Marchetti, M., Green, T. 2013. Mapping the distribution of forest ownership in Europe. EFI Technical Report 88. European Forest Institute.

Ranius, T., Roberge, J. M. 2011. Effects of intensified forestry on the landscape-scale extinction risk of dead wood dependent species. Biodiversity and Conservation, 20(13), 2867–2882. https://doi.org/10.1007/s10531-011-0143-8

Ratkiewicz, M., Matosiuk, M., Kowalczyk, R., Konopiński, M.K., Okarma, H., Ozolins, J., Männil, P., Ornicans, A., Schmidt, K. 2012. High levels of population differentiation in E urasian lynx at the edge of the species' western range in Europe revealed by mitochondrial DNA analyses. Animal Conservation, 15(6), 603–612. https://doi.org/10.1111/j.1469-1795.2012.00556.x

Reed, D.H., O'Grady, J.J., Brook, B.W., Ballou, J.D., Frankham, R. 2003. Estimates of minimum viable population sizes for vertebrates and factors influencing those estimates. Biological Conservation 113(1), 23–34. https://doi.org/10.1016/S0006-3207(02)00346-4

Reich, R. 2018. Just Giving: Why Philanthropy Is Failing Democracy and How It Can Do Better. Princeton University Press, Princeton, New Jersey, USA, 239p.

Renner, S. S., Zohner, C. M. 2018. Climate change and phenological mismatch in trophic interactions among plants, insects, and vertebrates. Annual Review of Ecology, Evolution, and Systematics, 49, 165–182. https://doi.org/10.1146/annurev-ecolsys-110617-062535

Rhodes, R. A. 2007. Understanding governance: Ten years on. Organization studies, 28(8), 1243–1264. https://doi.org/10.1177%2F0170840607076586

Rivers, M.C., Beech, E., Bazos, I., Bogunić, F., Buira, A., Caković, D., Carapeto, A., Carta, A., Cornier, B., Fenu, G., Fernandes, F., Fraga, P., Garcia Murillo, P.J., Lepší, M., Matevski, V., Medina, F.M., Menezes de Sequeira, M., Meyer, N., Mikoláš, V., Montagnani, C., Monteiro-Henriques, T., Naranjo Suárez, J., Orsenigo, S., Petrova, A., Reyes-Betancort, J.A., Rich, T., Salvesen, P.H., Santana López, I., Scholz, S., Sennikov, A., Shuka, L., Silva, L.F., Thomas, P., Troia, A., Villar, J.L. and Allen, D.J. 2019 European Red List of Trees. IUCN, Cambridge, UK and Brussels, Belgium.

Rodríguez, A., Hekkala, A. M., Sjögren, J., Strengbom, J., Löfroth, T. 2021. Boreal forest fertilization leads to functional homogenization of ground beetle assemblages. Journal of Applied Ecology, 58(6), 1145–1154. https://doi.org/10.1111/1365-2664.13877

Roleček, J., Řepka, R. 2020). Formerly coppiced old growth stands act as refugia of threatened biodiversity in a managed steppic oak forest. Forest Ecology and Management, 472, 118245. https://doi.org/10.1016/j.foreco.2020.118245

Rolfe, J., Whitten, S., Windle, J. 2017. The Australian experience in using tenders for conservation. Land Use Policy, 63, 611–620. https://doi.org/10.1016/j.landusepol.2015.01.037

Roques, A., Shi, J., Auger-Rozenberg, M. A., Ren, L., Augustin, S., Luo, Y. Q. 2020. Are invasive patterns of non-native insects related to woody plants differing between Europe and China? Frontiers in Forests and Global Change, 91. https://doi.org/10.3389/ffgc.2019.00091

Rosenvald, R., Lõhmus, A. 2008. For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. Forest Ecology and Management, 255(1), 1–15448, 543–548. https://doi.org/10.1016/j.foreco.2007.09.016

Ruiz-Benito, P., Ratcliffe, S., Jump, A.S., Gómez-Aparicio, L., Madrigal-González, J., Wirth, C., Kändler, G., Lehtonen, A., Dahlgren, J., Kattge, J., Zavala, M.A. 2017. Functional diversity underlies demographic responses to environmental variation in European forests. Global Ecology and Biogeography, 26(2), 128–141. https://doi.org/10.1111/geb.12515

Saarikoski, H., Primmer, E., Saarela, S-R., Antunes, P., Aszalós, R., Baró, F., Berry P., Garcia Blanco, G., Goméz-Baggethun, E., Carvalho, L., Dick, J., Dunford, R., Hanzu, M., Harrison, P., Izakovicova, Z., Kertész, M., Kopperoinen, L., Köhler, B., Langemeyer, J., Lapola, D., Liquete, C., Luque, S., Mederly, P., Niemela, J., Palomo, I., Martinez Pastur, G., Peri, P., Preda, E., Priess, J., Santos, R., Schleyer, C., Turkelboom, F., Vadineanu, A., Verheyden, W., Vikström, S., Young, J. 2018. Institutional challenges in putting ecosystem service knowledge in practice. Ecosystem Services 29, C, 579–598. https://doi.org/10.1016/j.ecoser.2017.07.019

Sabatini, F.M., Burrascano, S., Keeton, W.S. et al. 2018. Where are Europe's last primary forests? Diversity and Distributions 24, 1426–1439. https://doi.org/10.1111/ddi.12778

Sabatini, F.M., Bluhm, H., Kun, Z., Aksenov, D., Atauri, J.A., Buchwald, E., Burrascano, S., Cateau, E., Diku, A., Duarte, I.M., Fernández López, Á.B., Garbarino, M., Grigoriadis, N., Horváth, F., Keren, S., Kitenberga, M., Kiš, A., Kraut, A., Ibisch, P.L., Larrieu, L., Lombardi, F., Matovic, B., Melu, R.N., Meyer, P., Midteng, R., Mikac, S., Mikoláš, M., Mozgeris, G., Panayotov, M., Pisek, R., Nunes, L., Ruete, A., Schickhofer, M., Simovski, B., Stillhard, J., Stojanovic, D., Szwagrzyk, J., Tikkanen, O.P., Toromani, E., Volosyanchuk, R., Vrška, T., Waldherr, M., Yermokhin, M., Zlatanov, T., Zagidullina, A., Kuemmerle, T. 2021. European primary forest database v2. 0. Scientific data, 8(1), 1–14. https://doi.org/10.1038/s41597-021-00988-7

Sagoff, M. 2008. On the economic value of ecosystem services. Environmental values, 17(2), 239–257. https://www.jstor.org/stable/30302640

Sandom, C.J., Ejrnæs, R., Hansen, M.D., Svenning, J.C. 2014. High herbivore density associated with vegetation diversity in interglacial ecosystems. Proceedings of the National Academy of Sciences, 111(11), 4162–4167. https://doi.org/10.1073/pnas.1311014111

Sandström, J., Bernes, C., Junninen, K., Lõhmus, A., Macdonald, E., Müller, J., Jonsson B. G. 2019. Impacts of dead wood manipulation on the biodiversity of temperate and boreal forests. A systematic review. Journal of Applied Ecology 56, 1770–1781. https://doi.org/10.1111/1365-2664.13395

Santini, A. and Battisti, A. 2019 Complex Insect–Pathogen Interactions in Tree Pandemics. Frontiers in Physiology 10, 550. https://doi.org/10.3389/fphys.2019.00550

Schall, P., Gossner, M. M., Heinrichs, S., Fischer, M., Boch, S., Prati, D., ... & Ammer, C. 2018. The impact of even-aged and uneven-aged forest management on regional biodiversity of multiple taxa in European beech forests. Journal of Applied Ecology, 55(1), 267–278. https://doi.org/10.1111/1365-2664.12950

Scheffers, B.R., De Meester, L., Bridge, T.C., Hoffmann, A.A., Pandolfi, J.M., Corlett, R.T., Butchart, S.H., Pearce-Kelly, P., Kovacs, K.M., Dudgeon, D., Pacifici, M., Rondinini, C., Foden, W. B., Martin, T. G., Mora, C., Bickford, D., Watson, J. E. 2016. The broad footprint of climate change from genes to biomes to people. Science, 354, 6313. https://doi.org/10.1126/science.aaf7671

Schick, A., Porembski, S., Hobson, P.R., Ibisch, P.L. 2019. Classification of key ecological attributes and stresses of biodiversity for ecosystem-based conservation assessments and management. Ecological Complexity 38: 98–111. https://doi.org/10.1016/j.ecocom.2019.04.001

Schirkonyer, U., Bauer, C., Rothe, G.M. 2013. Ectomycorrhizal diversity at five different tree species in forests of the Taunus Mountains in Central Germany. Open Journal of Ecology, 3, 66–81. http://dx.doi.org/10.4236/oje.2013.31009

Schmitz, A., Sanders, T.G., Bolte, A., Bussotti, F., Dirnböck, T., Johnson, J., Peñuelas, J., Pollastrini, M., Prescher, A.K., Sardans, J., Verstraeten, A. 2019. Responses of forest ecosystems in Europe to decreasing nitrogen deposition. Environmental Pollution, 244, 980–994. https://doi.org/10.1016/j.envpol.2018.09.101

Schuldt, B., Buras, A., Arend, M., Vitasse, Y., Beierkuhnlein, C., Damm, A., ... & Kahmen, A. 2020. A first assessment of the impact of the extreme 2018 summer drought on Central European forests. Basic and Applied Ecology, 45, 86–103. https://doi.org/10.1016/j.baae.2020.04.003

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., ... Essl, F. 2017. No saturation in the accumulation of alien species worldwide. Nature communications, 8(1), 1–9. https://doi.org/10.1038/ncomms14435

Seibold, S., Bässler, C., Brandl, R., Büche, B., Szallies, A., Thorn, S., Ulyshen, M.D. and Müller, J. 2016. Microclimate and habitat heterogeneity as the major drivers of beetle diversity in dead wood. Journal of Applied Ecology, 53(3), 934–943. https://doi.org/10.1111/1365-2664.12607

Seibold, S., C. Bässler, R. Brandl, L. Fahrig, B. Förster, M. Heurich, T. Hothorn, F. Scheipl, S. Thorn, and J. Müller. 2017. An experimental test of the habitat-amount hypothesis for saproxylic beetles in a forested region. Ecology 98,1613–1622. https://doi.org/10.1002/ecy.1819

Seibold, S., Gossner, M. M., Simons, N. K., Blüthgen, N., Müller, J., Ambarlı, D., ... Weisser, W. W. (2019). Arthropod decline in grasslands and forests is associated with landscape-level drivers. Nature, 574(7780), 671–674. https://doi.org/10.1038/s41586-019-1684-3

Seidl, R., Thom, D., Kautz, M., Martin-Benito, D., Peltoniemi, M., Vacchiano, G., Wild, J., Ascoli, D., Petr, M., Honkaniemi, J., Lexer, M.J. 2017. Forest disturbances under climate change. Nature climate change, 7(6), 395–402. https://doi.org/10.1038/nclimate3303

Senf, C., Buras, A., Zang, C. S., Rammig, A., Seidl, R. 2020. Excess forest mortality is consistently linked to drought across Europe. Nature Communications, 11(1), 1–8. https://doi.org/10.1038/s41467-020-19924-1

Senf, C., Seidl, R. 2021a. Persistent impacts of the 2018 drought on forest disturbance regimes in Europe. Biogeosciences, 18(18), 5223–5230. https://doi.org/10.5194/bg-18-5223-2021

Senf, C., Seidl, R. 2021b. Storm and fire disturbances in Europe: distribution and trends. Global change biology, 27(15), 3605–3619. https://doi.org/10.1111/gcb.15679

Seymour, R.S., Hunter, M.L. Jr. 1992. New Forestry in Eastern Spruce-Fir Forests: Principles and Applications to Maine. Maine Agricultural and Forest Experimental Station 716, Orono, Maine.

Sicacha-Parada, J., Steinsland, I., Cretois, B., Borgelt, J., 2021. Accounting for spatial varying sampling effort due to accessibility in Citizen Science data: A case study of moose in Norway. Spatial Statistics, 42, 100446. https://doi.org/10.1016/j.spasta.2020.100446

Song, X. P., Hansen, M. C., Stehman, S. V., Potapov, P. V., Tyukavina, A., Vermote, E. F., Townshend, J. R. 2018. Global land change from 1982 to 2016. Nature, 560(7720), 639–643. https://doi.org/10.1038/s41586-018-0411-9

Sousa-Silva, R., Verheyen, K., Ponette, Q., Bay, E., Sioen, G., Titeux, H., Van de Peer, T., Van Meerbeek, K., Muys, B. 2018. Tree diversity mitigates defoliation after a drought-induced tipping point. Global Change Biology 24, 9, 4304–4315. https://doi.org/10.1111/gcb.14326

Spash, C. 2021. Conceptualising Nature: From Dasgupta to Degrowth. Environmental Values. 30 (3), 265–275. https://doi.org/10.3197/096327121X16141642287700

Steffen, W., Richardson, K., Rockström, J., Cornell, S.E., Fetzer, I., Bennett, E.M., Biggs, R., Carpenter, S.R., De Vries, W., De Wit, C.A., Folke, C. 2015. Planetary boundaries: Guiding human development on a changing planet. Science, 347(6223). https://doi.org/10.1126/science.1259855

Stemmelen, A., Jactel, H., Brockerhoff, E., Castagneyrol, B. 2022. Meta-analysis of tree diversity effects on the abundance, diversity and activity of herbivores' enemies. Basic and Applied Ecology, 58, 130–138. https://doi.org/10.1016/j.baae.2021.12.003

Stephens, P.A., Pettorelli, N., Barlow, J., Whittingham, M.J., Cadotte, M.W. 2015. Management by proxy? The use of indices in applied ecology. Journal of Applied Ecology, 52(1), 1–6. https://www.jstor.org/stable/43868379

Stephens, S.S., Wagner, M.R. 2007. Forest plantations and biodiversity: a fresh perspective. Journal of Forestry, 105(6), 307–313. https://doi.org/10.1093/jof/105.6.307

Sterling, E. J., Betley, E., Sigouin, A., Gomez, A., Toomey, A., Cullman, G., Malone, C., Pekor, Arengo, F., Blair, M., Filardi, C., Landrigan, K., Porzecanski, A. L. 2017. Assessing the evidence for stakeholder engagement in biodiversity conservation. Biological conservation, 209, 159–171. https://doi.org/10.1016/j.biocon.2017.02.008

Stokland, J., Siitonen, J.,Jonsson, B.G. 2012. Biodiversity in Dead Wood. Cambridge University Press.

Stokland, J.N. 2001. The coarse woody debris profile: an archive of recent forest history and an important biodiversity indicator. Ecological Bulletins, 49, 71–83. https://www.jstor.org/stable/20113265

Strengbom, J., Nordin, A. 2008. Commercial forest fertilization causes long-term residual effects in ground vegetation of boreal forests. Forest Ecology and Management, 256(12), 2175–2181. https://doi.org/10.1016/j.foreco.2008.08.009

Strengbom, J., Nordin, A., Näsholm, T., Ericson, L. 2001. Slow recovery of boreal forest ecosystem following decreased nitrogen input. Functional Ecology, 15(4), 451–457. https://www.jstor.org/stable/826665

Strubbe, D., Matthysen, E. 2007. Invasive ring-necked parakeets Psittacula krameri in Belgium: habitat selection and impact on native birds. Ecography, 30(4), 578–588. https://doi.org/10.1111/j.0906-7590.2007.05096.x

Sullivan, T. P., Sullivan, D. S. 2018. Influence of nitrogen fertilization on abundance and diversity of plants and animals in temperate and boreal forests. Environmental Reviews, 26(1), 26–42. https://doi.org/10.1139/er-2017-0026

Svancara, L. K., Brannon J. R., Scott, M., Groves, C. R., Noss, R. F., Pressey, R. L. 2005. Policy-driven versus evidence-based conservation: a review of political targets and biological needs. BioScience 55(11), 989–995. https://doi.org/10.1641/0006-3568

Svensson, J., Andersson, J., Sandström, P., Mikusiński, G., Jonsson, B.G. 2019. Landscape trajectory of natural boreal forest loss as an impediment to green infrastructure. Conservation Biology 33(1), 152–163. https://doi.org/10.1111/cobi.13148

Svoboda, M., Verkerk, P.J., Bauhus, J., Bruelheide, H., Burrascano, S., Debaive, N., Duarte, I., Garbarino, M., Grigoriadis, N., Lombardi, F., Mikoláš, M., Meyer, P., Motta, R., Mozgeris, G., Nunes, L., Ódor, P., Panayotov, M., Ruete, A., Simovski, B., Stillhard, J., Svensson, J., Szwagrzyk, J., Tikkanen, O., Vandekerkhove, K., Volosyanchuk, R., Vrska, T., Zlatanov, T., Kuemmerle, T. 2020. Protection gaps and restoration opportunities for primary forests in Europe. Diversity and Distributions 26(12), 1646–1662. https://doi.org/10.1111/ddi.13158

Taccoen, A., Piedallu, C., Seynave, I., Gégout-Petit, A., Nageleisen, L.M., Bréda, N., Gégout, J.C. 2021. Climate change impact on tree mortality differs with tree social status. Forest Ecology and Management, 489, 119048. https://doi.org/10.1016/j.foreco.2021.119048

Taylor, A. R., Victorsson, J. 2016. Short-term effects of stump harvesting on millipedes and centipedes on coniferous tree stumps. Forest Ecology and Management, 371, 67–74. https://doi.org/10.1016/j.foreco.2016.03.039

Tear, T. H., Kareiva, P., Angermeier, P.L., Comer, P., Czech, B., Kautz, R., Landon, L., Mehlman, D., Murphy, K., Ruckelshaus, M, Scott J.M., Wilhere, G. 2005. How much is enough? The recurrent problem of setting measurable objectives in conservation. BioScience 55(10), 835–849. https://doi.org/10.1641/0006-3568

TEEB 2010. The Economics of Ecosystems and Biodiversity Ecological and Economic Foundations. Earthscan: London and Washington.

Thorn, S., Bässler, C., Brandl, R., Burton, P.J., Cahall, R., Campbell, J.L., Castro, J., Choi, C.Y., Cobb, T., Donato, D.C.,Durska, E. 2018. Impacts of salvage logging on biodiversity: A meta-analysis. Journal of Applied Ecology, 55(1), 279–289. https://doi.org/10.1111/1365-2664.12945

Thorn, S., Chao, A., Georgiev, K.B., Müller, J., Bässler, C., Campbell, J.L., Castro, J., Chen, Y.H., Choi, C.Y., Cobb, T.P., Donato, D.C. 2020. Estimating retention benchmarks for salvage logging to protect biodiversity. Nature communications, 11(1), 1–8. https://doi.org/10.1038/s41467-020-18612-4

Turner, M. G. 1989. Landscape ecology: the effect of pattern on process. Annual review of ecology and systematics, 20(1), 171–197. https://doi.org/10.1146/annurev.es.20.110189.001131

Turnhout, E., Dewulf, A., Hulme, M. 2016. What does policy-relevant global environmental knowledge do? The cases of climate and biodiversity. Current Opinion in Environmental Sustainability, 18, 65–72. https://doi.org/10.1016/j.cosust.2015.09.004

Udvardy, M. F. 1959. Notes on the ecological concepts of habitat, biotope and niche. Ecology, 725–728.

Van de Peer, T., Verheyen, K., Ponette, Q., Setiawan, N.N., Muys, B. 2018. Overyielding in young tree plantations is driven by local complementarity and selection effects related to shade tolerance. Journal of Ecology 106 (3), 1096–1105. https://doi.org/10.1111/1365-2745.12839

Van der Plas, F., Manning, P., Allan, E., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M.A., Hector, A., Ampoorter, E., Baeten, L. and Barbaro, L. et al., 2016. Jack-of-all-trades effects drive biodiversity–ecosystem multifunctionality relationships in European forests. Nature communications, 7(1), 1–11.

van Dobben, H. F., de Vries, W. 2017. The contribution of nitrogen deposition to the eutrophication signal in understorey plant communities of European forests. Ecology and Evolution, 7(1), 214–227. https://doi.org/10.1002/ece3.2485

Van Meerbeek, K., Muys, B., Schowanek, S. D., Svenning, J. C. 2019. Reconciling Conflicting Paradigms of Biodiversity Conservation: Human Intervention and Rewilding. BioScience, 69(12), 997–1007. https://doi.org/10.1093/biosci/biz106

Varela, E., Pulido, F., Moreno, G., Zavala, M.Á., 2020. Targeted policy proposals for managing spontaneous forest expansion in the Mediterranean. Journal of Applied Ecology, 57(12), 2373–2380. https://doi.org/10.1111/1365-2664.13779

Vázquez, D.P.,Simberloff, D. 2004. Indirect effects of an introduced ungulate on pollination and plant reproduction. Ecological Monographs, 74(2), 281–308. https://doi.org/10.1890/02-4055

Vedel, S.E., Jacobsen, J.B., Thorsen, B.J. 2015. Forest owners' willingness to accept contracts for ecosystem service provision is sensitive to additionality. Ecological Economics 113, 15–24. https://doi.org/10.1016/j.ecolecon.2015.02.014

Victorsson, J., Jonsell, M. 2013. Ecological traps and habitat loss, stump extraction and its effects on saproxylic beetles. Forest ecology and management, 290, 22–29. https://doi.org/10.1016/j.foreco.2012.06.057

Virkkala, R. 2016. Long-term decline of southern boreal forest birds: consequence of habitat alteration or climate change? Biodiversity and Conservation, 25(1), 151–167. https://doi.org/10.1007/s10531-015-1043-0

Vogel, S., Bussler, H., Finnberg, S., Müller, J., Stengel, E. and Thorn, S. 2021. Diversity and conservation of saproxylic beetles in 42 European tree species: an experimental approach using early successional stages of branches. Insect Conservation and Diversity, 14, 132–143. https://doi.org/10.1111/icad.12442

Vogt, P. 2019. Pan-European forest fragmentation. European Commission, Joint Research Centre (JRC) [Dataset].

Vranckx, G., Jacquemyn, H., Muys, B.,Honnay, O. 2012. Meta-analysis of susceptibility of woody plants to loss of genetic diversity through habitat fragmentation. Conservation biology, 26(2), 228–237. https://doi.org/10.1111/j.1523-1739.2011.01778.x

Vranckx, G., Mergeay, J., Cox, K., Muys, B., Jacquemyn, H.,Honnay, O. 2014. Tree density and population size affect pollen flow and mating patterns in small fragmented forest stands of pedunculate oak (*Quercus robur* L.). Forest Ecology and Management, 328, 254–261. https://doi.org/10.1016/j.foreco.2014.05.044

Wagendorp, T., Gulinck, H., Coppin, P., Muys, B. 2006. Land use impact evaluation in life cycle assessment based on ecosystem thermodynamics. Energy 31, 112–125. https://doi.org/10.1016/j.energy.2005.01.002

Wagner, V., Chytrý, M., Jiménez-Alfaro, B., Pergl, J., Hennekens, S., Biurrun, I., ... & Pyšek, P. 2017. Alien plant invasions in European woodlands. Diversity and Distributions, 23: 969–981. https://doi.org/10.1111/ddi.12592

Walls, S. C., Barichivich, W. J., Brown, M. E. 2013. Drought, deluge and declines: the impact of precipitation extremes on amphibians in a changing climate. Biology, 2(1), 399–418. https://doi.org/10.3390/biology2010399

Ward, M., Tulloch, A. I., Radford, J. Q., Williams, B. A., Reside, A. E., Macdonald, S. L., ... & Watson, J. E. 2020. Impact of 2019–2020 mega-fires on Australian fauna habitat. Nature Ecology and Evolution, 4(10), 1321–1326. https://doi.org/10.1038/s41559-020-1251-1

Watson, M. F., Hawksworth, D. L., Rose, F. 1988. Lichens on elms in the British Isles and the effect of Dutch Elm Disease on their status. The Lichenologist, 20(4), 327–352. https://doi.org/10.1017/S0024282988000441

Weiss, G., Lawrence, A., Hujala, T., Lidestav, G., Nichiforel, L., Nybakk, E., Quiroga, S., Sarvašová, Z., Suarez, C., Živojinović, I. 2019. Forest ownership changes in Europe: state of knowledge and conceptual foundations. Forest Policy and Economics, 99, 9–20. https://doi.org/10.1016/j.forpol.2018.03.003

Weiss, G., Wolfslehner, B., Zivojinovic, I. 2021. Who owns the forests and how are they managed? In: Mauser, H (ed). Key questions on forests in the EU. Knowledge to Action 4. European Forest Institute. https://doi.org/10.36333/k2a04

Wellbrock, N., Grüneberg, E., Riedel, T., Polley, H. 2017.Carbon stocks in tree biomass and soils of German forests. Cent. Eur. For. J. 63, 105–112. https://doi.org/10.1515/forj-2017-0013

Wende, W., Tucker, G. M., Quétier, F., Rayment, M., Darbi, M. (Eds.). 2018. Biodiversity offsets: European perspectives on no net loss of biodiversity and ecosystem services. Springer. https://link.springer.com/book/10.1007/978-3-319-72581-9

Westergren, M., Bozic, G., Ferreira, A., Kraigher, H.2015. Insignificant effect of management using irregular shelterwood system on the genetic diversity of European beech (*Fagus sylvatica* L.): A case study of managed stand and old growth forest in Slovenia. Forest Ecology and Management, 335, 51–59. https://doi.org/10.1016/j.foreco.2014.09.026

Wiensczyk, A. M., Gamiet, S., Durall, D. M., Jones, M. D., Simard, S. W. 2002. Ectomycorrhizae and forestry in British Columbia: A summary of current research and conservation strategies. Journal of Ecosystems and Management, 2(1). https://jem-online.org/index.php/jem/article/view/224/0

Wilson, E.O. (ed.) 1988. Biodiversity. National Academy Press, Washington, D.C. https://doi.org/10.17226/989

Winter, S., 2012. Forest naturalness assessment as a component of biodiversity monitoring and conservation management. Forestry, (85) 293–304. https://doi.org/10.1093/forestry/cps004

With, K. A., Crist, T. O. 1995. Critical thresholds in species' responses to landscape structure. Ecology, 76(8), 2446-2459. https://doi.org/10.2307/2265819

Wolfslehner, B., Pülzl, H., Kleinschmit, D., Aggestam, F., Winkel, G., Candel, J., Eckerberg, K., Feindt, P., Mcdermott, C., Secco, L., Sotirov, M., Lackner, M., Roux, J., 2020. European forest governance post-2020. From Science to Policy 10. European Forest Institute. https://doi.org/10.36333/fs10

Young, J. C., Marzano, M., White, R. M., McCracken, D. I., Redpath, S. M., Carss, D. N., ... & Watt, A. D. 2010. The emergence of biodiversity conflicts from biodiversity impacts: characteristics and management strategies. Biodiversity and Conservation, 19(14), 3973–3990. https://doi.org/10.1007/s10531-010-9941-7

We are living in a time of accelerated changes and unprecedented global challenges: energy security, natural resource scarcity, biodiversity loss, fossil-resource dependence and climate change. Yet the challenges also demand new solutions and offer new opportunities. The cross-cutting nature of forests and the forest-based sector provides a strong basis to address these interconnected societal challenges, while supporting the development of a European circular bioeconomy.

The European Forest Institute is an unbiased, science-based international organisation that provides the best forest science knowledge and information for better informed policy making. EFI provides support for decision-takers, policy makers and institutions, bringing together cross-boundary scientific knowledge and expertise to strengthen science-policy dialogue.