Addressing biodiversity in plantation forests management in northwestern Portugal

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"Opinion is really the lowest form of human knowledge. It requires no accountability, no understanding. The highest form of knowledge is empathy, for it requires us to suspend our egos and live in another's world."

Bill Bullard

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Resumo

A biodiversidade é importante para a manutenção da saúde e funcionamento sustentável dos ecossistemas florestais e, por conseguinte, a sua conservação é significativa em todos os tipos de florestas, incluindo plantações. A região do Mediterrâneo é conhecida como um 'hotspot' de biodiversidade, mas também como uma área com plantações florestais recentemente estabelecidas, que são criticadas pelo baixo nível de diversidade biológica e pelos frequentes incêndios florestais. Deste modo, incluir a avaliação da biodiversidade no planeamento da gestão de plantações florestais pode beneficiar a conservação florestal, a prevenção de incêndios e desenvolver alguns novos serviços de ecossistema.

Neste trabalho, procedeu-se ao desenvolvimento de investigação sobre: (1) que indicadores de biodiversidade podem ser considerados no planeamento da gestão florestal, (2) como avaliar e qual é o estado da biodiversidade ao nível do povoamento nas plantações florestais do noroeste de Portugal, (3) como usar indicadores ao nível do povoamento para avaliar aspectos da biodiversidade da paisagem em plantações florestais no noroeste de Portugal?

Os resultados demonstram que: (1) o mais conveniente para gestores florestais com diversas formações é considerar indicadores estruturais na integração da biodiversidade no planeamento da gestão florestal; (2) indicadores estruturais como indicador de espécies arbóreas, diâmetro médio (cm) e biomassa arbustiva (Mgha-1) são adequados para a avaliação da biodiversidade ao nível do povoamento. Os povoamentos de eucalipto em locais de baixa qualidade com regeneração de arbustos por via seminal apresentaram a menor biodiversidade média, enquanto que os povoamentos mistos com dominância de pinheiros, em locais de melhor qualidade com regeneração de arbustos por rebrota, apresentaram a maior biodiversidade média; (3) o uso do conceito de espécie guarda-chuva é adequado para as avaliações de biodiversidade ao nível da paisagem e, portanto, aqui é estimada a adequação do habitat de nidificação do milhafre-real (*Milvus milvus*); os resultados sugeriram que o habitat menos favorável seria em plantações puras de eucalipto, enquanto a floresta de sobreiro maduro serviria como o habitat mais apropriado a longo prazo.

Palavras-chave: Biodiversidade, indicadores de biodiversidade, planeamento da gestão florestal, plantações

Abstract

Biodiversity is important for forest ecosystem health maintenance and sustainable functioning and therefore its conservation is significant in all types of forests, including plantations. The Mediterranean region is known as a biodiversity 'hotspot', but also as an areawith recently established forest plantations, that are criticized for the low level of biological diversity and frequent forest fires. Therefore, including biodiversity assessment in forest plantations management planning might benefit forest conservation, fire prevention and land some novel ecosystem services.

Here we attempted to investigate: (1) which biodiversity indicators could be considered in forest management planning, (2) how to assess and what is the state of biodiversity at a stand level in plantation forestsof northwestern Portugal, (3) how to use stand-level indicators to assess landscape biodiversity aspects in plantation forests in northwestern Portugal?

Results demonstrated that: (1) the most convenient for forest managers with various backgrounds is to consider structural indicators in integrating biodiversity in forest management planning; (2) structural indicators such as tree species indicator, mean diameter (cm) and shrub biomass (Mg ha^{-1}) are suitable for stand level biodiversity assessment. Pure blue gum stands on low-quality sites with shrub regenerating by seed had the lowest mean biodiversity, while mixed stands with a dominance of pine, on best-quality sites with shrub regeneration by resprouting had the highest mean biodiversity; (3) using umbrella species concept is suitable for the landscape level biodiversity assessments, and thus, here is estimated the red kite (*Milvus milvus*) nesting habitat suitability; the results suggested that the least favourable habitat would be in pure blue gum plantations, while mature cork oak forest would serve as the most appropriate long-term habitat.

Keywords: Biodiversity, biodiversity indicators, forest management planning, plantations

Resumo alargado

Portugal faz parte da eco-região do Mediterrâneo que é conhecida por ser uma das áreas de biodiversidade mais ricas do mundo. Os ecossistemas florestais geralmente hospedam a maioria da biodiversidade terrestre e desempenham uma função importante na conservaçãoda mesma. No entanto, quase metade das florestas em Portugal são plantações. As plantações florestais são frequentemente criticadas por apresentarem um nível muito baixo de biodiversidade, principalmente as espécies exóticas. Embora alguns estudos tenham descoberto que as plantações exóticas geralmente hospedam muitas espécies locais de ervas, arbustos e pássaros.No entanto, recentemente são também alvo de críticas, os incêndios florestais que se tornaram mais frequentes devido às alterações climáticas e impõem grandes riscos e consequências ecológicas, económicas e sociais. Além disso, os efeitos das alterações climáticas refletem-se em secas mais intensas, inundações, epidemias e surtos de doenças. Aqui, a conservação da biodiversidade tem um papel crucial no aumento da resiliência dos ecossistemas florestais em relação às alterações globais. Deste modo, as políticas e iniciativas mais recentes exigem que a biodiversidade seja considerada em todos os tipos de floresta e não apenas nas naturais e protegidas, mas também nas geridas, incluindo plantações. A gestão florestal tendo a a conservação da biodiversidade em mente é importante para as florestas locais, nacionais, regionais e globais, pois a biodiversidade desempenha uma função importante no funcionamento sustentável do ecossistema e, assim a sua avaliação e monitorização devem ser incluídos nos planos de gestão dos recursos naturais.

Os gestores florestais têm atualmente uma função crucial quando se considera a biodiversidade no planeamento da gestão florestal sustentável. Eles precisam de orientação para abordar as preocupações com a biodiversidade e selecionar indicadores apropriados para cumprir a meta planeada de biodiversidade. É particularmente importante identificar as práticas florestais que afetam a biodiversidade e como a gestão florestal pode beneficiar a biodiversidade florestal. Uma vez que a maioria das operações de gestão em plantações são realizadas ao nível do povoamento, esta escala é crítica para a consideração e conservação da biodiversidade neste caso. Além disso, é importante fornecer uma ampla visão geral do estado da biodiversidade pois algumas espécies que são mais móveis do que outras e têm grande variedade de habitats. Neste âmbito, o nível da paisagem é o mais adequado.

Para avaliar a biodiversidade, os gestores florestais precisam de recorrer a indicadores apropriados. Os indicadores são usados como 'substitutos' da biodiversidade, já que a avaliação detalhada de toda a biodiversidade na floresta é quase impossível devido a restrições técnicas e financeiras. No entanto, há uma questão em aberto sobre quais indicadores usar em diferentes escalas? Essa questão costuma constituir um desafio até mesmo para os ecologistas e quando se trata de gestores florestais sem experiência em ecologia, pode ser ainda mais complicada. Assim, o objetivo desta dissertação é avaliar a biodiversidade no Vale de Sousa, no noroeste de Portugal, de uma forma que seja adequada para gestores florestais com formações diversas. O caso de estudo estende-se por uma área de 14. 840 ha, dos quais 97% estão cobertos por florestas dominadas por eucalipto puro (*Eucalyptus globulus* Labill). Em menor percentagem encontram-se povoamentos mistos de eucalipto e pinheiro bravo (*Pinus pinaster*

Aiton) e a menor percentagem de castanheiro (*Castanea sativa* Mill).

Portanto, aqui examinamos:

1. Quais indicadores de biodiversidade podem ser considerados no planeamento da gestão florestal?
2. Como avaliar e qual é o estado da biodiversidade ao nível do povoamento nas plantações florestais do noroeste de Portugal?
3. Como usar indicadores ao nível do povoamento para avaliar aspectos da biodiversidade da paisagem em plantações florestais no noroeste de Portugal?

Relativamente ao questão 1, verifica-se que os indicadores ao nível do povoamento são mais práticos do que os indicadores ao nível da paisagem. Além disso, indicadores estruturais de biodiversidade (por exemplo, árvores grandes e povoamentos florestais antigos) são mais úteis em planos de gestão florestal do que indicadores de composição, pois são fácilmente observáveis por não profissionais e podem ser obtidos nos inventários florestais. Indicadores compósitos como plantas vasculares, fungos, briófitas, líquenes e espécies de invertebrados são difíceis de identificar por não profissionais e, por isso, são impraticáveis. Indicadores funcionais (por exemplo, ciclagem de nutrientes) não são suficientemente abordados na literatura. O uso das bases de dados existentes atualizadas recentemente (por exemplo, inventários florestais nacionais e atlas de pássaros) é muito eficiente em termos de tempo e custo. A deteção remota e outras tecnologias (por exemplo, aplicações de smartphone) são considerados promissores para uma recolha eficiente de dados no futuro.

Até que os indicadores para todos os aspectos da biodiversidade florestal se tornem mais adequados para serem tratados por gestores florestais com várias formações, é aconselhável o focus nos indicadores estruturais, pois podem ser muito úteis para a avaliação da biodiversidade florestal.

Em relação à questão 2, seguindo os resultados da questão 1, foram utilizados três indicadores para avaliação da biodiversidade: indicador de espécies arbóreas, diâmetro médio (cm) e biomassa arbustiva (Mgha^{-1}). A espécie eucalipto em locais de baixa qualidade com regeneração de arbustos por via seminal teve a menor biodiversidade média, enquanto os povoamentos mistos com dominância de pinheiros, em locais de melhor qualidade com regeneração de arbustos por rebrota tiveram a maior biodiversidade média. A qualidade do local e a existência de espécies de rebrotadoras afetaram a biodiversidade em termos de ritmo de desenvolvimento e os valores médios no horizonte de 90 anos.

Os indicadores usados no estudo parecem apropriados para avaliação da biodiversidade florestal para plantações assim como para outros tipos de florestas. Porém, em relação à biomassa arbustiva, é aconselhável ter alguma cautela para prevenção de incêndios florestais e considerar valores mais elevados deste indicador em áreas de alta importância para conservação.

Em relação à questão 3, seguindo os resultados da questão 1, foram utilizados quatro indicadores para avaliação da biodiversidade: altura das árvores, densidade de árvores, diâmetro médio (cm), e frequência das atividades silviculturais. Estes são os indicadores de

adequação de habitat de espécie ’guarda chuva’ Milhafre-real, cujo potencial de conservação beneficiaria a conservação da biodiversidade nas plantações do Noroeste de Portugal. Foi avaliada a adequabilidade do habitat da Milhafre-real no estudo de caso e, adicionalmente, comparada a adequação do habitat em sistemas florestais alternativos que são sugeridos para substituir os atuais devido aos recentes incêndios florestais. Os resultados mostram que o habitat menos favorável seria nas plantações de carvalho comum puro, enquanto o sobreiro maduro serviria como o habitat mais adequado a longo prazo. Os sistemas florestais mistos de pinheiro bravo e eucalipto são melhores devido a um período de rotação mais longo do pinheiro bravo, mas ainda são desfavoráveis devido a práticas silviculturais ainda mais frequentes. Os modelos atuais mais favoráveis são os que consideram os castanheiros, porém apenas na última década antes do corte raso. Em relação aos modelos alternativos, os povoamentos de pinheiro puro apresentam duas vezes mais densidade de árvores do que os modelos atuais, porém a rotação é menor. Os carvalhos comuns parecem desfavoráveis devido ao crescimento lento e altura que nunca atinge o valor ideal para os ninhos. O mais favorável de todos os modelos atuais e alternativos é o que considera os sobreiros devido à baixa densidade das árvores, altura suficiente para nidificação e ausência de corte raso para interromper os objetivos de conservação a longo prazo. Os indicadores que aplicamos podem ser úteis para a conservação de outras aves de rapina com requisitos de habitat semelhantes.

Palavras-chave: Biodiversidade, indicadores de biodiversidade, planeamento de gestão florestal, plantações

Preamble

The thesis contains 5 chapters: i) the introduction, ii-iv) a set of publishable articles, and v) final considerations. The introduction focuses on the motivation for the research, namely on the problems such as the absence of biodiversity addressing in forest management in plantation forests, literature gaps, challenges, a need for such scientific research and how I am planning to fill the knowledge gaps. Chapters ii-iv are answers to the research questions of the thesis. A set of publishable articles is organized as follows:

1) **The review article is published in the journal Forests:** Ćosović, M., Bugalho, M. N., Thom, D., Borges, J. G. (2020). StandStructural Characteristics Are the Most Practical Biodiversity Indicators for Forest Management Planning in Europe. MDPI Forests, 11(3), 343. https://doi.org/10.3390/f11030343
   It is the editor's choice and has 28 citations on Google Scholar, 21 on Scopus

2) **The research article is published in the journal Forest Systems:** Ćosović, M (2022) Using inventoryvariables for practical biodiversity assessment in plantation stands. Forest Systems, 31(2), e016. https://doi.org/10.5424/fs/2022312-18856

3) **The research article submitted to the journal Ardeola:** Ćosović, M., Bugalho, M. N., Vangansbeke, P., Borges, J. G.(2023). Forest plantations as a potential habitat for the red kite (*Milvus milvus*) raptor breeding

The final considerations chapter encompasses the result of the research of the thesis and discusses the impact of such results on existing literature and forest management practices, presenting limitations and suggestions for future research to follow up.

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Chapter I – General Introduction

1. Plantation forests in Portugal: current situation and challenges

Forest covers about 35% of Portugal's land, according to data from the 6th National Forest Inventory (ICNF 2013; Nunes et al. 2019). That is 3,155 000 ha, and out of the total forest area, 48.40% are plantation forests that spread over 1,526 000 ha (ICNF 2013; FAO 2015; Pra et al. 2019). According to FAO (2020), a plantation forest is defined as "an intensively managed planted forest that at maturity is composed of one or two species, has one age class, and has regular tree spacing". The most dominant tree species in Portugal's forests is blue gum (*Eucalyptus globulus*) covering 800,000 hectares (Salvado 2015; Nunes et al. 2019) which is 26% of all forest cover (Pra et al. 2019). In the 1950s the area covered with blue gum was about 50 000 ha (ICNF 2013; Branco et al. 2014), which signifies the proportions of the introduction of this exotic species in Portugal. The main products of blue gum plantations are paper and cardboard, where the production corresponds to 5% of all national exports (Sarmento and Dores 2013; Martins et al. 2011; Nunes et al. 2019). Therefore, the pulp and paper sector is very developed in Portugal, contributing to approx. 4000 jobs (Nunes et al. 2019). However, in the past decades, since the introduction of the eucalyptus longhorn beetle (*Phoracantha semipunctata* Fab.) and other diseases, the vitality of blue gum treesis seriously affected and threatens to cause significant economic loses in the future. Additionally,there are numerous negative ecological impacts of blue gum plantations such as compromising native biodiversity and water resources (Pra et al. 2019), and therefore forest managers are encouraged to think of converting plantations into woodlands with native tree species.

The second dominant tree species in Portugal is maritime pine (*Pinus pinaster*) and it covers over 714,000 ha, which is 23% of all forests (ICNF 2013; Branco et al. 2014). Maritime pine is native species, however, similarly to blue gum, the plantations expanded in the past century, particularly between 1928 and 1974, when one of the largest ever in Europe continuous pine forests was created (Nunes et al. 2019). The most important products from maritime pine plantations are timber, pallets, boards, and resin which make a revenue of around 306 million Euros in exports (Mendes et al. 2004; Branco et al. 2014). However, maritime pine is seriously affected by outbreaks of the pine wilt nematode (*Bursaphelenchus xylophilus*) since the 2000s, which causedgreat economic losses in the Portuguese sawmill industry, and a tendency among forest managersto convert pine plantations into eucalypt ones (Rodrigues 2008; Pra et al. 2019).

In the past, the most dominant species in northern Portugal was the common oak (*Quercus robur*) andin the southern area, it was the cork oak (*Quercus suber*) (Ramil-Rego et al. 1998; DGRF 2007; Proenca et al. 2010). While in the southern area, cork oak still covers a large area, northern Portugal suffered a significant transformation regarding forest cover to this day. The southern cork oak ecosystems are relatively ecologically stable and less prone to forest fires, while northernforests are affected by severe forest fires almost every year in the past decade. Also, pests and diseases are additionally causing economic losses, which are predicted to increase due to the mainly monocultural character of plantations.

Therefore, it is necessary to consider alternative forest management where forest ecosystem resilience providing is included in forest management plans. To create stable and resistant ecosystems that will provide sustainable ecosystem services, it is crucial to focus on biodiversity conservation. Only in such a way, the long-term social and economic benefits for forest stakeholders are possible. Also, it is important to understand that forests and the numerous plants andanimals that live there have intrinsic values that go beyond human benefit from it.

2. Forests and biodiversity

2.1 What is biodiversity?

Biodiversity is '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 a part; this includes diversity within species, between species and of ecosystems' (Convention on Biological Diversity, United Nations 1992). Forest biodiversity is defined as the variety of all forms of life and their organisation within the forest area (Hunter 1990; Winter et al. 2011).

2.2 Forest biodiversity policies and initiatives

Regarding policies and initiatives that address biodiversity conservation in European forests, there are (1) global-level policies and (2) EU-level policies. Global-level initiatives that advocate the management of forests in a sustainable manner and forest protecting were initiated with the United Nations Conference on Environment and Development (UNCED) in Rio 1992, where 'Rio forest principles' were adopted (European Commission 2010). Here, the concept of Sustainable Forest Management (SFM) has been first developed and it is defined as an integrated approach to the management and 'use of forests and forest land in a way and at a rate that maintains their biodiversity, productivity, regeneration capacity, vitality and potential to fulfil, now and in the future, relevant ecological, economic and social functions at local, national and global levels and does not cause damage to other ecosystems' (PEFS 2010).

Right after, the Convention on Biological Diversity (CBD) entered into force on 29 December 1993, intending to conserve biological diversity and call for the sustainable use of the components of biodiversity and fair and equitable sharing of the benefits originating from the use of genetic resources (CBD 2017).

In regards to the pan-European level, the Ministerial Conference on the Protection of Forests in Europe (MCPFE) 'led by European national forestry authorities', had a great contribution to defining SFM through resolutions, criteria and indicators (Winkel and Sotirov 2016). Since MCPFE in Helsinki 1993, subsequent conferences (Lisbon MCPFE 1998 and Vienna MCPFE 2003) continued to actualize recommendations for SFM and forest protection criteria, along with the indicators for national reporting (see Table 1) (European Commission 2010). Further, at the MCPFE conferences in Warsaw 2007, Oslo 2011 and finally, Madrid 2015, the achievements in regards to progress in SFM were discussed as well. However, until 2011 MCPFE remained a non-legally binding process and its implementation is questionable due to failed attempts to assess 'trade-offs between objectives such as enhancing biodiversity, timber production, and the importance of the European forests for recreation' (Winkel and Sotirov 2016). Nevertheless, the 'negotiations for a Legally Binding Agreement on Forests in Europe' were started in Oslo MCPFE 2011, thus making a historic step in the matter of European forests policy (Forest Europe 2017). To this day, the negotiations are still in process. In 2010, MCPFE became officially named Forest Europe. All EU Member States and the Commission have signed the resolutions, thus 'confirming SFM and multifunctionality as a core approach to forestry'.

The basis of European legislation related to biodiversity conservation consists of two policies: the Birds Directive and the Habitats Directive. The first was adopted in 1979 to protect wild birds and their habitats within the EU, while the latter was adopted in 1992 to protect wildlife habitats of rare, threatened or endemic species, in the same area (European Commission 2009). To provide more viable conservation of species and habitats listed under these two directives, the network of protected areas called Natura 2000, has been established and takes over 18 % of the EU territory (European Commission 2017a).

In 2005, the EU Commission presents a Communication on the implementation of the EU Forestry Strategy (FS)(European Commission 2016). The strategy 'sets out common principles of EU forestry – SFM and multi-functionality - and lists international processes and activities to be followed at EU level' (European Commission 2010). This policy is followed by The EU Forest Action Plan (FAP) in 2006, and is made based on FS and aims in coordinating policies and actions related to biodiversity maintaining and enhancing, carbon sequestration, integrity, health and resilience of forest ecosystems at multiple geographical scales (European Commission 2010). Further, the Green Paper on forest protection and information has been adopted in 2010 (European Commission 2016). The objective of the Green Paper is to 'launch the debate on options for a European Union (EU) approach to forest protection and information in the framework of the EU Forest Action Plan' (European Commission 2010).

Regulation 1293/2013 'establishes the Environment and Climate Action sub-programmes of the LIFE Programme for the funding period, 2014–2020' and also aims in supporting 'sustainable development and the achievement of the objectives and targets of the Europe 2020 Strategy' (European Commission 2017b). The LIFE Programme is the EU's funding instrument for the environment and climate action and was first initiated in 1992. On October 22, 2014, the Regulation on Invasive Alien Species (Regulation (EU) No 1143/2014) was adopted by European Parliament to manage the introduction and spreading of exotic invasive species in Europe (European Commission 2020a).

Finally, the EU biodiversity strategy is adopted as a response to the international Convention on Biological Diversity and its commitments taken by the EU in 2010. Initially, it was adopted as a Resolution on the mid-term review of the EU Biodiversity Strategy for 2020 (European Commission 2017c). While the most recent event was the adoption of the EU Biodiversity Strategy for 2030 in June 2020, where some of the aims are'(1) to turn at least 30% of the EU land into effectively managed and coherent protected areas, (2) to restore degraded ecosystems and stop any further damage to nature, (3) plant over 3 billion diverse, biodiversity-rich trees' (European Commission 2020b). Another initiative that addresses biodiversity is the European Green Deal, which is a set of proposals to create "EU's climate, energy, transport and taxation policies fit for reducing net greenhouse gas emissions by at least 55% by 2030, compared to 1990 levels" (European Commission 2023). Here, preserving the Environment, and therefore biodiversity, is one of the priorities.

Table 1: Policies and processes on forest biodiversity at the EU level

Year	EU level Policy	Legal/regulatory instrument	Non-legally binding instrument
1979	Birds Directive	+	
1992	Habitat Directive	+	
1993	MCPFE Helsinki		+
1998	MCPFE Lisbon		+
2000	MCPFE Vienna		+
2005	Forestry Strategy (FS)		+
2006	EU Forest Action Plan (FAP)		+
2007	MCPFE Warsaw		+
2010	Green Paper		+
2011	MCPFE Oslo		+
2012	EU resolution biodiversity strategy 2020	+	
2013	EU Regulation (LIFE)	+	
2014	Regulation on Invasive Alien Species	+	
2015	MCPFE Madrid		+
2020	EU Biodiversity Strategy for 2030	+	

Regarding forest biodiversity policies in Portugal, the most relevant law that recently entered into force in 2018, prevents any further spreading of eucalypt plantations to new areas (Law No. 77/2017) (Pra et al. 2019). The law followed one of the most devastating forest fires that took place in 2017 and left unprecedented consequences such as the loss of numerous animal and human lives anddestroyed hundreds of hectares of forests and shrublands.

According to the above, the evolution of policies and motivation for policy tailoring were initially related to halting biodiversity losses, which were occurring at an alarming rate globally, due to human actions. The focus was on threatened species and habitats. The implementation of policies took place at a very low pace and biodiversity is still having a declining trend globally. Particularly problematic are tropical countries where thousands of hectares of forests are being lost on yearly basis due to agriculture and illegal logging. In Europe, forest area is increasing, however, land fragmentation and simplification are still occurring. Such actions are causing changes in disturbance patterns (Noss and Csuti 1997; Paine et al. 1998; Elmqvist et al. 2003). Hence, the latest threat to forest biodiversity is climate change which is projected to have a detrimental effect and cause ecological, economic and social consequences. Due to climate change, the disturbance occurrence and intensity are projected to increase in both natural and managed forests (Woods et al.

2005; Pawson et al. 2013; Verheyen et al. 2016). The effects of climate change are already obvious where forest fires, droughts, floods, pests and disease outbreaks are intensified.

Natural resources and ecosystem services sustainability strongly depend on ecosystem resilience, particularly in the face of a changing climate that brings uncertainty (Gunderson and Holling 2002; Elmqvist et al. 2003). Ecosystem resilience is the measure of disturbance that an ecosystem can sustain and still keep its original state or appearance (Holling 1973, 1996; Elmqvist et al. 2003); it is also the capacity of a biological system affected by disturbance and change, to rearrange and re-establish itself (Elmqvist et al. 2003). Biodiversity plays a critical role in ecosystem resilience and stability. For instance, certain pests and diseases usually attack certain plant species and therefore, the ecosystems with rich biodiversity will be less affected by biological outbreaks than the ecosystems with low biodiversity. About 15-20% of European forests are yearly affected by pests and diseases that lead to tree dieback or decreased biomass production (Verheyen et al. 2016).

Correspondingly, forests rich in genetic, structural, functional and compositional biodiversityare more likely to adjust to changing environment, than forests with poor biodiversity such are monocultures (van Hensbergen 2006; Bauhus et al. 2010; Verheyen et al. 2016). Here, biodiversity conservation has a crucial role in increasing forest ecosystems' resilience towards global change facing. Therefore, future policies and initiatives should be inviting biodiversity conservation in all types of forests and not just natural and protected but managed as well.

Finally, the recent and still ongoing pandemic outbreak of the virus COVID-19, which has left severeconsequences on humankind, is linked to forests and forest biodiversity destruction by some worldwide organisations such as United Nations (UN). The latest Biodiversity strategy for 2030 (European Commission 2020b), underlines the importance of biodiversity for the resilience of future disease outbreaks, and plans ‘putting biodiversity on the path of recovery by 2030 for the benefit of people, climate and the planet’.

2.3 Forest management and biodiversity

Forest management is defined as “the practical application of biological, physical, quantitative, managerial, economic, social, and policy principles to the regeneration, management, utilization, and conservation of forests to meet specified goals and objectives while maintaining the productivity of the forest” CSU (2019). Around 30% of the world’s forests (ca. 1.2 billion ha) and 57% of European forests excluding Russia are managed for wood biomass production (FAO 2010; Purahong 2014). Forests are biological systems that host most of the world’s terrestrial biodiversity (Ozanne et al. 2003; Gao et al. 2014) and provide numerous ecosystem services, thus the role of forests in a changing environment is of unprecedented importance (Shvidenko et al. 2005; Proença et al. 2010).

Since addressing biodiversity in management became a necessity due to resilience provision, halting further biodiversity losses and demanding accordingly tailored policies, forest managers need to be knowledgeable about biodiversity conservation. A vast area of European forests, with different ownership patterns, management categories and ecologic conditions, needs to be managed in a manner that inevitably includes addressing biodiversity. Otherways, with ‘business as usual’ and environmental change, biodiversity losses will likely aggravate (Warren et al. 2013; Newbold et al. 2015). Besides all of those reasons, forest managers ought to be mindful that the forests they manage locally, have importance for the global biodiversity context (Noss 1999). Also, there are additional motivations for biodiversity enhancement in managed forests such as preventing losses of aesthetical and recreational values, supporting awareness of intrinsic values and rights of all species, and promoting economic and social values of species (Failing and Gregory 2003).

Therefore, the management of natural resources, and thus forests, need to focus on resilience reinforcement as a means of dealing with uncertainty and surprise. Here, biodiversity has a critical role and for this reason, forest management and policy need to be shaped accordingly (Folke et al. 2002; Elmqvist et al. 2003).

Forest managers are now having a crucial role in considering biodiversity in sustainable forest management planning (European Commission 2020b). They need guidance to address biodiversity concerns and selectappropriate indicators to fulfil the planned biodiversity goal. It is particularly important to identifyhow forest practices affect biodiversity and how forest management might benefit forest biodiversity (Noss 1999). However, there is a question on how effectively forest managers can undertake this quest, relatively new to them.

2.4 Integration of biodiversity assessment into forest management planning

In recent decades, forest management priorities have certainly changed and new standards have arisen, such as including aspects with non-monetary value (e.g., biodiversity) in forest management plans (Ezquerro et al. 2016). Mathematical models are used in forest management planning to evade extensive data collection and examinations (Pretzsch et al. 2008; Pretzsch et al. 2014). Models attempt to provide a numerical portrayal of physical processes and also, simulate different scenarios that can be calculated and visualized (Muys et al. 2010). However, forest models need to be tested and compared with field data for acquiring maximum capacity, and cannot solely replace basic field measurements (Pretzsch et al. 2014). The first forest model has been constructed by Réaumur in 1721 in the form of yield tables to help forest management planning and forecast the production of forest stands (Pretzsch 1992; Lexer et al. 2000). The yield tables are mainly designed for even-aged mono-specific stands and often assume a specific management scheme (van Gadow and Hui 1999) are thus inappropriate for assessing the impact of various forestry treatments on biodiversity (Lexer et al. 2000). There are many models developed since the first yield tables, however, there are two types that dominate in the research: empirical and process-based (Kimmins et al. 1990; Bossel 1991; Mohren and Burkhart 1994; Lexer et al. 2000). Empirical models represent trends observed in forest-based measurement plots (Twery and Weiskittel 2013), while process-based models integrate a mechanistic explanation of the interaction of the modelled aspects with the environment (Lexer et al. 2000). There are also hybrid models, and these are a combination of both empirical and mechanistic methods (Twery and Weiskittel 2013). Generally, all of these types of models are applicable on various scales such are: tree level, size class level, stand level, ecosystem, landscape, and the global level (Lexer et al. 2000). Single-tree models are explicitly designed to address the response of forest growth to a range of forestry treatments and therefore are appropriate for identifying biodiversity variables (Lexer et al. 2000). Size class models provide significantly more information about the composition and structure of the stand and allow for the assessment of silvicultural impacts on the development of the stand (Solomon et al. 1986; Lexer et al. 2000). Distance dependent tree models provide a thorough explanation of tree distribution and stand structure, (e.g. Pretzsch 1992; Hasenauer 1994) and thus allow the measurement of an array of proposed structural indicators for tree distribution and stand structure (Lexer et al. 2000). Lexer et al. (2000) argued that forest models that do not integrate tree regeneration are not suitable for biodiversity assessment. Also, the tree mortality variable, in intensively managed forests, is not a valuable indicator for biodiversityassessment because it is induced by human interventions and not naturally, while in unmanaged forests it is induced by competition and thus well represents the natural processes (Shugart 1984;Monserud 1976; Lexer et al. 2000).

Simulators or simulation tools are used to encapsulate models and to project outcomes of management options (Muys et al. 2010). In the last decade, simulation tools have gradually been paired with optimization and choice algorithms and approaches (Muys et al. 2010). Optimization serves for identifying appropriate treatment pathways for individual stands or solving complex planning problems with multiple objectives e.g., net present value, biodiversity indicator and landscape metric (Muys et al. 2010). The most commonly used optimization technique in forest management is Linear Programming (LP), particularly for strategic forest planning issues such as timber harvesting scheduling (Ezquerro et al. 2016). Other methods such are Metaheuristics (MH) (e.g., Falcão and Borges 2002, Bettinger et al. 2002; Pukkala and Kurtilla 2005) or mixed integer programming (e.g. Constantino et al. 2008, Tóth and McDill 2008) have been proposed to address concerns with a wider range of ecosystem services in forest management planning. When the decision-maker faces problems in which multiple parameters indicate a degree of disagreement between each other, decision analysis may be supported further by Multi-Criteria Decision-Making methods (MCDMs) (Ezquerro et al. 2016; Borges et al. 2017). MCDMs are used in strategic planning decisions, such as opting between optimum sustained yield and close-to-nature management and have been effectively integrated into Decision Support Systems (DSS) (Reynolds et al. 2008; Muys et al. 2010; Nordstrom et al. 2019). Ezquerro et al. (2016) conducted areview of operational research approaches used for the integration of biodiversity goals into forest management planning, and their results demonstrated that there is no universality in the choice of optimisation techniques used among cases as they identified significant variation in the optimization strategies used to incorporate biodiversity-related aspects into forest management. LP model building pioneered the use of optimization techniques to address concerns with wildlife management and biodiversity conservation (Thompson et al. 1973). Currently, the most predominantly used are MCDMs and MH techniques as well as mixed-integer programs (Könnyű et al. 2014; Ezquerro et al. 2016; St John et al. 2016).

It is important for forest managers first to know the temporal and spatial scale of the model and then to decide on the most appropriate operational research technique (Ezquerro et al. 2016). When selecting a model-building technique, managers should consider how well it may address multiple objectives and ecosystem services as well as if there is a need to incorporate the views of the various stakeholders (Ezquerro et al. 2016). Moreover, they should check whether there is software available to solve the models, namely in the framework of DSS (Reynolds et al. 2008; Ezquerro et al. 2016). Regarding the involvement of different stakeholders in decision-making, there is a study from North-Western Portugal where web-based DSS named wSADfLOR was used to promote stakeholder access to resources that can lead to improving planning for forestmanagement (Marto et al. 2019). That is a novel DSS that integrates the functionality of databases, simulators of vegetation dynamics and biodiversity indicators, optimization and multiple criteria techniques, and a web-based graphical interface. Also, wSADfLOR considers growth and yield simulation for holm oak, maritime pine, blue gum and chestnut. The results demonstrated that wSADfLOR is suitable for tackling diverse contexts of multi-objective and multi-decision-makermanagement planning including addressing biodiversity in forest management.

3. The thesis aim and research questions

There are numerous studies on considering biodiversity in forest management in Europe, however, it remains unclear how biodiversity assessment is feasible and which indicators to choose, particularly by forest managers with no background in forest ecology. Some studies attempted to address practical forest biodiversity assessment that is simple and cost-effective (e.g., Smith et al. 2008; Coote et al. 2013; Angelstam and Dönz-Breuss 2004). However, there is a lack of studies that examine the general practicality of forest biodiversity indicators used in forest management in Europe, in various types of forests and scales. Overall, there is a lack of studies that address biodiversity assessment in forests from the perspective of forest managers. There is a need for research that provides a starting point in biodiversity assessment that is clear, unambiguous, simple and cost-effective. Biodiversity is a very complex aspect and thus should be approached step-by-step. However, if the initial step in the assessment is not appropriately undertaken, the entire knowledge about biodiversity and the need for conservation would be under question. Therefore, it is necessary to provide an unambiguous knowledge base on practical ways to assessthe overall state of biodiversity in forests that would provide forest managers with the possibility to operate sustainably.

In this thesis, I developed a literature review of forest biodiversity indicators in Europe with a reference to their practicality. Further, I developed approaches to integrate biodiversity in forest management using these indicators and assessed the biodiversity value of stand-level forest management models in northwest Portugal. The main aim of the thesis was to address biodiversity in forest management in forest plantations of northwest Portugal in a practical manner at a stand and a landscape scale.

Therefore, the open questions are:

1. Which biodiversity indicators to consider in forest management in Europe?
2. How to assess and what is the state of biodiversity at a stand-level in plantation forests in northwestern Portugal?
3. How to use stand-level indicators to assess landscape biodiversity aspects in plantation forests in northwestern Portugal?

References

Angelstam, P.; Dönz-Breuss, M. (2004). Measuring Forest Biodiversity at the Stand Scale: An Evaluation of Indicators in European Forest History Gradients. Ecol. Bull. 51, 305–332.

Bossel, H. (1991). Modelling forest dynamics. Moving from description to explanation. For. Ecol. Manage. 42:129-142.

Bauhus, J., P. van der Meer, and M. Kanninen. (2010). Ecosystem goods and services fromplantation forests. London: Earthscan

Branco M., Bragança H., Sousa E., Phillips A.J. (2014) Pests and Diseases in Portuguese Forestry: Current and New Threats. In: Reboredo F. (eds) Forest Context and Policies in Portugal. World Forests, vol 19. Springer, Cham. https://doi.org/10.1007/978-3-319-08455-8_5

Coote, L., Dietzsch, A. C.,Wilson, M.W. et al. (2013). Testing indicators of biodiversity for plantation forests. Ecological Indicators 32 (2013) 107– 115. http://dx.doi.org/10.1016/j.ecolind.2013.03.020

CSU (2019). Colorado State Forest Service. Programmatic Areas. Colorado State University. Forest Management. Referred on 28.5. 2019. Available at: https://csfs.colostate.edu/programmatic-areas/

CBD (2017). Convention on Biological Diversity. Introduction. Referred on 20.5. 2017. Available at: https://www.cbd.int/intro/default.shtml

DGRF (2007). Resultados Do IFN 2005/06. Direcção Geral dos Recursos Florestais. http://www.dgrf.min-agricultura.pt.

Elmqvist, T., Folke, C., Nyström, M., Peterson, G., Bengtsson, J., Walker, B. and Norberg, J., (2003). Response diversity, ecosystem change, and resilience. Frontiers in Ecology and the Environment, 1(9), pp.488-494.

Ezquerro, M., Pardos, M., & Diaz-Balteiro, L. (2016). Operational research techniques used for addressing biodiversity objectives into forest management: an overview. Forests, 7(10), 229.

European Commission (2023). A European Green Deal. Reffered on 05.02.2023. Available at: https://commission.europa.eu/strategy-and-policy/priorities-2019-2024/european-green-deal_en

European Commission (2020a). Invasive alien species. Reffered on 20.11.2020. Available at: https://environment.ec.europa.eu/topics/nature-and-biodiversity/invasive-alien-species_en#law

European Commission (2020b). EU Biodiversity Strategy for 2030. Bringing nature back into our lives..Communication from the commission to the European parliament, the council, the European economic and social committee and the committee of the regions. European Commission, Brussels, 20.5.2020; COM(2020) 380 final

European Commission (2010). GREEN PAPER On Forest Protection and Information in the EU:Preparing forests for climate change. Brussels, 1.3.2010, COM (2010)66 final

European Commission (2009). Natura 2000, Europe's nature for you. European Communities. Luxembourg: Offi ce for Offi cial Publications of the European Communities, 2009

European Commission (2017a). Natura 2000. Referred on 20.12.2018. Available at: http://ec.europa.eu/environment/nature/natura2000/index_en.htm

European Commission (2017b). Referred on 20.12.2018. Available at: http://ec.europa.eu/environment/life/about/index.htm#life2014

European Commission (2017c). Referred on 20.12.2018. Available at: http://ec.europa.eu/environment/nature/biodiversity/strategy/index_en.htm

European Commission (2016). Referred on 6.01. 2019. Available at: http://ec.europa.eu/environment/forests/index_en.htm

FAO (2020). Terms and Definitions. FRA 2020.
Referred on 21.11.2022. Available at https://www.fao.org/3/I8661EN/i8661en.pdf

FAO (2015). Global Forest Resources Assessment 2015. Food and Agriculture Organization of the United Nations, Rome.

FAO (2010). Global forest resources assessment 2010 main report. Food and Agriculture Organization of the United Nations, Rome.

Failing L. & Gregory R. (2003). Ten common mistakes in designing biodiversity indicators forforest policy. Journal of Environmental Management 68 (2003) 121–132.

Folke C, Carpenter SR, Elmqvist T, et al. (2002). Resilience and sustainable development: building adaptive capacity in a world of transformations. Ambio 31: 437–40.

Forest Europe (2017). What is Forest Europe? Referred on 6.10. 2017. Available at: http://foresteurope.org/foresteurope/

Gao, T., Hedblom, M., Emilsson, T. & Busse Nielsen A. (2014). The role of forest stand structure as biodiversity indicator . Forest Ecology and Management 330 (2014) 82–93; http://dx.doi.org/10.1016/j.foreco.2014.07.007

Gunderson LH and Holling CS (Eds). (2002). Panarchy: understanding transformations in systems of humans and nature. Washington, DC: Island Press.

ICNF (2013). IFN6—Áreas dos usos do solo e das espécies florestais de Portugal continental. Resultados preliminares. Instituto da Conservação da Natureza e das Florestas. Lisboa, 34 pp

Hasenauer H., (1994). Ein Einzelbaumwachstumssimulator für ungleichaltrige Fichten-Kiefern- und Buchen-Fichtenmischbestände. Forstl. Schriftenreihe, Univ. f. Bodenkultur, Wien. Österr. Ges. f. Waldökosystemforschung und experimentelle Baumforschung an der Univ. f. Bodenkultur. Band 8, 152 p.

Holling CS. (1973). Resilience and stability of ecological systems. Annu Rev Ecol Syst 4: 1–23

Holling CS. (1996). Engineering resilience versus ecological resilience. In: Schulze PC (Ed). Engineering within ecological constraints. Washington DC: National Academy Press.

Kimmins, J. P., Comeau, P. G., Kurz, W. (1990). Modelling the interaction between moisture and nutrients in the control of forest growth. For. Ecol. Manage. 30:361-379.

Muys, B., Hynynen, J., Palahi, M., Lexer, M.J., Fabrika, M., Pretzsch, H., Gillet, F., Briceño, E., Nabuurs, G.J. and Kint, V. (2010). Simulation tools for decision support to adaptive forest management in Europe. Forest Systems, 3(4), pp.86-99.

Martins, J.B.; Gomes, L.M.F.; Beato, C.S.M. (2011). A revitalização do espaço rural e propostas de intervenção no sentido do turismo local: O caso de estudo de Santo André (Interior-Norte de Portugal). In Proceedings of the ICEUBI2011 International Conference on Engineering UBI2011 Innovation & Development, Covilhã, Portugal, 28–30 November 2011.

MFR (2008). Ministry of Forests and Range Glossary of Forestry Terms in British Columbia, March 2008 (https://www.for.gov.bc.ca/hfd/library/documents/glossary/Glossary.pdf).

Marto, M., Reynolds, K.M., Borges, J.G., Bushenkov, V.A., Marques, S., Marques, M., Barreiro, S., Botequim, B. and Tomé, M. (2019). Web-based forest resources management decision support system. Forests, 10(12), p.1079.

Mohren, G. M. J., Burkhart, H. E. (1994). Contrasts between biologically based process models and management-oriented growth and yield models. For. Ecol. Manage. 69:1-5.

Monserud, R. A. (1976). Simulation of forest tree mortality. For. Sci. 22:438-444

Mendes, A.M.S.C., Feliciano D., Tavares, M., Dias, R. (2004). The Portuguese forests country level report delivered to the EFFE Project, Evaluating financing of forestry in Europe. Portuguese Catholic University, Porto Regional Center, Faculty of Economics and Management, Porto

Newbold, T., Hudson, L. N., Hill, S. L., Contu, S., Lysenko, I., Senior, R. A., ... & Day, J. (2015). Global effects of land use on local terrestrial biodiversity. Nature, 520(7545), 45-50.

Noss R. F. (1999) Assessing and monitoring forest biodiversity: A suggested framework and indicators. Elsevier, Forest Ecology and Management 115 (1999) 135±146

Noss R.F. and Csuti, B. (1997). Habitat fragmentation. In: Meffe, G.K., Carroll, R.C. (Eds.), Principles of Conservation Biology, 2nd edn., Sinauer, Sunderland, MA, pp. 269±304.

Nunes, L. J. R., Meireles, C. I. R., Gomes, C. J.P., & Ribeiro, N. M. C. A. (2019). Historical development of the Portuguese forest: The introduction of invasive species. *Forests*, *10*(11). https://doi.org/10.3390/f10110974

Ozanne, C.H.P., Anhuf, D., Boulter, S.L., Keller, M., Kitching, R.L., Körner, C., Meinzer, F.C., Mitchell, A.W., Nakashizuka, T., Silva Dias, P.L., Stork, N.E., Wright, S.J., Yoshimura, M., (2003). Biodiversity meets the atmosphere: a global view of forestcanopies. Science 301, 183–186.

Pawson, S.M., A. Brin, and E.G. Brockeroff. (2013). Plantation forests, climate change and biodiversity. Biodiversity and Conservation 22: 1203–1227

PEFC International (2010). Sustainable Forest Management- – Requirements. PEFC ST 1003:2010. PEFC Council World Trade Center 1, 10 Route de l'Aéroport CH-1215Geneva, Switzerland

Paine RT, Tegner MJ, and Johnson EA. (1998). Compounded perturbations yield ecological surprises. Ecosystems 1: 535–45

Pra, A., Masiero, M., Barreiro, S., Tomé, M., Martinez De Arano, I., Orradre, G., … Pettenella, D. (2019). Forest plantations in Southwestern Europe: A comparative trend analysis on investment returns, markets and policies. *Forest Policy and Economics*, *109*(October), 102000. https://doi.org/10.1016/j.forpol.2019.102000

Pretzsch, H., (1992). Konzeption und Konstruktion von Wuchsmodellen für Rein- und Mischbestände. Institut für Ertragskunde der Forstl. Forschungsanstalt München, Band 115, 332 p.

Pretzsch, H., Grote, R., Reineking, B., Ro¨tzer, T. & Seifert, S. (2008). Models for forest ecosystem management: A European perspective. Ann. Bot. 101, 1065–1087

Pretzsch, H., Biber, P., Schütze, G., Uhl, E. and Rötzer, T., (2014). Forest stand growth dynamics in Central Europe have accelerated since 1870. Nature communications, 5(1), pp.1-10.

Proença, V.M. Pereira, H.M., Guilherme J.,Vicente J. (2010). Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal Acta Oecologica 36219e226

Purahong, W, Kapturska D, Pecyna MJ, Schulz E, Schloter M, et al. (2014). Influence of Different Forest System Management Practices on Leaf Litter Decomposition Rates, Nutrient Dynamics and the Activity of Ligninolytic Enzymes: A Case Study from Central European Forests. PLoS ONE 9(4): e93700. doi:10.1371/journal.pone.0093700 Forests. PLoS ONE 9(4): e93700. doi:10.1371/journal.pone.0093700

Ramil-Rego, P., Muñoz-Sobrino, C., Rodríguez-Guitián, M., Gómez-Orellana, L., (1998). Differences in the vegetation of the north of Iberian Peninsula during the last 16 000 years. Plant Ecol. 138, 41e62

Rodrigues, J.M., (2008). National eradication programme for the pinewood nematode. In: Mota, M.M., Vieira, P. (Eds.), Pine Wilt Disease: A Worldwide Threat to Forest Ecosystems. Springer, Dordrecht.

Sarmento, E.; Dores, V. A Fileira (2013). Florestal no Contexto da Economia Nacional: A Produtividadee a Especialização Regional. Silva Lusit., 21, 21–37.

Salvado, S. (2015) Eucalipto, a Árvore Que Reina Sobre a Floresta Nacional. Available online: https://www.rtp.pt/noticias/incendios-2015/eucalipto-a-arvore-que-reina-sobre-a-floresta-nacional_es86992

Shvidenko, A., Barber, C.V., Persson, R., (2005). Forest and woodland systems. In: Hassan, R., Scholes, R., Ash, N. (Eds.), Ecosystems and Human Well-being: Current State and Trends. Island Press, Washington D.C., pp. 585e621

Smith, G.F., Gittings, T., Wilson, M. et al. (2008) Identifying practical indicators of biodiversity for stand-level management of plantation forests. Biodivers Conserv 17:991–1015; DOI 10.1007/s10531-007-9274-3

Shugarth. H., (1984). A theory of forest dynamics. The ecological implications of forest succession models. Springer, New York, 278 p

Solomon, D. S., Hosmer, R., Hayslett, H. T., JR., (1986). A two-stage matrix model for predicting growth of forest stands in the Northeast. Can. J. For. Res. 16:521-528

Twery, M.J. and Weiskittel, A.R., (2013). 23 Forest-Management Modelling. Environmental Modelling: Finding Simplicity in Complexity, p.379; John Wiley & Sons, Ltd.

Van Hensbergen, H.J. (2006). Plantaciones, sustentabilidad y certificacio´n. Revista Ambiente y Desarollo 22: 21–28. (in Spanish)

Verheyen, K., Vanhellemont, M., Auge, H., Baeten, L., Baraloto, C., Barsoum, N., … Scherer-Lorenzen, M. (2016). Contributions of a global network of tree diversity experiments to sustainable forest plantations. *Ambio*, *45*(1), 29–41. https://doi.org/10.1007/s13280-015-0685-1

von Gadow, K. and Hui, G., (1999). Modelling forest development. Kluwer Academic Publishers, 213 p

Warren, R. et al. (2013). Quantifying the benefit of early climate change mitigation in avoiding biodiversity loss. Nature Clim. Chang. 3, 678–682

Winter, S.; McRoberts, R.E.; Bertini, R.; Chirici, G.; Bastrup-Birk, A.; Rondeux, J.; Brändli, U.B.; Marchetti, M. (2011). The Need for Harmonized Estimates of Forest biodiversity Indicators. In National Forest Inventories: Contributions to Forest Biodiversity Assessments; Springer: Dordrecht, the Netherlands

Winkel G. & Sotirov M. (2016). Whose integration is this? European forest policy between the gospel of coordination, institutional competition, and a new spirit of integration. Environment and Planning C: Government and Policy 2016, volume 34, pages 496– 514

Woods, A., K.D. Coates, and A. Hamann. (2005). Is an unprecedented dothistroma needle blight epidemic related to climate change? BioScience 55: 761–769.

Chapter II – Systematic review article

Stand structural characteristics are the most practical biodiversity indicators for forest management planning in Europe

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Review

Stand Structural Characteristics Are the Most Practical Biodiversity Indicators for Forest Management Planning in Europe

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Abstract: Including biodiversity assessments in forest management planning is becoming increasingly important due to the importance of biodiversity for forest ecosystem resilience provision and sustainable functioning. Here we investigated the potential to include biodiversity indicators into forest management planning in Europe. In particular, we aimed to (i) identify biodiversity indicators and data collection methods for biodiversity assessments at the stand and landscape levels, and (ii) evaluate the practicality of those indicators for forest management planning. We performed a literature review in which we screened 188 research studies published between 1990 and 2020. We selected 94 studies that fulfilled the inclusion criteria and examined in more detail. We considered three aspects of biodiversity: structure, composition, and function, and four forest management categories: unmanaged, managed, plantation, and silvopastoral. We used three criteria to evaluate the practicality of forest biodiversity indicators: cost-effectiveness, ease of application, and time-effectiveness. We identified differences in the practicality of biodiversity indicators for their incorporation into management plans. Stand-level indicators are more practical than landscape-level indicators. Moreover, structural biodiversity indicators (e.g., large trees, canopy openness, and old forest stands) are more useful in management plans than compositional indicators, as these are easily observable by non-professionals and can be obtained by forest inventories. Compositional indicators such are vascular plants, fungi, bryophyte, lichens, and invertebrate species are hard to identify by non-professionals and thus are impractical. Functional indicators (e.g., nutrient cycling) are not sufficiently addressed in the literature. Using recently updated existing databases (e.g., national forest inventories and bird atlases) is very time and cost-efficient. Remote sensing and other technology (e.g., smartphone applications) are promising for efficient data collection in the future. However, more research is needed to make these tools more accurate and applicable to a variety of ecological conditions and scales. Until then, forest stand structural variables derived from inventories can help improve management plans to prepare European forests towards an uncertain future.

Keywords: forest biodiversity indicators; forest composition; forest structure; forest ecosystem function; adaptive forest management; adaptive capacity; response diversity; practical indicators

1. Introduction

Forests host much of the world's biodiversity [1,2] and most terrestrial species inhabit these ecosystems [3,4]. Biodiversity is "the variety of life on Earth and the natural patterns it forms" [5]. Forest biodiversity is the variety of all forms of life and its organization within the forest area [6,7]. The world's terrestrial ecosystems encompass a forest area of 30.6%, with a declining trend [8]. In contrast, Europe's forest cover of currently 33% (215 million ha) is increasing [9]. The leading threat to biodiversity is the loss and the extreme alteration of 'once naturally dynamic forests' mainly due to competitive land use [10–12]. Even though there is limited proof of the current effect of climate change on biodiversity, researchers propose that climate change could outperform habitat destruction by land use and become the leading threat to biodiversity in the future [13,14].

Biodiversity is critical for forest ecosystem resilience as it determines the adaptive capacity of a community and sustainable provisioning of ecosystem services [15,16]. Therefore, to prepare forests for an uncertain future, biodiversity conservation became one of the most important aspects of forest management planning. Biodiversity includes a scope of spatial scales and has components related to the forest structure, (e.g., tree dimensions, canopy complexity, deadwood, and understory), composition (e.g., diversity within and between species, or species communities), and function (e.g., succession, decomposition, nutrient cycling) [17–20]. Due to the complexity of biodiversity across multiple components and scales, extensive evaluations of biodiversity are arduous and costly to embrace, even for stands of generally basic structure and organization [18–21]. Therefore, it is necessary to develop indicators to facilitate its assessment and integration into forest management plans [22]. Such indicators are 'surrogate measures of other components of forest biodiversity' that are used for the assessment of temporal and spatial changes of biodiversity [4,23]. Indicators need to be practical for the use of scientists and forest managers with different backgrounds. Practical indicators ideally are simple to evaluate, repeatable, economic, and ecologically important [24,25]. While numerous studies thoroughly addressed the importance of biodiversity indicators for biodiversity assessment [21,26,27], the practical usefulness of those indicators in sustainable forestry is still affected by perplexity and misconception [28–30]. Some of the leading problems that might cause the confusion about which indicator to choose from are (a) different spatial scales (i.e., the same indicator might not work for stand and landscape scale), (b) forest managers with different educational backgrounds (i.e., some managers might not understand why and where to use specific indicators), (c) unclear definition of the indicator and target levels (i.e., it is often unclear which biodiversity value is measured and according to what target level action needs to be undertaken) [30], and (d) facility to measure indicators repeatedly. Another challenge is to make a compromise between comprehensive biodiversity assessments and the cost-effectiveness of tools that can be used by forest managers to derive biodiversity indicators [12]. There are several reviews related to the use of biodiversity indicators in forest management planning in Europe [31–33]. However, no study has summarized yet the practicality of forest biodiversity indicators and methods for data collection. In particular, open questions are (i) what is the state of the art of the practicality of forest biodiversity assessment, and (ii) which biodiversity aspects are more and which are less amenable for forest managers, and why?

Our study sheds light into those questions. Based on a literature review, we discuss the practicality of biodiversity indicators and tools to collect biodiversity information in European forests. We distinguish between stand and landscape level as both are important in forest management planning. Moreover, we account for different forest management categories, including unmanaged forests, managed forests, plantations, and silvopastoral systems. The review includes four sections:

(1) Biodiversity indicators in European forest research
(2) Integrating biodiversity indicators into forest management
(3) Practical data collection
(4) Literature gaps and implications with forest management planning

2. Materials and Methods

2.1. Literature Review

We searched for scientific papers on forest biodiversity indicators used in forest management planning in Europe. We followed the PRISMA (Preferred Reporting Items for Systematic reviews and Meta-Analyses) statement in designing our review protocol [34,35]. The cutoff date for the inclusion of publications was February 16th, 2020. We used three combinations of search terms in Google Scholar and Web of Science: forest AND management AND planning AND biodiversity AND indicators; "forest management planning" AND biodiversity AND indicators; "forest management planning" AND "biodiversity indicators". The studies we found were published between 1990 and 2020. Additionally, we screened references cited by identified papers ('snowballing'). We evaluated the relevance of each study based on title and abstract first, and if necessary, we read the introduction and methods of the remaining studies to determine whether to include it in our review.

The inclusion criteria for the literature were:

- The research study was performed in Europe;
- Published in a peer-reviewed scientific journal;
- Written in English;
- The scale of research was stand or/and landscape;
- The focus was on forest biodiversity assessment;
- Biodiversity indicators and methods for data collection were clearly reported and extractable.

We selected 188 papers for further investigation, from which 94 papers were finally analyzed for the review (Figure 1). More details on the search strategy and the selection of the papers are listed as supporting information in supplementary material in Table S1.

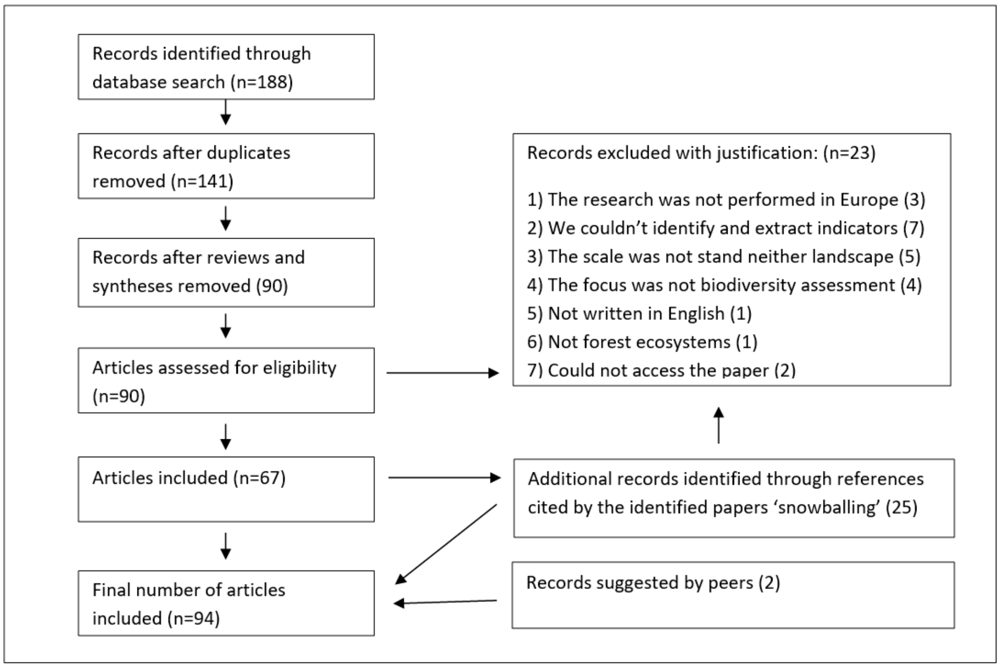

Figure 1. PRISMA (Preferred Reporting Items for Systematic reviews and Meta-Analyses) flow diagram of the literature review.

2.2. Data Extraction

From each study we extracted (a) the country/countries in which the study was conducted; (b) the type of biodiversity indicators (compositional, structural or functional); (c) the methods for data collection relevant for those indicators; (d) the scale at which the indicator was used; (e) type of forest where indicators were tested; (f) practicality of the indicator for management.

2.3. Evaluation of Practicality

We evaluated the practicality of biodiversity indicators and techniques for forest biodiversity data collection according to:

- Cost-effectiveness, i.e., what were the costs per hectare in Euros? How much workforce is required?
- Ease (simplicity) of application, i.e., are these indicators simple to use by forest managers with different backgrounds and can they identify the indicators (e.g., recognize the plant, animal species, or forest structural variables), and collect the data?
- Time-effectiveness, i.e., what is the time required for data gathering and assessment?

Ecological meaningfulness is another important aspect of practical indicators (apart from cost-effectiveness, ease, and time-effectiveness). However, we did not assess it here as different indicators may have multiple impacts on ecosystem functioning and services outcomes. For instance, saproxylic beetles and other decomposers are important for nutrient cycling and thus the productivity of forests [36], whereas structural complexity provides niches for several species with different life traits and thus structurally complex ecosystems exhibit a high resilience towards environmental change [36–39].

3. Biodiversity Indicators in European Forest Research

3.1. Geographic Distribution of Case Studies

The case studies were not evenly distributed across Europe. The majority of case studies were distributed around the Baltic Sea rim (Figure 2). The greatest number of case studies was from Italy (13 studies), Poland (12), Sweden (12), Germany (9), and Finland (8).

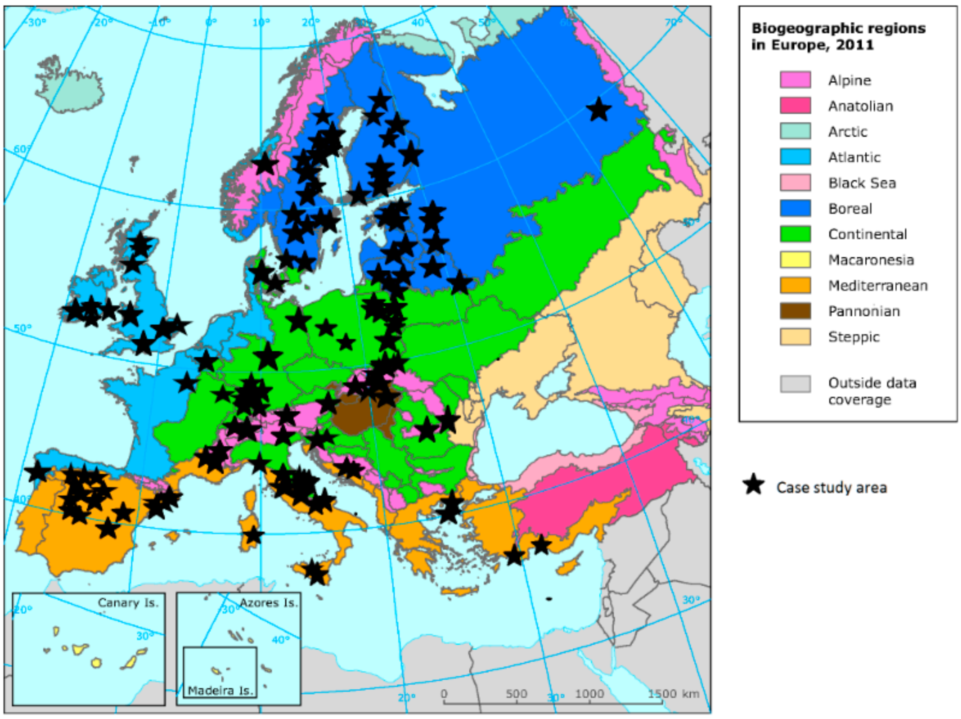

Figure 2. Distribution of case studies across biogeographic regions in Europe (note: nine studies had more than one case study area).

The practicality of forest biodiversity indicators and data collection methods was addressed in 57 studies (out of 94) (Figure 3). Out of the 57, 34 studies addressed practical data collection methods, 22 addressed practicality of indicators, and one addressed both. Six studies explicitly clearly stated that they have chosen indicators practical for forest managers or non-professionals and explained why [12,25,40–43].

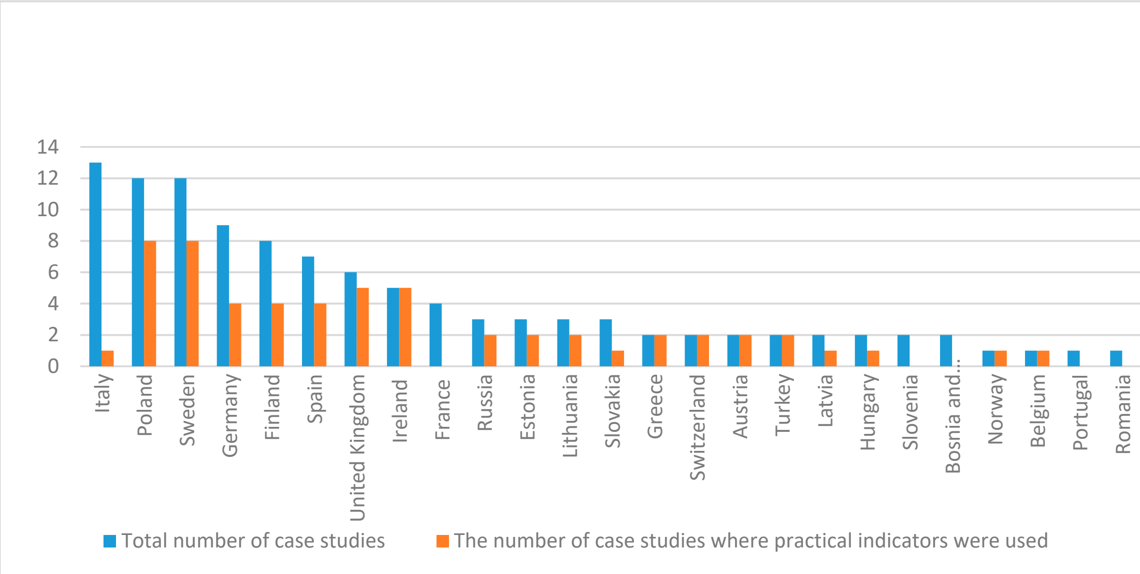

Figure 3. Ratio of the total number of case studies and the number of case studies that were addressing the practicality of the biodiversity indicators used. Analysis per country.

We did not identify any studies in the first decade of the study period (Figure 4). In the second decade, all the studies we selected were addressing practicality, while in the third decade approximately 60% of studies did not address practicality.

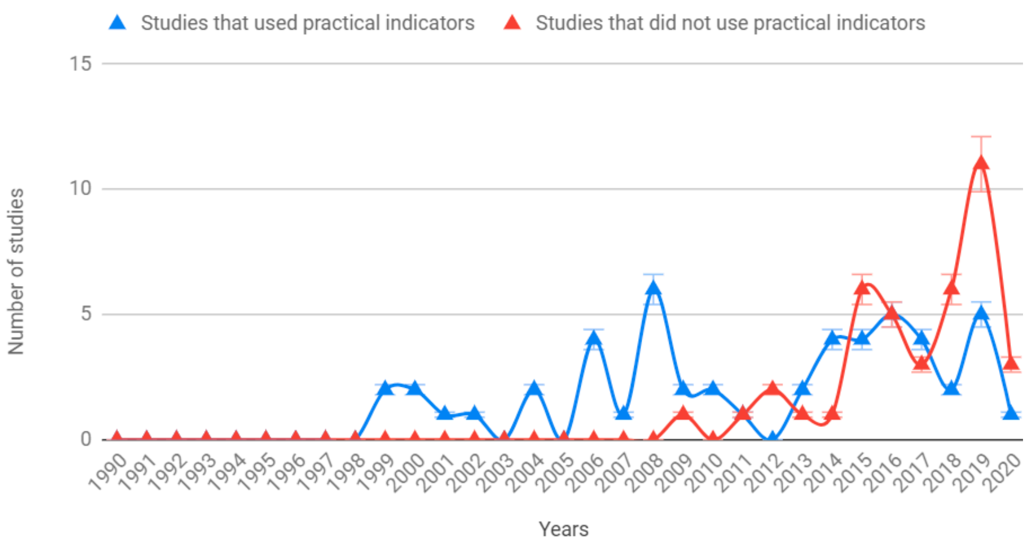

Figure 4. Temporal evolution of studies on biodiversity indicators, which address (blue) and do not address (red) practicability of the indicators and data collection methods applied. Note that our review includes studies until February 16 (i.e., the year 2020 is incomplete).

3.2. Forest Biodiversity Indicators Used in Studies

The most represented biodiversity attributes related to forest structure were forest inventory variables (Figure 5 and Table S2 in supporting material) and deadwood (DW). Regarding forest composition, taxa were the most represented where birds, bryophytes, and fungi dominated in the literature. Valuable flora and fauna were least covered. Generally, functional indicators (e.g., nitrogen, phosphorus, and other nutrients availabilities) were underrepresented in comparison to structural and compositional indicators. Studies were mainly addressing composition (39.6%), structure and composition (26.4%), structure (18.7%), structure, composition, and function (9.9%), structure and function (3.3%), composition and function (1.1%), and function (1.1%).

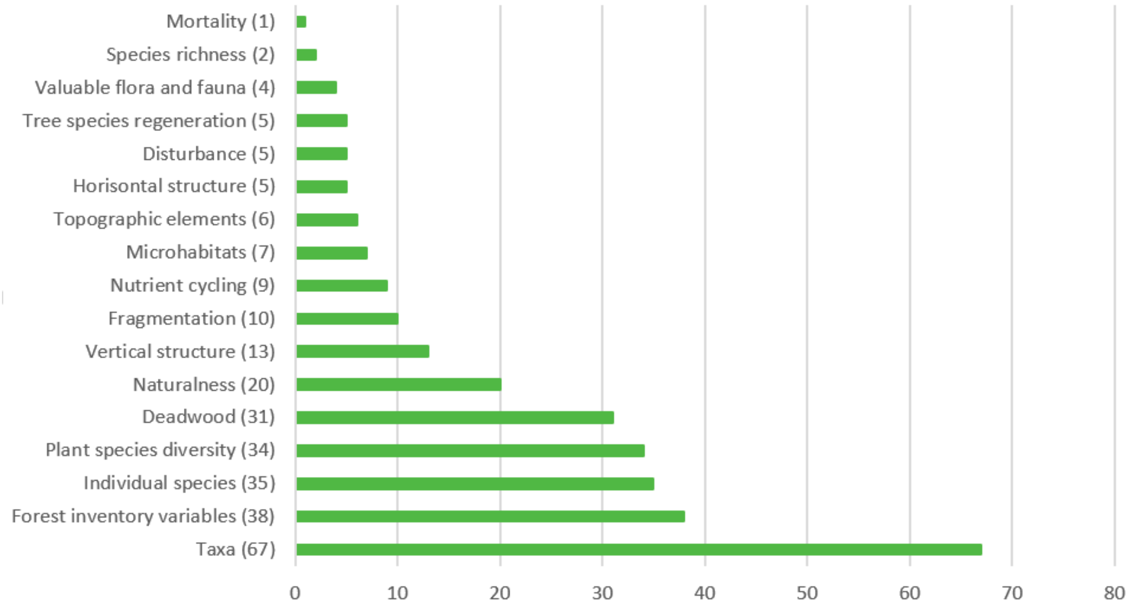

Figure 5. Number of biodiversity indicators used in reviewed studies. For details how indicators are explained and sorted by type of biodiversity (structure, composition, and function) see supporting material, Table S2.

Regarding the scale, the studies at the stand scale were dominant (55.3%) over the studies at the landscape scale (31.9%) in the 94 articles revised. Additionally, there were studies addressing stand and landscape levels simultaneously (12.8%). The greatest number of studies at the stand scale addressed managed conifer forests (18), followed by managed broadleaved (16) and managed mixed forests (15). At the landscape scale, managed conifer forests dominate the number of studies (12), followed by broadleaved managed forests (10) and mixed managed forests (9). Managed forests are fairly covered at both scales. The least number of studies are on silvopastoral systems at stand (1) and landscape scales (1) and broadleaved plantations at both scales (one each). We found no studies covering mixed plantations at the stand and landscape scale. Some studies did not clearly state the scale of analysis (6.4%) as well as type of forest (e.g., managed, unmanaged, plantation) (11.7%) being addressed.

4. Integrating Biodiversity Indicators into Forest Management

Only four studies reported costs and time effectiveness of biodiversity assessment [12,40,41,44]. Most of the studies reported practicality of indicators if these are 'easy to evaluate' but frequently without explaining why precisely. Additionally, the definition of 'easy to evaluate' varied among studies, where some linked it with ease to recognize the species [25,45] and others with the efforts to collect the data [4,46]. It can be concluded that the definition is fuzzy. However, it is challenging to improve clarity as practical features are rarely strictly independent, but rather correlated (e.g., easy to evaluate saves time and money).

4.1. The Practicality of Forest Biodiversity Indicators

4.1.1. Forest Species Composition

Tree species composition is easy to use (Table 1) and data is available from national (NFI) and local forest inventories [4]. Some studies have revealed that plant species diversity is a good proxy for overall biodiversity [4,47–49]. However, there were also studies reporting that assessing understory plant species composition is impractical, albeit without explaining why [50–52]. Possibly, identifying herbaceous or even shrub plant species may require specific and additional training as that is required for identifying tree species [43,53]. Some authors found plant species diversity and particularly understory vegetation (Table 2) less useful since it is often not included in NFIs or is not collected in a standardized way across countries [4,54]. We also found some authors reported vascular plants as more cost-effective indicators than other taxa [55,56]. In contrast, other authors found vascular plants and lichens hard to recognize by non-specialists [57,58]. Angelstam and Dönz-Breuss [12], for example, conducted studies in Austria, Poland, Russia, Italy, and Scotland and found that pendulous lichen is not a practical indicator at large scales, due to variable frequency of occurrence. Additionally, fungi are costly and time-consuming for the assessment due to seasonal variation and demand for professional knowledge for identification [59]. Moreover, fungi's fruiting bodies have a short life and are hard to detect [60,61]. Fungi monitoring typically requires many surveys within a year, but see Ambrosio et al. [61]. Nevertheless, species recognition by non-specialists may be possible in the future through molecular ecology method called DNA metabarcoding [62]. Barsoum et al. [62] applied this technique for assessing arthropod biodiversity and compared it with surrogate measures of biodiversity, with a high degree of correspondence.

Table 1. Practical indicators for various management categories at stand and landscape-level reported in the literature (* indicates the property; - indicates unknown).

Scale	Type of Biodiversity	Practical Indicator	Practical Aspects of Indicator			Management Category			
			Cost-efficient to Sample	Easy to Recognize	Time-efficient to Sample	Unmanaged Forest	Managed Forests	Plantations	Silvopastoral System
Stand	Compos.	Vascular plants [55,56}	*			*			
Stand	Compos.	Carabidae beetles [46]		*	*	*	*	*	*
Stand	Compos.	Spiders [25]		*				*	
Stand	Compos.	Hoverflies [25]		*				*	
Stand	Compos.	Tree species composition (richness, abundance, and diversity) [4,63]	*		*	*	*	*	*
Stand	Compos.	Shrub species composition (richness, abundance, and diversity) [63]	*		*	*	*	*	*
Stand	Struct.	Deadwood [2,12,20,63–65]	*	*	*	*	*		
Stand	Struct.	Canopy cover [2,4]	*	*	*	*	*	*	
Stand	Struct.	Special trees (occurrence of moss and lichen-covered, bent, damaged, hollow and forked trees) [12]		*		*	*	*	
Stand	Struct.	Proximity to native forests [25]	*	*	*	*	*	*	
Stand	Struct.	Large trees (mature trees) [12,63]		*		*	*		*
Stand	Struct.	Old forest stands [12]		*		*	*		
Stand	Struct.	Deciduous trees [12]		*		*	*		
Stand	Functi.	Stand age [2,12,25]	*	*	*			*	
Stand	Functi.	Available phosphorus (P) [25]						*	
Stand	Functi.	Elevation [25]	*	*	*			*	
Stand	Functi.	Uprooted trees [12]		*					
Stand	Functi.	Thinning frequency [25]	*	*	*			*	
Stand	Functi.	Wood-decaying bracket fungi [12]				*	*	*	
Lands.	Compos.	Birds [25,55,56,58]	*			*	*	*	*
Lands.	Compos.	Tree species richness [66]	*		*				*
Lands.	Compos.	Shrub species richness [66]	*		*				*
Lands.	Compos.	Valuable habitats [4]	*		*	*	*		
Lands.	Struct.	Patch shape, proximity, texture, diversity, and size [67]	-	-	-	-	-	-	-

Table 2. Impractical indicators for various management categories at stand scale reported in the literature (* indicates the property). Note that we did not identify impractical landscape scale indicators.

Scale	Type of Biodiversity	Impractical Indicator and Authors	Impractical Aspects of Indicator			Management Category			
			Expensive to Sample	Hard to Recognize	Time-consuming to Sample	Unmanaged Forest	Managed Forest	Plantation	Silvopastoral System
Stand	Compos.	Vascular plants [57,58]		*	*	*	*	*	*
Stand	Compos.	Lichens [12,57,58]		*		*	*	*	
Stand	Compos.	Fungi [59]	*	*	*	*	*		
Stand	Compos.	Bryophyte [25]		*		*	*		
Stand	Compos.	Invertebrate species [25]	*	*	*				
Stand	Compos.	Plant species diversity [4,54]		*	*	*	*	*	*
Stand	Compos.	Herb layer [68]			*	*	*	*	*

Carabidae (ground beetles) that feed on arthropods can be used as indicators of arthropod diversity [52,69]. Carabidae may have many advantages as indicators such as wide distribution across a range of terrestrial habitats, ecologically and taxonomically are a well-known group, and relatively easy to capture by trapping techniques [46,70–72]. Syrphidae (hoverflies) have also been suggested as indicators of a diversity of habitat conditions [52]. However, invertebrate species are challenging to identify by non-specialists, and time-consuming and expensive to sample [25].

Birds are frequently reported as the most practical biodiversity indicators, namely at larger scales [25,57,58] and are particularly relevant to assess habitat fragmentation [73–76]. However, birds are more suitable indicators of habitat structure than habitat species composition [77], since its abundance and richness is mainly correlated with forest structure and less with tree species composition [78–80].

Habitat requirements of birds are most commonly reported in comparison to all other taxa [58,81] which makes birds more practical than other taxa [55,56]. Nevertheless, for this reason, birds are so frequently used as surrogates of biodiversity, and not due to their 'unique intrinsic value' as biodiversity indicators [73,82]. However, some authors find birds and other large vertebrates unsuitable as indicators for overall biodiversity, as these are 'highly mobile generalists that lack established tolerance levels and correlations with ecosystem change' [46,83].

4.1.2. Forest Structure

Deadwood is an easily observable and thus practical indicator [2,20,64]. Pesonen et al. [41] compared the efficiency of various methods for the data collection on deadwood in Central Finland. The assessment of one type of deadwood material on a 400 m^2 area took on average 3.4 min, and the time required for the walk between plots was estimated at approximately 2 km h^{-1}.

Vertical structural diversity is often analyzed in forest management planning, and thus it is well known to forest managers [20,50,51]. Canopy cover, stand age, and the proximity of old woodlands are easily recognizable by non-specialists, ecologically meaningful, and suitable for various types of forests [2]. Large trees are also used as biodiversity indicators in some studies [12,63]. Further, these attributes are typically easily extractable from NFIs [4].

4.1.3. Forest Ecosystem Functioning

Practical functional forest biodiversity indicators at the stand scale, reported in the literature, are stand age, thinning frequency, wet microhabitats, elevation, and available phosphorus [25]. Stand age and thinning frequency can be extracted from stand registers or NFIs. However, the authors did not explain why they selected phosphorus as a potential indicator and how practical it is to assess. Additionally, while it is understandable that stand age reflects tree growth, it is not clear what is the contribution of thinning frequency, wet microhabitats, and elevation for forest ecosystem functioning. Uprooting, wood-decaying bracket fungi and ungulate browsing were used as practical indicators in a study of [12]. The authors indicated that the first two indicators are found together in most cases. However, fungi demand professional knowledge for the identification and are time-consuming and costly to monitor [59], which decreases the feasibility of this indicator for decaying processes. The authors did not explain why they selected browsing as a biodiversity indicator and why it is practical to use it.

4.1.4. Indicator and Umbrella Species

As it is mostly impossible to measure all aspects of biodiversity in practice, indicator species are often used as proxies for biodiversity [84–87]. Woodpeckers (*Dendrocopos* sp.) were used as an indicator species for biodiversity aspects at the landscape scale such as naturalness [88] and avian diversity [65]. Suitability of woodpeckers as an indicator species lies in the fact that this species depends on critical forest resources that are rarely found in managed forests (e.g., deadwood, large trees) and therefore it is expected that other taxa of conservation value (also dependent on these resources) could be found within the area of a woodpecker as well [89]. Treecreepers (*Certhia brachydactyla*) were used as an indicator of the effect of fragmentation on habitat suitability and abundance of local species. Among invertebrates, hoverflies were applied as an indicator species of the role of open spaces in maintaining biodiversity [90]. However, some authors argue that indicators should embrace different species, with diverging mobility and habitat preferences [30,91–93]. Therefore, Vangansbeke et al. [93] used a group of indicator species: crested tit (*Lophophanes cristatus*), coal tit (*Periparus ater*), nightjar (*Caprimulgus europaeus*), and common lizard (*Zootica vivipara*) to estimate biodiversity at stand-scale. The results demonstrated a significant relationship between biodiversity and occurrence probability of these species, except for the common lizard.

A similar aspect to indicator species is umbrella species. Conservation of umbrella species contributes to an array of other species depending on the same resources [89,94,95]. Typically, capercaillie (*Tetrao urogallus*) is tested for this role and particularly for avian biodiversity in temperate forests [96] or forests with rich diversity [97]. However, indicator and umbrella species should be taken cautiously, and the relationship between the indicator and indicandum needs to be tested and validated rigorously before using them [87,98–101].

4.1.5. Correlation and Surrogacy

Testing a correlation between different biodiversity aspects and therefore estimating if one aspect can be used as a surrogate for another may facilitate biodiversity assessment. The most used surrogates for biodiversity are bird species richness, and micro-habitat diversity as these are easy to sample and to quantify [45]. Stand structure parameters, soil class, and plant species composition are reported as positively correlated in a study by Gao et al. [4]. Landscape structural metrics such are patch shape, proximity, texture (e.g., land cover classes), diversity, and size, proved good indicators of overall species richness, woody plants, orthopterans, and reptiles in a study by Schindler et al. [67]. For further information on the results of correlation used in studies, see supplementary materials, Table S3.

To decrease the odds of creating erroneous or biased surrogates of biodiversity, some authors suggested the use of several taxonomic groups, instead of a single taxon [62,102,103]. It is also important to note that using many highly correlated indices creates a problem with the interpretation of the results and does not provide new information [104,105].

5. Practical Data Collection

Data collection was the most time-consuming part of biodiversity assessment (ca 70% of the time), while planning (ca 9%), data management (ca 15%), and analysis (ca 6%), required less time in studies from Scotland, Austria, Poland, Italy, and Russia [12]. The study also found that daily workforce of 23–43 person-days was needed for biodiversity assessment, where 23 were required in the case studies with lowland areas and 43 with hilly areas. The number of person-days also depended on the complexity of forests, weather conditions, and accessibility [12].

5.1. Sampling Methods

Sampling is a practical method for data collection since only a sub-set of interest (e.g., trees) and their spatial relationship is measured if the area is large [106]. However, some sampling methods are more practical than others. Namely, line-intersect sampling for deadwood volume estimation is less time consuming than other sampling methods such as circular sample plots [107] or systematically distributed sample plots [65,108,109]. However, Fridman and Walheim [65] stressed that 'the use of line intersect sampling would have caused problems with the determination of all variables needed for breaking down the DW-results on, e.g., stand age, forest type, and forest management operations performed'. Medium size plots (ca 25 m) proved to be more efficient than large plots (50 m) for deadwood volume estimation [41]. Further, Motz et al. [106] tested the efficiency of angle count and fix radius methods for tree diversity measures sampling. Fixed-radius plots were superior in measuring most indicators (e.g., stems per hectare and all spatially explicit diversity indices), and the effectiveness of this method increased with higher diameter variation. The results of this study refer to most representative Central European forests and apply to various scales.

5.2. National Forest Inventories (NFIs)

Using existing datasets, such as NFIs, saves time and money [4,68]. The data on forest structure from NFIs can be used for the development of ecological indicators for the assessment of valuable habitats and forest protection zones at the landscape level [68,110,111]. Martín-Queller et al. [66] used the data for gamma tree, and shrub species richness assessment in silvopastoral systems from the Third Spanish NFI (Table 3). Similarly, Torras and Saura [63] also used NFI data for the estimation of snags (stems/ha), large-diameter trees, shrub species abundance, shrub species richness, tree species richness, and tree species diversity in managed and unmanaged conifer, mixed, and broadleaved forests. However, the information related to the herb layer was not available in the NFI.

The data on deadwood is usually not adequately provided in NFIs such as in boreal countries [12]. Only recently, the information on lying and standing deadwood is starting to be included in forest inventories in some regions. Another challenge is the accuracy of information regarding the amount of deadwood, generated by severe disturbances [112]. Not all NFIs have the same standards for deadwood assessment, which creates the problem for comparison of the results from different inventories [65].

Table 3. Practical methods for data collection for different biodiversity attributes in three types of forests and types of biodiversity (structure, composition, and function) Note: not all the authors reported the types of forests clearly, and thus we put in the table information only about those which are clearly reported.

Scale and Type of Biodiversity	Indicator (Attribute)	Practical Data Collection Method for Managed Forests	Practical Data Collection Method for Unmanaged Forests	Practical Data Collection Method for Plantations	Practical Data Collection for Silvopastoral Systems
Stand Structure	Deadwood	Smartphone app [42]; NFI [63]	Smartphone app [42]; LiDAR [41]; NFI [63]; line-intersect sampling [107]	-	-
Structure	Big trees	Smartphone app [43]; NFI [63]	Smartphone app [43]; NFI [63]	-	-
Structure	Tree density	Smartphone app [43]	Smartphone app [43]	-	-
Structure	Micro-habitat diversity	Satellite images [45]	Satellite images [45]	-	-
Structure	Biomass	LiDAR [113]	LiDAR [113]	LiDAR [113]	-
Structure	Height	LiDAR [113]	LiDAR [113]	LiDAR [113]	-
Composition	Tree species diversity	Smartphone app [43]; NFI [63]	Smartphone app [43]; NFI [63]	-	-
Composition	Shrub species diversity	NFI [63]	NFI [63]	-	-
Composition	Herbs	Smartphone app [42]	Smartphone app [42]	-	-
Composition	Bird species richness	Satellite images [45]; National Ornithological Society [114]; National bird atlas [89]; gamekeeper register [115]	Satellite images [45]; National Ornithological Society [114]; gamekeeper register [115]	-	-
Composition	Fungal species richness	LiDAR [59]		-	-
Composition	Composition of forest-dwelling beetles	LiDAR [40]	LiDAR [40]	-	-
Function	Disturbances	Smartphone app [42]	Smartphone app [42]	-	-
	Regeneration	Smartphone app [42]	Smartphone app [42]		-
Landscape Structure	Deadwood	LiDAR + inventory data + aerial photographs [41]	LiDAR [41]	-	-
Structure	Micro-habitat diversity	Satellite images [45]	Satellite images [45]	-	-

Table 3. *Cont.*

Scale and Type of Biodiversity	Indicator (Attribute)	Practical Data Collection Method for Managed Forests	Practical Data Collection Method for Unmanaged Forests	Practical Data Collection Method for Plantations	Practical Data Collection for Silvopastoral Systems
Structure	Height	LiDAR [113]		-	-
Structure	Biomass	LiDAR [113]		-	-
Composition	Tree species			-	NFI [66]
Composition	Shrub species			-	NFI [66]
Composition	Bird species richness	Satellite images [45]; National Ornithological Society [114,116]; Museum of Natural History [68]; National bird atlas [87]	Satellite images [45]; National Ornithological Society [116]; National bird atlas [87]	-	-

5.3. Flora and Fauna Atlases

The data on bird populations of some studies investigated here were from National Ornithological Society [114,116], or Museums of Natural History [68]. Further, a National Bird Atlas was a source of information on bird composition and occupancy in some studies [87,89]. Data on bird composition were obtained from the National Forest Research Institute [97], and gamekeepers register [115]. Studies collecting fauna data harnessed atlases and registers for bird data only, while no data of other animal species were obtained from such sources. Most of the authors used bird maps and atlases as a baseline for the comparison with their own data. We have not found any study using vegetation maps or registers to investigate the flora of forests.

5.4. Remote Sensing

Collecting forest biodiversity data by remote sensing technology was applied in numerous studies and is considered broad-scale, accurate, and more cost-effective and faster than field sampling [40,41,45,59,68,113,117–119]. Ozdemir et al. [45] used satellite images to predict bird species richness and micro-habitat diversity in brutian pine (*Pinus brutia*) forest ecosystems in Turkey. They reported this approach as 'potentially' more efficient (faster and cost-efficient) than field measurements. Thers et al. [59] used the 2014–2015 Danish national airborne Light Detection and Ranging (LiDAR) scanning survey (Danmarks Højdemodel, DHM/Punktsky) for assessing fungal species richness. The results showed that it is 'promising that LiDAR-based variables hold information suitable for detecting major gradients in fungal richness and composition'. Müller and Brand [40] also used LiDAR to estimate habitat variables, to model activity, richness, and composition of assemblages of forest-dwelling beetles, and compare it to ground-based measurements. The results demonstrated that remote sensing provides more cost-effective data on biodiversity, even in mountain forests, in comparison to ground-based assessment.

Airborne LiDAR data is an accurate and feasible source of information on forest structure, such as height and biomass-related aspects, even on large and complex areas [113]. Pesonen et al. [41] found that the assessment of laying and standing deadwood by the airborne LiDAR is successful in conservation areas in Finland. However, regarding managed forest, this method can only be used as an auxiliary source, since the dynamics of deadwood in managed forests is quite different. Instead, using LiDAR data together with aerial photographs or stand-register data adds to deadwood sampling efficiency, even more than using only LiDAR data [41].

Some authors reported remote sensing as 'potentially' more efficient than field data collection [45,59], which leads to the conclusion that precise estimation of efficiency is lacking. However, Müller and Brand [40] provided a precise report on the costs of biodiversity assessment by remote sensing technology. They used LiDAR to estimate habitat variables, to model activity, richness, and composition of assemblages of forest-dwelling beetles, and compare it to ground-based measurements. The price for LiDAR assessment was 16 €/ha, the price for field data for habitat variables was 100€/ha and the price for data on beetles was 260€/ha. An additional advantage of LiDAR assessment is that costs decrease with the extent of the study area due to some fixed costs [40]. Given that costs of LiDAR variables are 5%–10% of ground-based measurement costs and that the proportion of explained variance compared with field measurement is high, remote sensing data has great potential in forest biodiversity modelling [40,120].

5.5. Camera Traps

Güthlin et al. [44] compared two techniques of field measurements for the estimation of red fox (*Vulpes vulpes*) abundance: camera traps and feces counting. The comparison took costs and precision into account. They divided costs into categories: initial costs, running costs for the equipment, travelling costs, and person-days. The total costs of feces counting were 17,057€, while camera traps cost 16,323 €. However, the precision of the camera traps was lower than the precision of the feces counts.

5.6. Smartphone Applications

A participatory GIS application for smartphones has been developed in Finland to collect the data on cultural, recreational, and biodiversity aspects from local people [43]. The application contains a component with information on forest management, ecology, and history of the area. In the second component, the visitors are asked to leave general feedback on landscape, infrastructure, or application functioning, and attach the photo. The third component is a game that is developed to attract more participants to share their opinion on the aesthetical and ecological state of the forest. In the game, the visitors are asked to answer the questions regarding structural and compositional biodiversity such as: should there be more trees; should there be bigger trees; should it be more open; should there be more tree species. No training or age limit is required, as long as the individual is capable of using a smartphone.

A similar smartphone application for forest biodiversity data collection was developed in Hungary to complement existing forest and conservation data with missing aspects such are canopy composition and structure, deadwood, herbs, microhabitats, disturbances, shrubs, and regeneration [42]. The advantage is that this method is simple, fast, and requires less effort than forest inventory. The method provides more detailed and reproducible information that is comparable with existing databases. It requires relatively low workforce per plot and has user-friendly direct database recording (no special equipment is required). The users do not need to be professionals; however, the training is necessary. The application was developed for low-mountain forests in Hungary; nevertheless, it could be modified for other forests.

Possible implications in regard to the use of applications for biodiversity data collection are in the structuring questions for the application users [43]. Hence, the more the questions are non-structured and open-ended, the harder it is for forest managers to analyze it further. Therefore, it would be optimal for forest managers to receive mainly straightforward numerical information. The advantage is that smartphone applications, as a technique, are very popular and will be unmistakably progressively used in the future [43]. Most of the forest stakeholders own a smartphone, which makes the use of it more amenable [43,121]. However, the stakeholders who tested the app in the study from Finland were secondary school students and teachers [43]. The study from Hungary presented the protocol and, thus there was no testing. Though, they stated that the app is intended for forest management and nature conservation purposes.

6. Literature Gaps and Implications for Forest Management Planning

Our review revealed a lack of studies on biodiversity indicators for the Atlantic region, particularly France. Eastern Europe is poorly represented, in particular Boreal, Continental, Steppic and Anatolian regions. In Central Europe, the case studies are mainly concentrated on the Alpine region, while Continental regions are missing. The research on biodiversity in Europe is still 'heavily biased toward countries with high gross domestic product' [122,123]. Additionally, there is a bias in case studies areas, where the studies from Central Europe repeatedly use the same study areas (e.g., the Black Forest, the Carpathians, the Alps).

Another problem our study identified here is an inconsistency in the definition of biodiversity. Many scientists have defined composition as biodiversity aspect, while, e.g., nutrient cycling or soil pH, they defined as environmental variables. Even though these interpretations are not erroneous, there is a problem with comparing the results from different studies that did not define biodiversity aspects equally. Even worse, the biodiversity definition is often missing in studies. Therefore, Feest [77,124] suggests the development of a common measure of biodiversity consisting of indices such are species richness, evenness, population, biomass, and conservation value, to facilitate biodiversity assessments. Additionally, species interactions should assemble the list of biodiversity aspects [124,125].

Most authors focused more on data collection practical methods and less on indicators themselves. There is a need to test and report more information on time and cost-efficiency and amenability of indicators. Though, it is clear that data collection is the most time-consuming part of biodiversity assessment [12]. Most of the studies focused on the response of biodiversity on some pressure and very few on measuring the state of biodiversity per se. In such way, the indicators are mainly indirect [77] and only reflect the changes in biodiversity response to certain pressure over time.

We found that plant species diversity is often used as a surrogate for overall biodiversity [4,47–49]. However, herb species may be particularly hard to identify by non-specialists [57,58]. Data collection is expensive and mostly unavailable from NFIs [4]. Therefore, future research needs to focus on resolving which plant species or taxonomic groups may be easier to identify by non-specialists and from such species assess those working better as biodiversity indicators. Additionally, DNA metabarcoding is an innovative method for species recognizing and seems promising for non-specialist use [62]. Combining this method with smartphone applications for species identification, that are lately progressively developed, would be a more cost-efficient way for overcoming taxonomic impediment then investing in taxonomic training.

Further, we found only one study that reported functional indicators at the landscape scale, and in general, a few studies reported compositional biodiversity at the landscape scale. The practicality of landscape biodiversity indicators needs to be addressed in future research, particularly functional indicators which seems to be underrepresented and rarely estimated in European research studies. Since there are only a few studies that used functional biodiversity indicators, there is not enough evidence to discuss the practicality of this category of indicators. We assume that underlying reason is the fact that functional indicators are very complex and require expensive laboratory equipment for the assessment. No studies included nitrogen deposition as an indicator, even though that Nitrogen Critical Load Exceedance (NCLE) was projected to have greater consequences to European biodiversity than global change [77,124,126–129]. For instance, nitrogen deposition could create new interconnections between lichens and forest stand properties [130,131].

With a view on data collection, more research and investment into modern technologies (LiDAR and smartphone apps) is needed to improve the accuracy and applicability. Overall, future research should integrate forest structure, tree species composition, and ecosystem functionality to provide broader knowledge on habitat assessment and modelling [113].

7. Conclusions

Vascular plants and generally herb layer, fungi, bryophyte, lichens, and invertebrates are mostly reported as impractical biodiversity indicators as their identification is challenging for non-professionals, and the data is mainly unavailable in NFIs. This is problematic since plant species diversity is a key indicator of overall biodiversity [4,47–49,63]. Structural variables (e.g., old stands, large trees, and canopy cover) at stand scale [2,4,12,25] are easily observable by non-professionals and are available in NFIs, which makes them practical. Deadwood is easily observable at stand [2,20,63] and landscape scales [67,132,133], but often unavailable in NFIs. Generally, stand-level biodiversity indicators are more practical for forest managers than landscape-level indicators, because forest management is primarily acting on stand scale [25,134]. Even though there is great relevance in maintaining biodiversity in production forest stands [135], landscape level is also important and requires more attention in future research. Additionally, functional indicators are not sufficiently reported in the literature, and, thus, there is not enough evidence to discuss their practicality. Using the results from correlations and surrogacy relationships between indicators is a straightforward way to decrease the time and expenses for biodiversity assessments. However, surrogacy relationships can be spurious and must be tested under multifarious conditions [87,98–101].

Choosing the right sampling method and the right size of a plot can also contribute to efficient biodiversity assessments [41]. NFIs are a good source of data on indicators such as tree and shrub species diversity, and stand age [2,63,66]. Bird registers and atlases can be very useful and are mainly used as a baseline in bird biodiversity research. Remote sensing technique for data collection is more cost-effective and faster than field one at both stand and landscape scales, though more research is needed to provide a precise estimation of biodiversity. Another advantage is that remotely sensed data is easier to update in comparison to traditional ways of collecting data, which simplifies monitoring and reduces costs [123]. Smartphone applications are promising tools for biodiversity assessment by non-professionals, but future research needs to focus on increasing their accuracy for the assessment on all levels of biodiversity, at different scales and ecological conditions. A more accurate and precise estimation of biodiversity will help scientists and practitioners to design biodiversity-oriented management plans that ultimately increase the adaptive capacity of forests towards future environmental changes. Additionally, it will facilitate testing policy goal implementation and clarify setting new policy targets in biodiversity conservation.

Supplementary Materials: The following are available online at http://www.mdpi.com/1999-4907/11/3/343/s1, Table S1: Search strategy table; list of papers included in review/lists of papers excluded from the review; Table S2: Biodiversity attributes and indicators used in the literature sorted by type of biodiversity (structure, composition, function): Table S3: Correlations; File S1: Definitions.

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References

1. Kapos, V.; Iremonger, S. Achieving global and regional perspectives on forest biodiversity and conservation. In *Assessment of Biodiversity for Improved Forest Planning*; Bachman, P., Kohl, M., Paivinen, R., Eds.; Springer: Dordrecht, the Netherlands, 1998; pp. 3–14.
2. Coote, L.; Dietzsch, A.C.; Wilson, M.W.; Graham, C.T.; Fuller, L.; Walsh, A.T.; Irwin, S.; Kelly, D.L.; Mitchell, F.J.; Kelly, T.C.; et al. Testing indicators of biodiversity for plantation forests. *Ecol. Ind.* **2013**, *32*, 107–115. [CrossRef]
3. Ozanne, C.H.P.; Anhuf, D.; Boulter, S.L.; Keller, M.; Kitching, R.L.; Körner, C.; Meinzer, F.C.; Mitchell, A.W.; Nakashizuka, T.; Silva Dias, P.L.; et al. Biodiversity meets the atmosphere: a global view of forest canopies. *Science* **2003**, *301*, 183–186. [CrossRef] [PubMed]
4. Gao, T.; Hedblom, M.; Emilsson, T.; Nielsen, A.B. The role of forest stand structure as biodiversity indicator. *For. Ecol. Manage.* **2014**, *330*, 82–93. [CrossRef]
5. CBD, Convention on Biological Diversity 2000. Available online: https://www.cbd.int/convention/guide/?id=web (accessed on 28 January 2020).
6. Hunter, M.L., Jr. *Wildlife, forests, and forestry: Principles of managing forests for biological Diversity*; Prentice-Hall: Englewood Cliffs, NJ, USA, 1990; p. 370.
7. Winter, S.; McRoberts, R.E.; Bertini, R.; Chirici, G.; Bastrup-Birk, A.; Rondeux, J.; Brändli, U.B.; Marchetti, M. The Need for Harmonized Estimates of Forest biodiversity Indicators. In *National Forest Inventories: Contributions to Forest Biodiversity Assessments*; Springer: Dordrecht, the Netherlands, 2011.
8. FAO. The State of World's Forests. 2018. Available online: http://www.fao.org/state-of-forests/en/ (accessed on 27 July 2019).
9. Forest Europe. State of Europe's forests 2015. In Proceedings of the Ministerial Conference on the Protection of Forests in Europe, Madrid, Spain, 20–1 October 2015.

10. Peterken, G. *Natural Woodland: Ecology and Conservation in Northern Temperate Regions*; Cambridge University Press: Cambridge, UK, 1996.
11. Anonymous. Decision No 1600/2002/EC of the European parliament and of the council of 22 July 2002 laying down the Sixth Community Environment Action Programme. *Off. J. Eur. Communities* **2002**, *10*.
12. Angelstam, P.; Dönz-Breuss, M. Measuring Forest Biodiversity at the Stand Scale: An Evaluation of Indicators in European Forest History Gradients. *Ecol. Bull.* **2004**, *51*, 305–332.
13. Leadley, P.; Pereira, H.M.; Alkemade, R.; Fernandez-Manjarres, J.F.; Proenca, V.; Scharlemann, J.P.W.; Walpole, M.J. Biodiversity scenarios: projections of 21st century change in biodiversity and associated ecosystem services. *Tech. Ser. 50* **2010**, 132.
14. Bellard, C.; Bertelsmeier, C.; Leadley, P.; Thuiller, W.; Courchamp, F. Impacts of climate change on the future of biodiversity. *Ecol. Lett.* **2012**, *15.4*, 365–377. [CrossRef]
15. MEA-Millennium Ecosystem Assessment. 2005. Available online: https://millenniumassessment.org/en/Framework.html (accessed on 1 May 2019).
16. Proença, V.M.; Pereira, H.M.; Guilherme, J.; Vicente, J. Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal. *Acta Oecol.* **2010**, *36*, 219–226. [CrossRef]
17. Anonymous. *Criteria and Indicators for the Conservation and Sustainable Management of Temperate and Boreal Forests*; The Montreal process; Canadian Forest Service: Quebec, Canada, 1995.
18. Ferris-Kaan, R.; Peace, A.J.; Humphrey, J.W. Assessing structural diversity in managed forests. In *Assessment of Biodiversity for Improved Forest Planning. European Forest Institute Proceedings No. 18*; Bachmann, P., Kohl, M., Paivinen, R., Eds.; Springer: Dordrecht, the Netherlands, 1998; pp. 331–342.
19. Heywood, V.H.; Baste, I. Introduction. In *Global Biodiversity Assessment. United Nations Environment Programme*; Heywood, V.H., Ed.; Cambridge University Press: Cambridge, UK, 1995; pp. 1–2.
20. Ferris, R.; Peace, A.J.; Newton, A.C. Macrofungal communities of lowland Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* (L.) Karsten.) plantations in England: relationships with site factors and stand structure. *For. Ecol. Manage.* **2000**, *131*, 255–267. [CrossRef]
21. Noss, R.F. Indicators for monitoring biodiversity: A hierarchical approach. *Conserv. Biol.* **1990**, *4*, 355–364. [CrossRef]
22. Morelli, F. Indicator species for avian biodiversity hotspots: Combination of specialists and generalists is necessary in less natural environments. *J. Nat. Cons.* **2015**, *27*, 54–62. [CrossRef]
23. Boutin, S.; Haughland, D.L.; Schieck, J.; Herbers, J.; Bayne, E. A new approach to forest biodiversity monitoring in Canada. *For. Ecol. Manage.* **2009**, *258*, S168–S175. [CrossRef]
24. Ferris, R.; Humphrey, J.W. A review of potential biodiversity indicators for application in British forests. *Forestry* **1999**, *72*, 313–328. [CrossRef]
25. Smith, G.F.; Gittings, T.; Wilson, M.; French, L.; Oxbrough, A.; O'Donoghue, S.; Iremonger, S. Identifying practical indicators of biodiversity for stand-level management of plantation forests. *Biodiv. Cons.* **2008**, *17*, 991–1015. [CrossRef]
26. Noss, R.F. Assessing and monitoring forest biodiversity: A suggested framework and indicators. *For. Ecol. Manage.* **1999**, *115*, 135–146. [CrossRef]
27. Lindenmayer, D.B.; Margules, C.R.; Botkin, D.B. Indicators of biodiversity for ecologically sustainable forest management. *Cons. Bio.* **2000**, *14*, 941–950. [CrossRef]
28. Failing, L.; Gregory, R. Ten common mistakes in designing biodiversity indicators for forest policy. *J. Environ. Manage.* **2003**, *68*, 121–132. [CrossRef]
29. Whitman, A.A.; Hagan, J.M. *Final Report to the National Commission on Science For sustainable Forestry: Biodiversity Indicators for Sustainable Forestry*; National Center for Science and the Environment: Washington, DC, USA, 2003; 48p.
30. Heink, U.; Kowarik, I. What are indicators? On the definition of indicators in ecology and environmental planning. *Ecol. Ind.* **2010**, *10*, 584–593. [CrossRef]
31. Gao, T.; Nielsen, A.B.; Hedblom, M. Reviewing the strength of evidence of biodiversity indicators for forest ecosystems in Europe. *Ecol. Ind.* **2015**, *57*, 420–434. [CrossRef]
32. Ezquerro, M.; Pardos, M.; Diaz-Balteiro, L. Operational research techniques used for addressing biodiversity objectives into forest management: an overview. *Forests* **2016**, *7*, 229. [CrossRef]
33. Lexer, M.J.; Lexer, W.; Hasenauer, H. The use of forest models for biodiversity assessment at the stand level. *For. Syst.* **2000**, *9*, 297–316.

34. Moher, D.; Liberati, A.; Tetzlaff, J.; Altman, D.G. The PRISMA Group. Preferred Reporting Items for Systematic Reviews and Meta-Analyses: The PRISMA Statement. *PLoS Med.* **2009**, *6*, e1000097. [CrossRef] [PubMed]
35. Xiao, Y.; Watson, M. Guidance on Conducting a Systematic Literature Review. *J. Plann. Edu. Rese.* **2017**, *39*, 1–20. [CrossRef]
36. Parisi, F.; Pioli, S.; Lombardi, F.; Fravolini, G.; Marchetti, M.; Tognetti, R. Linking deadwood traits with saproxylic invertebrates and fungi in European forests—A review. *iForest* **2018**, *11*, 423–436. [CrossRef]
37. Winter, S.; Fischer, H.S.; Fischer, A. Relative quantitative reference approach for naturalness assessments of forests. *For. Ecol. Manage.* **2010**, *259*, 1624–1632. [CrossRef]
38. Bani, A.; Pioli, S.; Ventura, M.; Panzacchi, P.; Borruso, L.; Tognetti, R.; Tonon, G.; Brusetti, L. The role of microbial community in the decomposition of leaf litter and deadwood. *Appl. Soil Ecol.* **2018**, *126*, 75–84. [CrossRef]
39. Parisi, F.; Di Febbraro, M.; Lombardi, F.; Biscaccianti, A.B.; Campanaro, A.; Tognetti, R.; Marchetti, M. Relationships between stand structural attributes and saproxylic beetle abundance in a Mediterranean broadleaved mixed forest. *For. Ecol. Manage.* **2019**, *432*, 957–966. [CrossRef]
40. Müller, M.; Brand, R. Assessing biodiversity by remote sensing in mountainous terrain: The potential of LiDAR to predict forest beetle assemblages. *J. Appli. Ecol.* **2009**, *46*, 897–905. [CrossRef]
41. Pesonen, A.; Kangas, A.; Maltamo, M.; Packalen, P. Effects of auxiliary data source and inventory unit size on the efficiency of sample-based coarse woody debris inventory. *For. Ecol. Manage.* **2010**, *259*, 1890–1899. [CrossRef]
42. Standovár, T.; Szmorad, F.; Kovács, B.; Kelemen, K.; Plattner, M.; Roth, T.; Pataki, Z. A novel forest state assessment methodology to support conservation and forest management planning. *Commun. Ecol.* **2016**, *17*, 167–177. [CrossRef]
43. Kangas, A.; Rasinmäk, J.; Eyvindson, K.; Chambers, P. A Mobile Phone Application for the Collection of Opinion Data for Forest Planning Purposes. *Envir. Manage.* **2015**, *55*, 961–971. [CrossRef] [PubMed]
44. Güthlin, D.; Storch, I.; Küchenhoff, H. Toward Reliable Estimates of Abundance: Comparing Index Methods to Assess the Abundance of a Mammalian Predator. *PLoS ONE* **2014**, *9*, e94537. [CrossRef] [PubMed]
45. Ozdemir, I.; Mert, A.; Ozkan, U.Y.; Aksan, S.; Unal, Y. Predicting bird species richness and micro-habitat diversity using satellite data. *For. Ecol. Manage.* **2018**, *424*, 483–493. [CrossRef]
46. Mullen, K.; O'Halloran, J.; Breen, J.; Giller, P.; Pithon, J.; Kelly, T. Distribution and composition of carabid beetle (Coleoptera, Carabidae) communities across the plantation forest cycle—Implications for management. *For. Ecol. Manage.* **2008**, *256*, 624–632. [CrossRef]
47. Simonson, S.E.; Opler, P.A.; Stohlgren, T.J.; Chong, G.W. Rapid assessment of butterfly diversity in a montane landscape. *Biodiv. Conserv.* **2001**, *10*, 1369–1386. [CrossRef]
48. Sauberer, N.; Zulka, K.P.; Abensperg-Traun, M.; Berg, H.M.; Bieringer, G.; Milasowszky, N.; Moser, D.; Plutzar, C.; Pollheimer, M.; Storch, C.; et al. Surrogate taxa for biodiversity in agricultural landscapes of eastern Austria. *Biol. Conserv.* **2004**, *117*, 181–190. [CrossRef]
49. Bräuniger, C.; Knapp, S.; Kühn, I.; Klotz, S. Testing taxonomic and landscape surrogates for biodiversity in an urban setting. *Landsc. Urban Plan.* **2010**, *97*, 283–295. [CrossRef]
50. Ratcliffe, P.R. *Biodiversity in Britain's Forests*; Forestry Authority: Edinburgh, Scotland, 1993.
51. Arsenault, A.; Bradfield, G.E. Structural-compositional variation in three age-classes of temperate rainforests in southern coastal British Columbia. *Can. J. Bot.* **1995**, *73*, 54–64. [CrossRef]
52. Humphrey, J.W.; Hawes, C.; Peace, A.J. Relationships between insect diversity and habitat characteristics in plantation forests. *For. Ecol. Manage.* **1999**, *113*, 11–21. [CrossRef]
53. Michel, A.K.; Winter, S. Tree microhabitat structures as indicators of biodiversity in Douglas-fir forests of different stand ages and management histories in the Pacific Northwest. USA. *For. Ecol. Manage.* **2009**, *257*, 1453–1464. [CrossRef]
54. Chirici, G.; McRoberts, R.E.; Winter, S.; Bertini, R.; Brändli, U.B.; Asensio, I.A.; Bastrup-Birk, A.; Rondeux, J.; Barsoum, N.; Marchetti, M. National forest inventory contributions to forest biodiversity monitoring. *For. Sci.* **2012**, *58*, 257–268. [CrossRef]
55. Juutinen, A.; Mönkkönen, M. Testing alternative indicators for biodiversity conservation in old-growth boreal forests: ecology and economics. *Ecol. Econom.* **2004**, *50*, 35–48. [CrossRef]

56. Thingstad, P.G.; Daverdin, M.; Speed, J.D.M. Modelling Capercaillie Tetrao urogallus lek sites distribution and their limiting factors in a central Norwegian managed forest landscape. *Ornis Norv.* **2018**, *41*, 1–12. [CrossRef]
57. Uliczka, H.; Angelstam, P.; Roberge, J.M. Indicator species and biodiversity monitoring systems for non-industrial private forest owners: is there a communication problem? *Ecol. Bull.* **2004**, *51*, 379–384.
58. Naumov, V.; Manton, M.; Elbakidze, M.; Rendenieks, Z.; Priednieks, J.; Uhlianets, S.; Angelstam, P. How to reconcile wood production and biodiversity conservation? The Pan-European boreal forest history gradient as an "experiment". *J. Envir. Manage.* **2018**, *218*, 1–13. [CrossRef]
59. Thers, H.; Brunbjerg, A.K.; Læssøe, T.; Ejrnæs, R.; Bøcher, P.K.; Svenning, J.C. Lidar-derived variables as a proxy for fungal species richness and composition in temperate Northern Europe. *Rem. Sens. Envir.* **2017**, *200*, 102–113. [CrossRef]
60. Kirk, P.M.; Cannon, P.F.; Minter, D.W.; Stalpers, J.A. *Dictionary of the Fungi*, 10th ed.; CAB Internation Wallingford: Wallingford, UK, 2008; p. 771.
61. Ambrosio, E.; Mariotti, M.G.; Zotti, M.; Cecchi, G.; Di Piazza, S.; Feest, A. Measuring macrofungal biodiversity quality using two different survey approaches: A case study in broadleaf Mediterranean forests. *Ecol. Ind.* **2018**, *85*, 1210–1230. [CrossRef]
62. Barsoum, N.; Bruce, C.; Forster, J.; Ji, Y.Q.; Yu, D.W. The devil is in the detail: Metabarcoding of arthropods provides a sensitive measure of biodiversity response to forest stand composition compared with surrogate measures of biodiversity. *Ecol. Ind.* **2019**, *101*, 313–323. [CrossRef]
63. Torras, O.; Saura, S. Effects of silvicultural treatments on forest biodiversity indicators in the Mediterranean. *For. Ecol. Manage.* **2008**, *255*, 3322–3330. [CrossRef]
64. Hodge, S.J.; Peterken, G.F. Deadwood in British forests: priorities and a strategy. *Forestry* **1998**, *71*, 99–112. [CrossRef]
65. Fridman, J.; Walheim, M. Amount, structure, and dynamics of dead wood on managed forestland in Sweden. *For. Ecol. Manage.* **2000**, *131*, 23–36. [CrossRef]
66. Martín-Queller, E.; Gil-Tena, A.; Saura, S. Species richness of woody plants in the landscapes of Central Spain: the role of management disturbances, environment and non-stationarity. *J. Veg. Sci.* **2011**, *2*, 238–250. [CrossRef]
67. Schindler, S.; von Wehrden, H.; Poirazidis, K.; Wrbka, T.; Kati, V. Multiscale performance of landscape metrics as indicators of species richness of plants, insects and vertebrates. *Ecol. Ind.* **2013**, *31*, 41–48. [CrossRef]
68. Romero-Calcerrada, R.; Luque, S. Habitat quality assessment using Weights-of-Evidence based GIS modelling: The case of Picoides tridactylus as species indicator of the biodiversity value of the Finnish forest. *Ecol. Modell.* **2006**, *196*, 62–76. [CrossRef]
69. Day, K.R.; Carthy, J. Changes in carabid beetle communities accompanying a rotation of Sitka spruce. *Agric. Ecosys. Environ.* **1988**, *24*, 407–415. [CrossRef]
70. Thiele, H.U. *Carabid Beetles in their Environments: A Study on Habitat Selection by Adaptations in Physiology and Behaviour*; Springer: Berlin, Germany, 1997; p. 369.
71. Butterfield, J. Carabid community succession during the forestry cycle in conifer plantations. *Ecography* **1997**, *20*, 614–625. [CrossRef]
72. Vanbergen, A.J.; Woodcock, B.A.; Watt, A.D. Effect of land-use heterogeneity on carabid communities at the landscape scale. *Ecography* **2005**, *28*, 3–16. [CrossRef]
73. Gregory, R.D.; Voříšek, P.; Noble, D.G.; Van Strien, A.; Klvaňová, A.; Eaton, M.; Burfield, I.J. The generation and use of bird population indicators in Europe. *Bird Conserv. Internat.* **2008**, *18*, 223–244. [CrossRef]
74. Carrillo-Rubio, E.; Kery, M.; Morreale, S.J. Use of multispecies occupancy models to evaluate the response of bird communities to forest degradation associated with logging. *Conserv. Bio.* **2014**, *28*, 1034–1044. [CrossRef]
75. Czeszczewik, D.; Zub, K.; Stanski, T.; Sahel, M.; Kapusta, A.; Walankiewicz, W. Effects of forest management on bird assemblages in the Bialowieza Forest, Poland. *iForest* **2014**, *8*, 377–385. Available online: http://www.sisef.it/iforest/contents/?id=ifor1212-007 (accessed on 9 January 2019). [CrossRef]
76. Basile, M.; Valerio, F.; Balestrieri, R.; Posillico, M.; Bucci, R.; Altea, T.; Matteucci, G. Patchiness of forest landscape can predict species distribution better than abundance: the case of a forest-dwelling passerine, the short-toed treecreeper, in central Italy. *PeerJ* **2016**, *4*, e2398. [CrossRef] [PubMed]
77. Feest, A. The Utility of the Streamlining European Biodiversity Indicators 2010 (SEBI 2010). *Ecol. Ind.* **2013**, *28*, 16–21. [CrossRef]

78. Mosimann, P.; Naef-Daenzer, B.; Blattner, M. Die Zusammensetzung der Avifauna in typischen Waldgesellschaften der Schweiz. *D. Ornitholog. Beobachter* **1987**, *84*, 275–299.
79. Müller-Buser, M. Die Avifauna der Bergföhrenwälder des Schweizerischen Nationalparks im Ofenpassgebiet. *D. Ornitholog. Beobachter* **2002**, *99*, 1–18.
80. Bujoczek, M.; Rybicka, J.; Bujoczek, L. Effects of disturbances in a subalpine forest on its structural indicators and bird diversity. *Ecol. Ind.* **2020**, *112*, 106126. [CrossRef]
81. Angelstam, P.; Dönz-Breuss, M.; Roberge, J.M. Targets and tools for the maintenance of forest biodiversity: An introduction. *Ecol. Bull.* **2004**, *51*, 11–24.
82. Lešo, P.; Kropil, R.; Kajtoch, Ł. Effects of Forest Management on Bird Assemblages in Oak-Dominated Stands of the Western Carpathians—Refuges for Rare Species. *For. Ecol. Manage.* **2019**, *453*, 117620. [CrossRef]
83. Hilty, J.; Merenleder, A. Faunal indicator taxa selection for monitoring ecosystem health. *Bio. Conser.* **2000**, *92*, 185–197. [CrossRef]
84. Hawskworth, D.L.; Rose, F. Qualitative scale for estimating sulphur dioxide in England and Wales using epiphytic lichens. *Nature* **1970**, *227*, 145–148.
85. Block, W.M.; Brennan, L.A.; Gutiérrez, R.J. The use of guilds and guild-indicators for assessing habitat suitability. In *Wildlife*; Verner, J., Morrison, M.L., Ralph, C.J., Eds.; The University of Wisconsin Press: Madisonm, WI, USA, 1996; pp. 109–113.
86. Furness, R.W.; Greenwood, J.J.D. (Eds.) *Birds as Monitors of Environmental Change*; Chapman & Hall: London, UK, 1993.
87. Mikusiński, G.; Gromadzki, M.; Chylarecki, P. Woodpeckers as indicators of forest bird diversity. *Conserv. Biol.* **2001**, *15*, 208–217. [CrossRef]
88. Roberge, J.-M.; Angelstam, P.; Villard, M.-A. Specialised woodpeckers and naturalness in hemiboreal forests—Deriving quantitative targets for conservation planning. *Bio. Conserv.* **2008**, *141*, 997–1012. [CrossRef]
89. Roberge, J.-M.; Mikusinski, G.; Svensson, S. The white-backed woodpecker: umbrella species for forest conservation planning? *Biodiv. Conserv.* **2008**, *17*, 2479–2494. [CrossRef]
90. Gittings, T.; O'Halloran, J.; Kelly, T.; Giller, P.S. The Contribution of Open Spaces to the Maintenance of Hoverfly (Diptera, Syrphidae) Biodiversity in Irish Plantation Forests. *For. Ecol. Manage.* **2006**, *237*, 290–300. [CrossRef]
91. Rodriguez-Prieto, I.; Bennett, V.J.; Zollner, P.A.; Mycroft, M.; List, M.; Fernández-Juricic, E. Simulating the responses of forest bird species to multi-use recreational trails. *Landsc. Urban. Plan.* **2014**, *127*, 164–172. [CrossRef]
92. Pakkala, T.; Linden, A.; Tiainen, J.; Tomppo, E.; Kouki, J. Indicators of forest biodiversity: which bird species predict high breeding bird assemblage diversity in boreal forests at multiple spatial scales? *Ann. Zool. Fenn.* **2014**, *51*, 457–476. [CrossRef]
93. Vangansbeke, P.; Blondeel, H.; Landuyt, D.; De Frenne, P.; Gorissen, L.; Verheyen, K. Spatially combining wood production and recreation with biodiversity conservation. *Biodiv. Conserv.* **2017**, *26*, 3213–3239. [CrossRef]
94. Fleishman, E.; Murphy, D.D.; Brussard, P.F. A new method for selection of umbrella species for conservation planning. *Ecol. Appl.* **2000**, *10*, 569–579. [CrossRef]
95. Suter, W.; Graf, R.F.; Hess, R. Capercaillie (Tetrao urogallus) and Avian Biodiversity: Testing the Umbrella-Species Concept. *Cons. Bio.* **2002**, *16*, 3. [CrossRef]
96. Roberge, J.-M.; Angelstam, P. Usefulness of the umbrella species concept as a conservation tool. *Cons. Bio.* **2004**, *18*, 76–85. [CrossRef]
97. Mikoláš, M.; Svitok, M.; Tejkal, M.; Leitão, P.J.; Morrissey, R.C.; Svoboda, M.; Fontaine, J.B. Evaluating forest management intensity on an umbrella species: Capercaillie persistence in central Europe. *For. Ecol. Manage.* **2015**, *354*, 26–34. [CrossRef]
98. Niemi, G.J.; Hanowski, J.M.; Lima, A.R.; Nicholls, T.; Weiland, N. A critical analysis on the use of indicator species in management. *J. Wildl. Manage.* **1997**, *61*, 1240–1252. [CrossRef]
99. Prendergast, J.R.; Eversham, B.C. Species richness covariance in higher taxa: empirical tests of the biodiversity indicator concept. *Ecography* **1997**, *20*, 210–216. [CrossRef]
100. Landres, P.B.; Verner, J.; Thomas, J.W. Ecological uses of vertebrate indicator species: A critique. *Cons. Bio.* **1988**, *2*, 316–328. [CrossRef]

101. Lindenmayer, D.B. Future directions for biodiversity conservation in managed forests: Indicator species, impact studies and monitoring programs. *For. Ecol. Manage.* **1999**, *115*, 277–287. [CrossRef]
102. Sabatini, F.M.; Burrascano, S.; Azzella, M.M.; Barbati, A.; De Paulis, S.; Di Santo, D.; Facionia, L.; Giuliarelli, D.; Lombardi, F.; Maggi, O.; et al. One taxon does not fit all: Herb-layer diversity and stand structural complexity are weak predictors of biodiversity in Fagus sylvatica forests. *Ecol. Ind.* **2016**, *69*, 126–137. [CrossRef]
103. Larrieu, L.; Gosselin, F.; Archaux, F.; Chevalier, R.; Corriol, G.; Dauffy-Richard, E.; Deconchat, M.; Gosselin, M.; Ladet, S.; Savoie, J.-M.; et al. Cost-efficiency of cross-taxon surrogates in temperate forests. *Ecol. Ind.* **2018**, *87*, 56–65. [CrossRef]
104. Jones, K.B.; Neale, A.C.; Nash, M.S. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. *Lands. Ecol.* **2001**, *16*, 301–312. [CrossRef]
105. Li, H.; Wu, J. Use and misuse of landscape metrics. *Lands. Ecol.* **2004**, *19*, 389–399. [CrossRef]
106. Motz, K.; Sterba, H.; Pommerening, A. Sampling measures of tree diversity. *For. Ecol. Manage.* **2010**, *260*, 1985–1996. [CrossRef]
107. Karahalil, U.; Başkent, E.Z.; Sivrikaya, F.; Kılıç, B. Analyzing deadwood volume of Calabrian pine (Pinus brutia Ten.) in relation to stand and site parameters: a case study in Köprülü Canyon National Park. *Environ. Monit. Assess.* **2017**, *189*, 112. [CrossRef]
108. Warren, W.G.; Olsen, P.F. A line intersect technique for assessing logging waste. *For. Sci.* **1964**, *10*, 267–276.
109. Polit, J.-I.; Brown, S. Mass and nutrient content of deadwood in a central Illinois floodplain forest. *Wetlands* **1996**, *16*, 488–494. [CrossRef]
110. Luque, S.; Riutta, T.; Joensuu, J.; Rautjarvi, N.; Tomppo, E. Multi-source forest inventory data for biodiversity monitoring and planning at the forest landscape level. In *Monitoring and Indicators of Forest Biodiversity in Europe—From Ideas to Operationality*; Marchetti, M., Ed.; EFI, European Forest Institute: Joensuu, Finland, 2004; proceedings no. 51; pp. 430–444.
111. Luque, S.; Riutta, T.; Joensuu, J.; Tomppo, E. Spatial analysis and remote sensing for biodiversity monitoring and planning at the forest landscape level. In *Actes Hermes-science CASSINI'04, 7eme Conference GDR SIGMA*; Geomatique et Analyse Spatiale: Grenoble, France, 2004; pp. 149–155.
112. Roberge, J.-M.; Lämås, T.; Lundmark, T.; Ranius, T.; Felton, A.; Nordin, A. Relative contributions of set-asides and tree retention to the long-term availability of key forest biodiversity structures at the landscape scale. *J. Environ. Manage.* **2015**, *154*, 284–292. [CrossRef] [PubMed]
113. Bottalico, F.; Chirici, G.; Giannini, R.; Mele, S.; Mura, M.; Puxeddu, M.; Travaglini, D. Modeling Mediterranean forest structure using airborne laser scanning data. *Internat. J. App. Earth Observ. Geoinfo.* **2017**, *57*, 145–153. [CrossRef]
114. Treinys, R.; Mozgeris, G.; Skuja, S. Can intensified forestry be responsible for changes in habitat usage by the forest-dwelling Black Stork? *Eur. J. Forest Res.* **2016**, *135*, 1175–1186. [CrossRef]
115. Suchant, R.; Braunisch, V. Multidimensional habitat modelling in forest management—A case study using capercaillie in the Black Forest, Germany. *Ecol. Bull.* **2004**, *51*, 455–469.
116. Lõhmus, A.; Nellis, R.; Pullerits, M.; Leivits, M. The Potential for Long-Term Sustainability in Seminatural Forestry: A Broad Perspective Based on Woodpecker Populations. *Environ. Manage.* **2016**, *57*, 558–571. [CrossRef]
117. Lindberg, E.; Roberge, J.-M.; Johansson, T.; Hjältén, J. Can Airborne Laser Scanning (ALS) and Forest Estimates Derived from Satellite Images Be Used to Predict Abundance and Species Richness of Birds and Beetles in Boreal Forest? *Remote Sens.* **2015**, *7*, 4233–4252. [CrossRef]
118. Mura, M.; McRoberts, R.E.; Chirici, G.; Marchetti, M. Estimating and mapping forest structural diversity using airborne laser scanning data. *Remote Sens. Environ.* **2015**, *170*, 133–142. [CrossRef]
119. Tracz, W.; Ciurzycki, W.; Zaniewski, P.; Kwaśny, Ł.; Marciszewska, K.; Mozgawa, J. Identification of Zones with High Potential for Biological Diversity on Dormant Forested Landslides. *Eur. J. Forest Res.* **2019**, *138*, 363–373. [CrossRef]
120. Lefsky, M.A.; Cohen, W.B.; Parker, G.G.; Harding, D.J. Lidar remote sensing for ecosytem studies. *BioScience* **2002**, *52*, 19–30. [CrossRef]
121. MacKerron, G.; Mourota, S. Happiness is greater in natural environments. *Glob. Environ. Change* **2013**, *23*, 992–1000. [CrossRef]

122. Pereira, H.M.; Cooper, H.D. Towards the global monitoring of biodiversity change. *Tren. Ecol. Evol.* **2006**, *21*, 123–129. [CrossRef] [PubMed]
123. Jones, J.P.; Collen, B.E.N.; Atkinson, G.; Baxter, P.W.; Bubb, P.; Illian, J.B.; Nicholson, E. The Why, What, and How of Global Biodiversity Indicators Beyond the 2010 Target. *Cons. Bio.* **2011**, *25*, 450–457. [CrossRef] [PubMed]
124. Feest, A.; Aldred, T.D.; Jedamzik, K. Biodiversity Quality: A Paradigm for Biodiversity. *Ecol. Ind.* **2010**, *10*, 1077–1082. [CrossRef]
125. Hooper, D.U.; Chapin, F.S., III; Ewel, J.J.; Hector, A.; Inchausti, P.; Lavorel, S.; Lawton, J.H.; Lodge, D.M.; Loreau, M.; Naeem, S.; et al. Effects of biodiversity on ecosystem functioning: A consensus of current knowledge. *Ecol. Monographs* **2005**, *75*, 3–35. [CrossRef]
126. Bobbink, R.; Hornung, M.; Roelofs, J.G.M. The effects of air-borne nitrogen pollutants on species diversity in natural and semi-natural European vegetation. *J. Ecol.* **1998**, *86*, 717–738. [CrossRef]
127. Feest, A. *Research to Test the Integration of the Nitrogen Critical Load Exceedence Model (EG4) into the 2010 Target by Linking it to the Butterfly Indicator (EG1)*; European Environment Agency: Copenhagen, Denmark, 2008; p. 34.
128. Maskell, L.C.; Smart, S.M.; Bullock, J.M.; Thompson, K.; Stevens, C.J. Nitrogen deposition causes widespread loss of species richness in British Habitats. *Glob. Change Biol.* **2010**, *16*, 671–679. [CrossRef]
129. Stevens, C.J.; Dise, N.B.; Mountford, J.O.; Gowing, D.J. Impact of nitrogen deposition on the species richness of grasslands. *Science* **2004**, *303*, 1876–1879. [CrossRef]
130. Dirnböck, T.; Grandin, U.; Bernhardt-Römermann, M.; Beudert, B.; Canullo, R.; Forsius, M.; Grabner, M.-T.; Holmberg, M.; Kleemola, S.; Lundin, L.; et al. Forest floor vegetation response to nitrogen deposition in Europe. *Glob. Change Biol.* **2014**, *20*, 429–440. [CrossRef]
131. Miina, J.; Hallikainen, V.; Härkönen, K.; Merilä, P.; Packalen, T.; Rautio, P.; Salemaa, M.; Tonteri, T.; Tolvanen, A. Incorporating a Model for Ground Lichens into Multi-Functional Forest Planning for Boreal Forests in Finland. *For. Ecol. Manage.* **2020**, *460*, 117912. [CrossRef]
132. Groom, G.; Mücher, C.A.; Ihse, M.; Wrbka, T. Remote sensing in landscape ecology: experiences and perspectives in a European context. *Lands. Ecol.* **2006**, *21*, 391–408. [CrossRef]
133. Schindler, S.; Poirazidis, K.; Wrbka, T. Towards a core set of landscape metrics for biodiversity assessments: A case study from Dadia National Park, Greece. *Ecol. Ind.* **2008**, *8*, 502–514. [CrossRef]
134. Simila, M.; Kouki, J.; Monkkonen, M.; Sippola, A.L.; Huhta, E. Co-variation and indicators of species diversity: Can richness of forest-dwelling species be predicted in northern boreal forests? *Ecol. Ind.* **2006**, *6*, 686–700. [CrossRef]
135. Nordström, E.-M.; Nieuwenhuis, M.; Başkent, E.Z.; Biber, P.; Black, K.; Borges, J.G.; Bugalho, M.N.; Corradini, G.; Felton, A. Forest decision support systems for analysis of ecosystem services provisioning at landscape scale under global climate and market change scenarios. *Eur. J. Forest Res.* **2019**, *138*, 561–581. [CrossRef]

Appendices

1. Search strategy

Table S1: List of papers included in the review/Lists of papers excluded from the review

Page nb	**Google Scholar** Keywords: forest AND management AND planning AND biodiversity AND indicators Custom range 1990-2020 About 50,700 results Selected: 59 papers	**Google Scholar** Keywords: "forest management planning" AND biodiversity AND indicators Custom range 1990-2020 4 220 results Selected: 29	**Google Scholar** Keywords: “forest management planning” AND “biodiversity indicators” Custom range 1990-2020 318 results Selected: 27	**Google Scholar** Keywords: forest AND management AND planning AND biodiversity AND indicators Custom range 2019-2020 About 16,700 results Selected: 23	**Web of Science** Keywords: forest AND management AND planning AND biodiversity AND indicators Time-span: 1990-2020 Refined for research articles and reviews 303 results Selected: 41	**Web of Science** Keywords: "forest management planning" AND biodiversity AND indicators Time-span: 1990-2020 Refined for research articles and reviews 23 results Selected: 8	**Web of Science** Keywords: "forest management planning" AND “biodiversity indicators” Time-span: 1990-2020 Refined for research articles and reviews 3 results Selected: 1
1.	Butchard et al. 2010; Noss 1999; Lindenmayer et al. 2000; Noss 1990; Franklin 1993; Failing & Gregory 2003; Feris & Humphrey 1999; Lindenmayer et al. 2006	Franklin 1993; Angelstam & Petterson 1997	Franklin 1997; Humphrey et al. 1999; Monkkonen et al. 2014; Angelstam et al. 2004a,b; Angelstam & Dönz-Breuss 2004	Versluijs et al. 2019; Augustynczik et al. 2019a, 2020;		Jayathunga et al. 2020; Vangansbeke et al. 2017; Karahalil et al. 2017; Trivino et al. 2017	Trivino et al. 2017
2.	Hagan & Whitman 2006; Lindenmayer 1999; Angelstam & Petterson 1997	Edenius & Mikusiński 2006; Romero-Calcerada & Luque 2006	Gao et al. 2014; Makela et al. 2012; Romero-Calcerada & Luque 2006	Barsoum et al. 2019; Lõhmus & Lõhmus 2019; Evans et al. 2019		Standovar et al. 2016; Baskent et al. 2008; Edenius et al. 2006;	
3.	Simberloff 1999	Gao et al. 2014; Monnkonen et al. 2014; Angelstam et al. 2004a	Trivino et al. 2016; Laarmaan et al. 2009; Lexer et al. 2000; Angelstam et al. 2004b	Brown et al. 2020; Parisi et al. 2020;	Tracz et al. 2019; Augustynczik et al. 2019a; Bertini et al. 2019; Lelli et al 2019	Baskent et al. 1996	

4.	Smith et al. 2008; Barbati et al. 2014; Brockerhoff 2008	Suter et al. 2002; Humphrey etal.1999; Siitonen et al. 2001	Anselme et al. 2010; Sippola et al. 2014	Augustynczik et al. 2019b; Pohjanmies et al. 2019;	Blattert et al. 2018		
5.	Maleque et al. 2009	Angelstam et al. 2004b; Makela et al. 2012; Ferris et al. 2000		Reise et al. 2019; Muurinen et al. 2019; Selkimäki et al. 2019; Miina et al. 2020; Mölder et al. 2019	Ozdemir et al. 2018; Naumov et al. 2018;		
6	Jonsonn & Jonsel 1999; Uuemaa et al. 2013; Stephens et al. 2007	Fridman & Walheim 2000; Pukkala et al. 1997; Lexer et al. 2000; Baskent et al. 2005	Redon et al. 2014; Ezquerro et al. 2016	Bujoczek et al. 2020	Angelstam et al. 2018;		
7	Romero-Calcerada & Luque 2006; Heink & Kowarik 2010; Angelstam et al. 2004a		La fond et al. 2015		Korkmaz et al. 2018; Vangansbeke et al. 2017;		
8	Linnell et al. 2000; Fridman & Walheim 2000		Mart´ın-Queller et al. 2011; Motz et al. 2010	Lelli et al. 2019;			
9	Tischendorf & Farig 2003; Baskent & Jordan 1996; Mortberg et al. 2007		Pach & Podlaski 2015	Lecina-Diaz et al. 2019	Karahalil et al. 2017; Trivino et al. 2017		
10	McElhinny et al. 2005; Angelstam et al. 2003; Fabbio et al. 2003; Mikusinski et al. 2001	Angelstam et al. 2004b; Laarman et al. 2009			Santos et al. 2016; Standovar et al. 2016		
11	Leitao & Ahern 2002; Puumalainen et al. 2002; Poiani et al. 2000; Lindermayer & Likens 2009; Muler & Brandl 2009; Ranio & Niemela 2003; Rodrigues & Books 2007	Trivino et al. 2016					

12	Corona et al. 2011; Frank et al. 2012; Schnidler et al. 2013; Simberloff 1999;		Vangansbeke et al. 2017; Angelstam et al. 2018	Broome et al. 2019	Loehmus et al. 2016		
13	Roberts & Gilliam 1995; Nilsson et al. 2001; Roberge et al. 2008a; Roberge et al 2008b;	Anselme et al. 2010	Naumov et al. 2018		Kovac et al. 2016; Mura et al. 2015; Lafond et al. 2015;		
14	Dale & Byeler 2001						
15	Roberge & Angelstam 2006; Thompson 2006;		Badalamenti et al. 2017				
16	Mönnkönnen et al 2014		Bazile et al. 2016		Redon et al. 2014;		
17	Suter et al. 2002		Treiynis et al. 2016	Hanish et al. 2019			
18	Winter et al. 2012; Humphrey et al. 1999						
19					Rudolf et al. 2012 Lundstrom 2011; Lohmus et al. 2012		
20	Angelstam 1997			Magg et al. 2019			
21				Dantas de Paula 2019			
22		Suchant & Braunisch 2004			Rubio et al. 2011		
23	Nascimbene et al. 2009	Mönkönnen 1999; Redon et al. 2014			Gil- tena et al. 2010; Alday et al. 2010;		
24					Maleque et al. 2009;		
25	Kangas et al. 2015	Lafond et al. 2015	Pesonen et al. 2010		Mullen et al. 2008; Roberge et al. 2008a; Baskent 2008;		
26		Masom & Zapponi 2015			Cullota et al. 2007; Edenius & Mikusiński 2006; Ericsson 2006;		
27	Siitonen 2001			Velázquez et al 2019	Oxborough et al. 2006; Gittings et al. 2006		
28	Fleishman et al. 2006	Mikolas et al. 2015; Mura et al. 2015			Maleque et al. 2006; Rempel et al. 2004;		
29		Roberge et al. 2015					

30	Schindler et al. 2008						
31					de Warnaffe & Devillez 2002		
32					Ferris et al. 2000; Noss 1999		
33					Baskent & Jordan 1996; Hanley 1996		

2. List of papers included in the review:

1. Karahalil et al. 2017
2. Rubio et al. 2011
3. Jonsson & Jonsell 1999
4. Torras & Saura 2008
5. Roberge & Angelstam 2004
6. Smith et al. 2008
7. Schindler et al. 2008
8. Schindler et al. 2013
9. Romero-Calcerrada & Luque 2006
10. Roberge et al. 2008a
11. Roberge et al. 2008b
12. Müller & Brand 2009
13. Mikusiński et al. 2001
14. Angelstam & Dönz-Breuss 2004
15. Angelstam et al. 2018
16. Anselme et al. 2010
17. Badalamenti et al. 2017
18. Basile et al. 2016
19. Gao et al. 2014
20. Humphrey et al. 1999
21. Laarmann et al. 2009
22. Lafond et al. 2015
23. Mart´ın-Queller et al. 2011
24. Motz et al. 2010
25. Naumov et al. 2018
26. Pach & Podlaski 2015
27. Pesonen et al. 2010
28. Redon et al. 2014
29. Sippola et al. 2014
30. Treinys et al. 2016
31. Trivino et al. 2016
32. Vangansbeke et al. 2017
33. Ferris et al. 2000
34. Fridman & Walheim 2000
35. Kangas et al. 2015
36. Mikoláš et al. 2015
37. Mura et al. 2015
38. Roberge et al. 2015

39. Suchant & Braunisch 2004
40. Suter et al. 2002
41. Gittings et al. 2006
42. Rudolf et al. 2012
43. Lõhmus et al. 2016
44. Mullen et al. 2008
45. Oxbrough et al. 2006
46. Ozdemir et al. 2018
47. Santos et al. 2016
48. Standovár et al. 2016
49. Morelli 2015
50. Thingstad et al. 2018
51. Montané et al. 2016
52. Stachura-Skierczynska & Kosinski 2016
53. Bottalico et al. 2017
54. Keren & Diaci 2018
55. Kosewska et al. 2018
56. Ranius et al. 2016
57. Keren et al. 2017
58. Asbeck et al. 2019
59. Parisi et al. 2019
60. Güthlin et al. 2014
61. Czeszczewik et al. 2015
62. Kaufmann et al. 2017
63. Lindberg et al. 2015
64. Coote et al. 2013
65. Morelli et al. 2013
66. Thers et al. 2017
67. Straw et al. 2017
68. Renner et al. 2018
69. Purahong et al. 2014
70. Pakkala et al. 2015
71. Paffetti et al. 2012
72. Cullota et al. 2015
73. Kovač et al. 2016
74. Ambrosio et al. 2018
75. Durak & Durak 2016
76. Mikulova et al. 2019
77. Wei et al. 2020
78. Lecina-Diaz 2019
79. Parisi et al. 2020
80. Lešo et al. 2019
81. Miina et al. 2020
82. Augustznczik et al. 2019a
83. Barsoum et al. 2019
84. Broome et al. 2019
85. Kermavnar 2019
86. Lohmus & Lohmus 2019
87. Lelli et al. 2019
88. Magg et al. 2019
89. Velasquez et al. 2019

90. Muurinen et al. 2019
92. Bujoczek et al. 2020
93. Bertini et al. 2019
94. Tratcz et al. 2019

3. Reviews and synthesis removed from the list:

1. Noss 1999
2. Lindenmayer et al. 2000
3. Noss 1999
4. Noss 1990
5. Failing & Gregory 2003
6. Feris & Humphrey 1999
7. Lindenmayer et al. 2006
8. Hagan & Whitman 2006
9. Lindenmayer 1999
10. Angelstam & Petterson 1997
11. Simberloff 1999;
12. Brockerhoff 2008
13. Maleque et al. 2009
14. Uuemaa et al. 2013
15. Stephens et al. 2007
16. Heink & Kowarik 2010
17. Angelstam et al. 2004a
18. Linnell et al. 2000
19. Tischendorf & Farig 2003
20. McElhinny et al. 2005
21. Angelstam et al. 2003
22. Fabbio et al. 2003
23. Leitao & Ahern 2002
24. Poiani et al. 2000
25. Lindermayer & Likens 2009
26. Ranio & Niemela 2003
27. Rodrigues & Books 2007
28. Corona et al. 2011
29. Simberloff 1999
30. Roberts & Gilliam 1995
31. Nilsson et al. 2001
32. Dale & Byeler 2001
33. Thompson 2006
34. Winter et al. 2012
35. Angelstam 1997
36. Siitonen 2001
37. Fleishman et al. 2006
38. Edenius & Mikusiński 2006
39. Makela et al. 2012
40. Ezquerro et al. 2016
41. Angelstam et al. 2004b
42. Pukkala et al. 1997
43. Baskent et al. 2005

44. Baskent et al. 1996
45. Mönkönnen 1999
46. Masom & Zapponi 2015
47. Baskent & Jordan 1996
48. Korkmaz et al. 2018
49. Rempel et al. 2004
50. Eriksson et al. 2006
51. Molder et al. 2019

4. Research papers removed with reasons:

1. Butchard et al. 2010 (we couldn’t extract forest biodiversity indicators)
2. Franklin 1993 (not from Europe)
3. Barbati et al. 2014 (we couldn’t extract forest biodiversity indicators, and the context is focused on policy decisions on biodiversity)
4. Mörtberg et al. 2006 (not forest ecosystems)
5. Frank et al. 2012 (could not extract biodiversity indicators and ecosystems not defined)
6. Mönnkönnen et al. 2014 (the focus is not biodiversity assessment, but the competitiveness of timber production and biodiversity)
7. Nascimbene et al. 2009 (the scale is a tree)
8. Alday et al. 2010 (we could not identify how biodiversity was calculated)
9. De Warnaffe & Devillez 2002 (not written in English but French)
10. Gil-tena et al. 2010 (the focus is not biodiversity assessment)
11. Lundstrom et al. 2011 (the scale is the region)
12. Jayathunga et al. 2020 (couldn’t access)
13. Hanish et al. 2019 (not from Europe)
14. Dantas de Paula 2019 (the scale is global)
15. Evans et al. 2019 (the focus is not biodiversity assessment, but tree dieback)
16. Augustznczik et al. 2020 (could not extract biodiversity indicators)
17. Lohmus et al. 2012 (could not access the paper)
18. Augustznczik et al. 2019b (could not extract biodiversity indicators)
19. Brown et al. 2019 (not from Europe)
20. Pohjanmies et al. 2019 (focus is not biodiversity assessment)
21. Reise et al. 2019 (the scale is national)
22. Selkimäki et al. 2019 (could not extract the indicators)
23. Versluijs et al. 2019 (the scale is biome)

Table S2: Biodiversity attributes and indicators used in the literature sorted by type of biodiversity (structure, composition, function)

Biodiversity type	Attribute	Explanation	Biodiversity indicator	Author

Structure	Deadwood (DW)	Deceased laying or standing trees, branches, pieces of wood, wood stamps...	DW volume; coarse woody debris (pieces of DW); standing DW; laying DW; DW logs...	Karahalil et al. 2017; Jonsson & Jonsell 1999; Torras & Saura 2008; Smith et al. 2008; Roberge et al. 2008a; Angelstam & Dönz-Breuss 2004; Badalamenti et al. 2017; Laarmann et al.2009 ; Lafond et al. 2015; Pesonen et al. 2010; Redon et al. 2014; Sippola et al. 2004; Trivino et al. 2016; Ferris et al. 2000; Fridman & Walheim 2000; Roberge et al. 2015; Mullen et al. 2008 ;Standovár et al. 2016; Keren & Diaci 2018; Coote et al. 2013; Parisi et al. 2019; Kaufmann et al. 2017; Bujoczek et al. 2020; Muurinen et al.2019; Augustynczik et al. 2019; Barsoum et al. 2019; Lohmus &Lohmus 2019; Parisi et al. 2020; Lešo et al. 2019; Kovač et al. 2016
			Decay stage	Bujoczek et al. 2020
	Naturalness	The level of human influence in the forest	Uneven-age stands	Stachura-Skierczynska & Kosinski 2016; Roberge et al. 2008b
			Large trees	Angelstam & Dönz-Breuss 2004; Badalamenti et al. 2017; Lafond et al. 2015; Kangas et al. 2015; Roberge et al. 2015; Standovár et al. 2016; Roberge et al. 2008b; Stachura-Skierczynska & Kosinski 2016; Lohmus &Lohmus 2019; Lešo et al. 2019; Wei et al. 2020
			Mature trees/forests	Torras & Saura 2008; Redon et al. 2014; Gittings et al. 2006; Angelstam & Dönz-Breuss 2004; Standovár et al. 2016; Badalamenti et al. 2017; Stachura-Skierczynska & Kosinski 2016;
	Forest inventory variables	-	Volume	Paffetti et al. 2012; Bottalico et al. 2017; Kovač et al. 2016
			Diameter heterogeneity	Pach & Podlaski 2015; Redon et al. 2014; Keren et al. 2017; Bottalico et al. 2017; Kaufmann et al. 2017; Bertini et al. 2019; Wei et al. 2020
			Canopy cover	Smith et al. 2008; Gao et al. 2014; Coote et al. 2013; Bujoczek et al. 2020; Kermavnar et al. 2019; Parisi et al. 2020; Lešo et al. 2019
			Tree height	Paffetti et al. 2012; Bottalico et al. 2017; Bertini et al. 2019; Parisi et al. 2020
			Basal area	Paffetti et al. 2012; Bottalico et al. 2017; Bertini et al. 2019; Augustynczik et al. 2019; Parisi et al. 2020; Miina et al. 2020; Wei et al. 2020; Kovač et al. 2016
			Biomass	Bottalico et al. 2017; Bertini et al. 2019
			Stand age	Coote et al. 2013; Lešo et al. 2019; Miina et al. 2020
			Tree density	Bertini et al. 2019; Barsoum et al. 2019;
	Vertical structure	Stratification, layers of	Vegetation in three layers	Smith et al. 2008; Gao et al. 2014; Humphrey et al. 1999; Mura et al. 2015; Kaufmann et al. 2017; Muurinen et

		understory and overstory vegetation		al.2019; Kermavnar et al. 2019; Lešo et al. 2019; Mikulova et al. 2019
			Shrub layer	Coote et al. 2013
			Litter cover	Smith et al. 2008; Lešo et al. 2019; Mikulova et al. 2019
	Horizontal structure	The density of vegetation cover		Mura et al. 2015
			Open space	Mullen et al. 2008
			Spacing	Smith et al. 2008; Mullen et al. 2008
			Proximity of old woodlands	Coote et al. 2013
	Microhabitats	Habitat small in size (e.g. a tree, small water pool)		Parisi et al. 2020; Smith et al. 2008
			Tree-related micro-habitats	Roberge et al. 2008b; Ozdemir et al. 2018; Standovár et al. 2016; Asbeck et al. 2019; Augustynczik et al. 2019
	Fragmentation	The level of fragmentation and the connectivity of habitats		Schindler et al. 2018; Angelstam et al. 2018; Basile et al. 2016; Schindler et al. 2008
			Patch diversity, size, shape and aggregation, the core area	Kovač et al. 2016; Schindler et al. 2008; Basile et al. 2016; Schindler et al. 2018
			Edge density, contrast	Basile et al. 2016; Schindler et al. 2018
	Topographic elements	Physical properties of the area	Slope	Tratcz et al. 2019; Augustynczik et al. 2019; Lešo et al. 2019; Mikulova et al. 2019
			Elevation	Augustynczik et al. 2019; Lešo et al. 2019
Composition	Species richness	The number of species per number of individuals or biomass		Schindler et al.2013; Lelli et al. 2019
	Plant species diversity	The number of species per e.g. stand or landscape	Plants	Gao et al. 2014; Durak & Durak 2016
			Woody plants	Schindler et al. 2013; Kermavnar et al. 2019; Lecina-Diaz et al. 2019
			Shrubs	Torras & Saura 2008; Martín-Queller et al. 2011; Gittings et al. 2006; Wei et al. 2020
			Trees	Torras & Saura 2008; Badalamenti et al. 2017; Gao et al. 2014; Martín-Queller et al. 2011; Motz et al. 2010; Redon et al. 2014; Sippola et al. 2004; Ferris et al. 2000; Kangas et al. 2015; Barsoum et al. 2019; Kermavnar et al. 2019; Lohmus &Lohmus 2019; Parisi et al. 2020; Kovač et al. 2016
			Ground vegetation	Rubio et al. 2011; Humphrey et al. 1999; Lafond et al. 2015; Gittings et al. 2006; Mullen et al. 2008
			Native species	Roberge et al. 2008b; Mikulova et al. 2019

			Invasive tree species	Standovár et al. 2016
	Endangered species	species threatened with declining or extinction		Anselme et al. 2009; Ranius et al. 2016
	Valuable flora and fauna	Animals and plants of particular ecological value		Kangas et al. 2015; Lelli et al. 2019
	Individual species	-	Orchids	Schindler et al.2013;
			Spiders	Smith et al. 2008; Mullen et al. 2008; Kosewska et al. 2018; Coote et al. 2013; Barsoum et al. 2019
			Hoverflies	Smith et al. 2008; Humphrey et al. 1999; Gittings et al. 2006; Straw et al. 2017
			Butterflies	Magg et al. 2019
			Saprophylic beetles	Ranius et al. 2016; Parisi et al. 2019; Magg et al. 2019; Parisi et al. 2020
			Carabid beetles	Kosewska et al. 2018; Barsoum et al. 2019; Parisi et al. 2020
			Ground-living beetles	Lindberg et al. 2015
			Meadow vipers	Anselme et al. 2009
			Capercaillies	Trivino et al. 2016;Mikoláš et al. 2015; Suchant & Braunisch 2004, Suter et al. 2002
			Woodpeckers	Romero-Calcerrada & Luque 2006; Roberge et al.2008a; Mikusiński et al. 2001, Basile et al. 2016; Trivino et al. 2016 ; Lõhmus et al. 2016
			Black storks	Treinys et al. 2016
			Hazel grouses	Trivino et al. 2016
			Long-tailed tits	Trivino et al. 2016
			Flying squirrels	Trivino et al. 2016
			Red foxes	Güthlin et al. 2014
	Taxa	A taxonomic groupof any rank, suchas a species, family, or class.	Birds	Czeszczewik et al. 2015; Morelli 2015; Lindberg et al. 2015; Coote et al. 2013; Morelli et al. 2013; Renner et al. 2018; Bujoczek et al. 2020; Velasquez et al. 2019; Magg et al. 2019; Lelli et al. 2019; Augustynczik et al. 2019; Broome et al. 2019; Lecina-Diaz et al. 2019; Lešo et al. 2019
			Bryophyte	Jonsson & Jonsell 1999; Smith et al. 2008; Humphrey et al. 1999; Kaufmann et al. 2017; Coote et al. 2013; Barsoum et al. 2019; Broome et al. 2019
			Coleoptera	Jonsson & Jonsell 1999; Müller & Brand 2009; Humphrey et al. 1999; Mullen et al. 2008
			Flying arthropods	Barsoum et al. 2019
			Fungi	Jonsson & Jonsell 1999; Angelstam & Dönz-Breuss 2004; Sippola et al. 2004; Ferris et al. 2000; Kinga et al. 2012; Thers et al. 2017; Lelli et al. 2019; Broome et al.

				2019; Ambrosio et al. 2018
			Moss	Mikulova et al. 2019
			Lichens	Jonsson & Jonsell 1999; Angelstam & Dönz-Breuss 2004; Kaufmann et al. 2017; Lelli et al. 2019; Broome et al. 2019; Lohmus &Lohmus 2019; Miina et al. 2020
			Liverworts	Broome et al. 2019
			Resident birds	Roberge & Angelstam 2006
			Small terrestrial birds	Schindler et al.2013
			Vascular plants	Smith et al. 2008; Humphrey et al. 1999; Kaufmann et al. 2017; Coote et al. 2013; Lelli et al. 2019; Barsoum et al. 2019; Broome et al. 2019; Mikulova et al. 2019
			Orthopterans	Schindler et al.2013;
			Amphibians	Schindler et al.2013; Velasquez et al. 2019; Broome et al. 2019
			Bats	Renner et al. 2018
			Reptiles	Schindler et al.2013; Vangansbeke et al. 2017; Velasquez et al. 2019; Magg et al. 2019; Broome et al. 2019
			Mammals	Velasquez et al. 2019; Magg et al. 2019; Broome et al. 2019
Function	Disturbance	Natural or human-induced changes in the ecosystem	The proportion of plotswith uprooting	Roberge et al. 2008b; Angelstam & Dönz-Breuss 2004
			The proportion of plotswith periodic flooding	Roberge et al. 2008b
			Uprooted trees	Bujoczek et al. 2020
			Rockfall	Standovár et al. 2016
	Mortality	Dying of forest trees	Individual tree mortality	Laarmann et al.2009
	Tree sp. regeneration		Saplings and shoots	Bujoczek et al. 2020; Cullota et al. 2015
			Type of regeneration	Kovač et al. 2016; Lohmus & Lohmus 2019; Standovár et al. 2016
	Nutrient cycling	The movement of nutrients in the environment	Leaf litter nutrient cycling	Purahong et al. 2014
		Soil properties	Available Phosphorus (P)	Smith et al. 2008
			Nitrogen	Durak & Durak 2016
			Nutrients	Kermavnar et al. 2019; Mikulova et al. 2019
			The thickness of organic layer	Kermavnar et al. 2019;
			Ph	Kermavnar et al. 2019; Durak & Durak 2016; Mikulova et al. 2019

References

Jonsson, B.G.; Jonsell, M. Exploring potential biodiversity indicators in boreal forests. Biodiv. Cons. 1999, 8, 1417–1433. doi:10.1023/A:1008900309571

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

Anselme, B.; Bousquet, F.; Lyet, A.; Etienne, M.; Fady, B.; Le Page, C. Modelling of spatial dynamics and biodiversity conservation on Lure mountain (France). Environ. Modell. Software 2010,25, 1385–1398. doi:10.1016/j.envsoft.2009.09.001

Badalamenti, E.; La mantia, T.L.; Mantia, G.; La, Cairone, A.; La Mela Veca, D.S. Living and dead aboveground biomass in mediterranean forests: Evidence of old-growth traits in a quercus pubescenswilld. s.l. stand. Forests 2017, 8, 187. doi:10.3390/f8060187

Laarmann, D.; Korjus, H.; Sims, A.; Stanturf, J.A.; Kiviste, A.; Köster, K. Analysis of forest naturalness and tree mortality patterns in Estonia. For. Ecol. Manage. 2019, 258, S187–S195. doi:10.1016/j.foreco.2009.07.014

Lafond, V.; Cordonnier, T.; Courbaud, B. Reconciling Biodiversity Conservation and Timber Production in Mixed Uneven-Aged Mountain Forests: Identification of Ecological Intensification Pathways. Environ. Manage. 2015, 56, 1118–1133. doi:10.1007/s00267-015-0557-2

Pach, M.; Podlaski, R. Tree diameter structural diversity in Central European forests with Abies alba and Fagus sylvatica: managed versus unmanaged forest stands. Ecol. Research, 2015, 30, 367–384. doi:10.1007/s11284-014-1232-4

Redon, M.; Luque, S.; Gosselin, F.; Cordonnier, T. Is generalisation of uneven-aged management in mountain forests the key to improve biodiversity conservation within forest landscape mosaics? Annals For. Sci. 2014, 71, 751–760. doi:10.1007/s13595-014-0371-7

Sippola, A.L.; Similä, M.; Mönkkönen, M.; Jokimäki, J. Diversity of polyporous fungi (Polyporaceae) in northern boreal forests: Effects of forest site type and logging intensity. Scand. J. For. Res. 2004, 19, 152–163. doi:10.1080/02827580410026294

Triviño, M.; Pohjanmies, T.; Mazziotta, A.; Juutinen, A.; Podkopaev, D.; Le Tortorec, E.; Mönkkönen, M. Optimizing management to enhance multifunctionality in a boreal forest landscape. J.App. Ecol. 2017, 54, 61–70. doi:10.1111/1365-2664.12790

Rudolf, K.; Morschhauser, T.; Pál-Fám, F. Macrofungal diversity in disturbed vegetation types in North-East Hungary. Cent. Eur. J. Bio. 2012, 7, 634–647. doi:10.2478/s11535-012-0050-3

Oxbrough, A.G.; Gittings, T.; O'Halloran, J.; Giller, P.S.; Kelly, T.C. The influence of open space on ground-dwelling spider assemblages within plantation forests. For. Ecol. Manage. 2006, 237, 404–417. doi:10.1016/j.foreco.2006.09.063

Santos, M.; Ferreira, D.; Bastos, R.; Vicente, J.; Honrado, J.; Kueffer, C.; Cabral, J.A. Linking landscape futures with biodiversity conservation strategies in northwest Iberia—A simulation study combining surrogates with a spatio-temporal modelling approach. Ecol. Inform. 2016, 33, 85–100. doi:10.1016/j.ecoinf.2016.04.008

Montané, F.; Guixé, D.; Camprodon, J. Canopy cover and understory composition determine abudance of Vaccinium myrtillus L., a key plant for capercaillie (Tetrao urogallus), in subalpine forests in the Pyrenees. Plant Ecol. Divers. 2016, 9, 187–198. doi:10.1080/17550874.2016.1180562

Stachura-Skierczyńska, K.; Kosiński, Z. Do factors describing forest naturalness predict the occurrence and abundance of middle spotted woodpecker in different forest landscapes? Ecol. Ind.2016, 60, 832–844. doi:10.1016/j.ecolind.2015.08.020

Keren, S.; Diaci, J.; Motta, R.; Govedar, Z. Stand structural complexity of mixed old-growth and adjacent selection forests in the Dinaric Mountains of Bosnia and Herzegovina. For. Ecol. Manage. 2017, 400, 531–541. doi:10.1016/j.foreco.2017.06.009

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

Ranius, T.; Korosuo, A.; Roberge, J.M.; Juutinen, A.; Mönkkönen, M.; Schroeder, M. Cost-efficient strategies to preserve dead wood-dependent species in a managed forest landscape. Biol. Conserv. 2016, 204, 197–204. doi:10.1016/j.biocon.2016.10.017

Keren, S.; Diaci, J. Comparing the quantity and structure of deadwood in selection managed and old-growth forests in South-East Europe. Forests 2018, 9, 1–16.doi:10.3390/f9020076

Asbeck, T.; Pyttel, P.; Frey, J.; Bauhus, J. Predicting abundance and diversity of tree-related microhabitats in Central European montane forests from common forest attributes. For. Ecol. Manage. 2019, 432, 400–408. doi:10.1016/j.foreco.2018.09.043

Kaufmann, S.; Hauck, M.; Leuschner, C. Effects of natural forest dynamics on vascular plant, bryophyte, and lichen diversity in primeval Fagus sylvatica forests and comparison with production forests. J. Ecolo. 2018, 106, 2421–2434. doi:10.1111/1365-2745.12981

Straw, N.A.; Williams, D.T.; Fielding, N.J.; Jukes, M.; Connolly, T.; Forster, J. The influence of forest management on the abundance and diversity of hoverflies in commercial plantations of Sitka spruce: the importance of sampling in the canopy. For. Ecol. Manage. 2017, 406, 95–111. doi:10.1016/j.foreco.2017.06.010

Renner, S.C.; Suarez-Rubio, M.; Kaiser, S.; Nieschulze, J.; Kalko, E.K.V.; Tschapka, M.; Jung, K. Divergent response to forest structure of two mobile vertebrate groups. For.Eco. Manag. 2018, 415–416, 129–138. doi:10.1016/j.foreco.2018.02.028

Purahong, W.; Kaptrska, D.; Pecyna, M.J.; Schulz, E.; Schloter, M.; Buscot, F.; Kruğer, D. Influenceof different forest system management practices on leaf litter decomposition rates, nutrient dynamics and the activity of ligninolytic enzymes: A case study from Central European forests. PLoS ONE 2014, 9, 1–11. doi:10.1371/journal.pone.0093700

Paffetti, D.; Travaglini, D.; Buonamici, A.; Nocentini, S.; Vendramin, G.G.; Giannini, R.; Vettori, C. The influence of forest management on beech (Fagus sylvatica L.) stand structure and genetic diversity. For. Ecol. Manage. 2012, 284, 34–44. doi:10.1016/j.foreco.2012.07.026

Cullotta, S.; Puzzolo, V.; Fresta, A. The southernmost beech (Fagus sylvatica) forests of Europe (Mount Etna, Italy): ecology, structural stand-type diversity and management implications. Plant Biosys. 2015, 149, 88–99. doi:10.1080/11263504.2013.814603

Kovač, M.; Kutnar, L.; Hladnik, D. Assessing biodiversity and conservation status of the Natura 2000 forest habitat types: Tools for designated forestlands stewardship. For. Ecol. Manage. 2016, 359, 256–267. doi:10.1016/j.foreco.2015.10.011

Mikulová, K.; Jarolímek, I.; Bacigál, T.; Hegedüšová, K.; Májeková, J.; Medvecká, J.; Šibíková, M. The effect of non-native black pine (Pinus nigra J. F. Arnold) plantations on environmental conditions and undergrowth diversity. Forests 2019, 10, 548. doi:10.3390/f10070548

Durak, T.; Durak, R. How biotic differentiation of human impacted nutrient poor deciduous forests can affect the preservation status of mountain forest vegetation. Forests 2016, 7, 1–13. doi:10.3390/f7100241

Wei, L.; Archaux, F.; Hulin, F.; Bilger, I.; Gosselin, F. Stand attributes or soil micro-environment exert greater influence than management type on understory plant diversity in even-aged oak high forests. For. Ecol. Manage. 2020, 460, 117897. doi:10.1016/j.foreco.2020.117897

Lecina-Diaz, J.; Alvarez, A.; De Cáceres, M.; Herrando, S.; Vayreda, J.; Retana, J. Are protectedareas preserving ecosystem services and biodiversity? Insights from Mediterranean forests and shrublands. Lands. Ecol. 2019, 34, 2307–2321. doi:10.1007/s10980-019-00887-8

Augustynczik, A.L.D.; Asbeck, T.; Basile, M.; Bauhus, J.; Storch, I.; Mikusiński, G.; Hanewinkel, M. Diversification of Forest Management Regimes Secures Tree Microhabitats and Bird Abundance under Climate Change. Sci. Total Environ. 2019, 650, 2717–2730.

Broome, A.; Bellamy, C.; Rattey, A.; Ray, D.; Quine, C.P.; Park, K.J. Niches for Species, a Multi-Species Model to Guide Woodland Management: An Example Based on Scotland's Native Woodlands. Ecol. Ind. 2019, 103, 410–424. doi:10.1016/j.ecolind.2019.04.021.

Kermavnar, J.; Marinšek, A.; Eler, K.; Kutnar, L. Evaluating Short-Term Impacts of Forest Management and Microsite Conditions on Understory Vegetation in Temperate Fir-Beech Forests: Floristic, Ecological, and Trait-Based Perspective. Forests 2019, 10, 909.

Lõhmus, P.; Lõhmus. A. The Potential of Production Forests for Sustaining Lichen Diversity: A Perspective on Sustainable Forest Management. Forests 2019, 10, 1063.

Magg, N.; Ballenthien, E.; Braunisch, V. Faunal Surrogates for Forest Species Conservation: A Systematic Niche-Based Approach. Ecol. Ind. 2019, 102, 65–75. doi:10.1016/j.ecolind.2019.01.084.

Lelli, C.; Bruun, H.H.; Chiarucci, A.; Donati, D.; Frascaroli, F.; Fritz, Ö.; Heilmann-Clausen, J. Biodiversity Response to Forest Structure and Management: Comparing Species Richness, Conservation Relevant Species and Functional Diversity as Metrics in Forest Conservation. For. Ecol. Manage. 2019, 432, 707–17. doi:10.1016/j.foreco.2018.09.057.

Muurinen, L.; Oksanen, J.; Vanha-Majamaa, I.; Virtanen, R. Legacy Effects of Logging on Boreal Forest Understorey Vegetation Communities in Decadal Time Scales in Northern Finland. For. Eco. Manage. 2019, 436, 11–20. doi:10.1016/j.foreco.2018.12.048.

Velázquez, J.; Gutiérrez, J.; García-Abril, A.; Hernando, A.; Aparicio, M.; Sánchez, B. Structural connectivity as an indicator of species richness and landscape diversity in Castilla y León (Spain). For. Ecol. Manage. 2019, 432, 286–297. doi:10.1016/j.foreco.2018.09.035

Bertini, G.; Becagli, C.; Chiavetta, U.; Ferretti, F.; Fabbio, G.; Salvati, L. Exploratory analysis of structural diversity indicators at stand level in three Italian beech sites and implications for sustainable forest management. J. For. Res. 2019, 30, 121–127. doi:10.1007/s11676-018-0616-7.

Rubio, A.; Gavilán, R.G.; Montes, F.; Gutiérrez-Girón, A.; Díaz-Pines, E.; Mezquida, E.T. Biodiversity measures applied to stand-level management: Can they really be useful? Ecol. Ind, 2011,11, 545–556. doi:10.1016/j.ecolind.2010.07.011

Table S3: Correlation

Author		Positive correlation	Negative correlation	Method	Tool	Scale

	Indicators used in the study					
Karahalil et al. 2017	Deadwood, crown closure, understory trees, altitude, stand age	Deadwood volume and that of living trees, crown closure, altitude ($p < 0.01$), and stand age ($p < 0.05$).	he number of understory trees ($p < 0.01$) and DW	Pearson correlation	SPSS 16.0 ™ software.	Stand
Jonsson & Jonsell 1999	the abundance of dead trees, overall habitat diversity, stand age, the total richness of bryophytes	A strong correlation ($P < 0.05$) between the abundance of dead trees, overall habitat diversity and stand age		Principal Component Analysis (PCA)		Stand
		Mosses and vascular plants		Pearson correlation coefficients		1 ha
		the number of indicator species of wood-living fungi and vascular plants		Pearson correlation coefficients		1 ha
		The total richness of bryophytes and wood-living fungi		Pearson correlation coefficients		0.25 ha
		wood-living fungi and wood-inhabiting beetle species		Pearson correlation coefficients		1 ha
		wood-living fungi and the total number of beetle species		Pearson correlation coefficients		1 ha
		number of wood-inhabiting beetle species and number of fungi indicator species	number of indicator bryophytes and number of wood-inhabiting beetle species	Pearson correlation coefficients		1 ha
Smith et al. 2008	bryophytes, vascular plants, spiders, hoverflies and birds; structural and functional attributes	Dunnock, Wren and Blackbird with bird richness and abundances; Species richness of forest vascular plants, bryophyte and spiders increased with forest age; available P was positively correlated with vascular plant species richness	Goldcrest with bird species richness; Total bird species richness in Older forests was negatively correlated with site elevation	ANOVA/t-tests for categorical variables and correlation (Pearson's r) for continuous variables		Stand
Coote et al. 2013	bryophytes, vascular plants, spiders and birds; structural and functional attributes	Bryophyte species richness and relatively high canopy cover plantations on poorly drained soils; bird species richness and more open plantations with high shrub cover; coarse woody debris and forest-associated bryophytes; Both		ANOVAs or t-tests for categorical variables and correlation analysis (Pearson's) for continuous variables.	SPSS (2007); R Development Core Team (2012) using the ppcor package (Kim 2011)	stand

		proximity to old woodland and stand age and forest-associated vascular plants; stand age and forest-associated spiders				
Mikusiński et al. 2001	the richness of woodpecker species (strongly related to forest and those visiting other habitats) and the richness of other forest birdspecies (strongly related to forest and those visiting other habitats)	White-backed and Tree-toed woodpeckers (strongly related to forest) and forest bird species diversity		Linear regression analysis		landscape
		White-backed and Tree-toed woodpeckers (strongly related to forest) and bird species diversity strongly related to forest				
Gao et al. 2014	Stand structure parameters (canopy coverage, age of canopy trees, tree species composition and canopy stratification); soil classes (9); plant species diversity	soil class, stand structure parameters and plant species diversity/composition are all positively correlated in general; semi-open canopy and plant species diversity in young and middle-aged stands; plant diversity had a strong positive association with soil pH in mesic to moist soil conditions in temperate and boreal regions		General Linear Mixed Model	Microsoft Office Excel 2007	Stand
Humphrey et al. 1999	syrphid (hoverflies) and carabid (ground beetles) community composition and diversity, and stand structure and field layer vegetation.	vertical stand structure showed the best correlation with species richness and diversity of both carabids and syrphids				Stand
	individual tree mortality and composition and deadwood structure as an indicator of naturalness					

Laarmann et al.2009	mean deadwood mingling index (DMi), nature value score, diversity index of mortality causes (CMDI), number of mortality causes (CM) and recent deadwood volume (RDV5)	Nature value score significantly correlated with the diversity index of mortality causes (CMDI), indicating that CM are more diverse in semi-natural stands		Spearman correlation matrix		Stand
Sippola et al. 2004	species richness of polypores and timber variables; and between CWD volume and the management intensity.	The results show that the species richness of polypores in the boreal forest is connected not with the fertility the gradient of the forest site type, but with the amount and quality of CWD.		Spearman's non-parametric correlation		Stand
Treinys et al. 2016	Macrohabitat scale(Proportion of deciduous, coniferous, mixed forest and water body in a 2.8-km radius; Hydrological network density, km/km2) Nesting territory scale (Volume proportion of pine, spruce, broadleaves…; Shortest distance to the forest edge Shortest distance to the paved road Shortest distance to the dirt road	distances to the forest edge and houses were strongly interrelated; the coniferous proportion around nests at the macrohabitat scale correlated with the pine proportion at the nesting territory scale ($r = 0.74$,) and the pine proportion at the nesting territory scale correlated with the pine proportion at the nest site scale ($r = 0.55$).				Stand and landscape
Vangansbeke 2017	crested tit (Lophophanes *cristatus*), coal tit (*Periparus ater*), nightjar (*Caprimulgus europaeus*) and common lizard (*Zootica vivipara*) for estimation of the biodiversity of a patch	Coal tits seemed to prefer closed high forest without open patches, without too much recreation and from age class 81–100. Crested tits had a higher probability of occurrence in large high forest stands, with the low recreational intensity and a limited	The probability of occurrence of the coal tit was strongly negatively related to a higher recreation pressure and the amount of adjacent open patches	General linear model	R 3.0.1 (R Core Team 2013), using the multimodel inference package (MuMIn)	Stand

		amount of border with open habitat. The probability of occurrence for churring nightjars was higher in smaller stands with a high amount of adjacent open habitat. Also, some stand age classes had a much higher probability of occurrence for nightjars, particularly stands from age class 81–100, 21–40 and uneven-aged stands.				
Ferris et al. 2000	macrofungi species and plot environmental variables	Positive relationships were recorded between the increased volume of deadwood and the number of species of wood saprotrophs, and also between the species richness of ectomycorrhizal fungi and the number of tree species present in each plot. Significant correlations were also recorded between the number of parasitic fungal species and soil pH (a positive response to increasing alkalinity), and between the number of litter colonizing saprotrophs and tree species richness	the number of species of litter saprotroph was found to be negatively correlated with the number of tree species present in each plot	Pearson correlation		Stand
Suchant & Braunisch 2004	Capercaillie and stand variables (forest stand type, canopy closure, age class, species mixture, successional stage, stand height, vertical stratification, ground vegetation, soil type and cover as well as height of blueberry Vaccinium myrtillus)			Pearsons correlation coefficient and logistic regression		Stand
						Landscape

	Capercaillie and landscape variables (Altitude (m), Forest cover %, Slope°, Linear infrastructure m ha^{-1}, Exposition)			Pearsons correlation coefficient and logistic regression		
Suter et al. 2002	relationships between vegetation structure and avian diversity	Both Capercaillie and mountain birds responded positively to forest structure characterized by intermediate openness, multistoried tree layer, presence of ecotonal conditions, and the abundant cover of ericaceous shrubs		multiple-regression analysis		Stand
Gittings et al. 2006	forest road width; open space area with species the richness of the forest; small open space; large open space; open scrub hoverfly species groups	species richness of the open space fauna positively correlated with forest road width; Species with larvae feeding on the foliage of trees and shrubs associated with the presence of broad-leaved woody vegetation; Species with larvae developing in surface water habitats associated with wet habitat features.		Pearson's correlations		Stand
	habitat structure and hoverfly species richness	nearly 80%, of the species associated with open-space habitats rather than closed-canopy forest		ordination analyses of the habitat parameters; non-metric multidimensional scaling analysis (NMS)		
Oxborough et al. 2006	open space on ground-dwelling spider assemblages	At a large scale, the totalamount of open space within 200 m of sampling plots positively correlated with species richness and abundance.		Pearson's correlation		Stand and landscape
Montané et al. 2016	*V. myrtillus* and overstory and understory	*V. myrtillus* showed mostly positive associations with grasses and mosses.	Overstorey cover negatively influenced *V. myrtillus* cover, its height, and particularly, the number of fruits	Chisquare ($\chi 2$) and 2 × 2 contingency tables derived from presence/absence of the two cover types from the series	R Development Core Team 2011	Stand and landscape

				of point contacts, including Yates's correction (Kent & Coker 1992).		
Czeszczewik et al. 2015	Bird assemblages and forest management practices	The basal area of live trees had a positive effect on the abundance of birds	the density of live trees had a negative significant effect on bird abundance and species diversity	Not clear	R Development Core Team 2010	Stand

References:

Kim, S. 2011. ppcor: Partial and Semi-partial (Part) Correlation. R package Version; 1.0. http://cran.r-project.org/package=ppcor

Kent, M.; Coker P. 1992. Vegetation description and analysis. Boca Raton (FL): CRC Press

Definitions

Alpha diversity refers to the diversity within a particular area or ecosystem and is usually expressedby the number of species (i.e., species richness) in that ecosystem (Whittaker 1972).

ALS Airborne laser scanning, also commonly known by the acronym LiDAR (Light Detection And Ranging) is an active remote sensing technique, used to record the surface of the earth, specifically the topography of large areas of terrain and objects appearing on it. (gmv.cast.uark.edu/scanning-2/airborne-laser-scanning)

Beta diversity is diversity between ecosystems (Whittaker 1972).

Compositional variables should represent the types of elements that are characteristic of forests with a high degree of naturalness (Roberge et al. 2008b);[87]

Coppice systems consist of stands that originate from stool shoots or suckers of vegetative origin(Mura eta l. 2015).

Coarse Woody Debris (CW), i.e., deadwood pieces with a diameter >= 7 cm (Lafond et al. 2015).

Diversity is 'the variability among living organisms from all sources including among other things, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part. This definition includes diversity within species, between species and of ecosystems' (Convention on Biological Diversity 1992).

Double Brest Height (DBH), is the height at which tree diameter is typically measured (1.3m above the ground). (http://www.fao.org/3/ae578e/AE578E06.htm)

Indicator species are expected to indicate the status of an environment or to serve as proxies for a larger number of species (Hawskwort & Rose 1970; Block et al. 1986; Furness & Greenwood 1993 Mikusinski et al. 2001).

Indicandum (i.e. the indicated aspect of biodiversity).

Light detection and ranging (LiDAR), 'also known as laser detection and ranging (LaDAR) or optical radar, or ALS (Airborne laser scanning) is an active remote sensing technique which uses electromagnetic energy in the optical range to detect an object (target), determine the distance between the target and the instrument (range), and deduce physical properties of the object based on the interaction of the radiation with the target through phenomena such as scattering, absorption, reflection, and fluorescence'. (https://link.springer.com/referenceworkentry/10.1007%2F978-3-319-23386-4_44).

Fine woody debris, i.e., deadwood pieces with a diameter < 7 cm (Lafond et al. 2015).

Focal species approach is based on the idea that the conservation of specialised and area-demanding species can contribute to the protection of many naturally co-occurring species (Lambeck 1997; Hess& King 2002; Roberge & Angelstam 2004; Naumov et al. 2018).

Forest biodiversity is the diversity of all forms of life and their organisation within the forest area (Winter et al. 2011; Hunter 1990).

Functional variables concern the processes that are characteristic of naturally dynamic forests aswell as the anthropogenic processes that tend to move the ecosystems away from naturalness (Roberge et al. 2008b); [87].

Forest of high conservation value as any forest area which has been officially recognised asimportant for the conservation of forest biodiversity **(**Roberge et al. 2008a); [88].

Forest naturalness is a 'complex issue converging forest dynamics, disturbances at different scales, adaptation to changing the environment and human influence. Also, the level of naturalness is the extent to which human influence has affected the current forest structure and here forests are classifiedas e.g. old-growth, natural, recovering, or commercial forests, depending on the signs of management activities' (Uotila et al. 2002; Laarmann et al.2009).

Forest inventories gather compositional and structural information from samples that represent large areas and are used for monitoring national or regional forests (Standovár et al. 2016).

Gamma diversity is a measure of the overall diversity of the different ecosystems within a region (Whittaker 1972).

Large trees- e.g., trees with a diameter > 70 cm (Lafond et al. 2015).

Macrofungi are those species of fungi which produce a relatively conspicuous sporocarp (fruiting body); this group includes many Basidiomycetes (excluding rusts, smuts and yeasts) and some Ascomycetes (Pezizales) (Watling 1995).

Silvopastoral systems (dehesas or montado) are 'the integration of trees and shrubs in pastures with animals for economic, ecological and social sustainability' (http://www.fao.org/3/i1880e/i1880e09.pdf). These systems are typical in southwestern regions of the Iberian Peninsula (Mart´ın-Queller et al. 2011).

Species richness is the number of species, **species diversity** is the number of species in relation to their abundances and **species density** is the number of species per unit area. Different indexes can beused to measure species diversity (e.g. McIntosh index, which requires a number of individuals of allspecies; or Berger-Parker index which requires a number of individuals of the most abundant species and a total number of individuals) Lexer et al. (2000).

Structural variables refer to the spatial configuration of the elements, their quantities and habitats found in natural forests (Roberge et al. 2008a).

Forest birds are species that forage or nest in trees and that can breed in extensive forestland.

The natural forest can be defined as an idealized virgin forest condition that is not influenced by large-scale, systematic human activity (Bradshaw, 2005; Laarmann et al.2009)

The participatory GIS (Geographic Information System), recommends ways to collect, model, and visualize local information and opinions with GIS tools (Kangas et al. 2015).

Remote sensing is defined as 'the art, science and technology through which the characteristics of objects/targets either on, above or even below the Earth's surface are identified, measured and analysed without direct contact existing between the sensors and the objects or events being observed' (https://www.sciencedirect.com/topics/earth-and-planetary-sciences/remote-sensing).

Tree species diversity is based on the number of species and their relative abundance in basal area (Lafond et al. 2015).

Umbrella species are 'species whose conservation confers protection to a large number of naturally co-occurring taxa' (Fleishman et al. 2000; Roberge & Angelstam 2004; Roberge et al. 2008a).

DNA **metabarcoding: '** is an approach that combines DNA barcoding with next-generation sequencing (NGS), which enables sensitive high-throughput multispecies identification on the basis of DNA extracted from complex samples (Taberlet et al. 2012). DNA metabarcoding uses more or less universal polymerase chain reaction (PCR) primers to mass-amplify informative DNA barcode sequences (Straats et al. 2016; Fahner et al. 2016). Subsequently, the obtained DNA barcodes are sequenced and compared to a DNA sequence reference database from well-characterized species for taxonomic assignment (Taberlet et al. 2012; Fahner et al. 2016)' (Arulandhu et al. 2017).

References

Arulandhu, A. J.; Staats, M.; Hagelaar, R., et al. Development and validation of a multi-locus DNA metabarcoding method to identify endangered species in complex samples. GigaScience, 2017, 6(10), 1–18. https://doi.org/10.1093/gigascience/gix080

Taberlet, P.; Coissac, E.; Pompanon, F. et al. Towards next-generation biodiversity assessment using DNA metabarcoding. Mol Ecol 2012;21:2045–50.

Staats, M.; Arulandhu, A.J.; Gravendeel, B. et al. Advances in DNA metabarcoding for food andwildlife forensic species identification. Anal Bioanal Chem, 2016; 408:4615–30.

Fahner, N.A.; Shokralla, S.; Baird, D.J. et al. Large-scale monitoring of plants through environmental DNA metabarcoding of soil: recovery, resolution, and annotation of four DNA markers. PLoS One, 2016;11:e0157505.

Whittaker, R. J. Meta-analyses and mega-mistakes: calling time on a meta-analysis of the species richness-productivity relationship. Ecology 91, 2010, 2522–2533.

Lafond, V.; Cordonnier, T., Courbaud, B.. Reconciling Biodiversity Conservation and Timber Production in Mixed Uneven Aged Mountain Forests: Identification of Ecological Intensification Pathways. Environmental Management , 2015, 56:1118–1133

DOI 10.1007/s00267-015-0557-2

Watling, R. Assessment of fungal diversity: macromycetes, the problems. *Canadian Journal of Botany*, 1995, *73*(S1), 15-24.

Laarmann, D.; Korjus, H.; Sims, A. et al. Analysis of forest naturalness and tree mortality patterns in Estonia**.** Forest Ecology and Management, 2009, 258S S187–S195; doi:10.1016/j.foreco.2009.07.014

Uotila, A.; Kouki, J.; Kontkanen, H.; Pulkkinen, P. Assessing the naturalness of boreal forests ineastern Fennoscandia. Forest Ecol. Manag. 2002, 161, 257–277

Lambeck, R. J. Focal species: a multi-species umbrella for nature conservation. ConservationBiology, 1997, Pages: 849-856. Volume No.11

Hess, G.R.; King, T.J. Planning open spaces for wildlife: I. Selecting focal species using a Delphisurvey approach. Landsc. Urban Plan. 2002, 58, 25e40.

Chapter III - Using inventory variables for practical biodiversityassessment in plantation stands

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RESEARCH ARTICLE

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Using inventory variables for practical biodiversity assessment in plantation stands

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Abstract

Aim of study: Practically and simply assessing biodiversity by using inventory variables in four types of forest plantation stands (mixed and pure) including species such are chestnut, blue gum and maritime pine.

Area of study: Northwest Portugal in Vale do Sousa (14,840 ha), which is 97% covered with plantation forests.

Materials and methods: Simulated data, from 90-year stand-level forest management planning, were considered using three indicators: tree species (number of different species and species origin—native or exotic), mean diameter at breast height (DBH), and shrub biomass. Two shrub regeneration types (fully regenerated by seed and fully regenerated by resprouting), and three site quality conditions were also considered.

Main results: Mean biodiversity scores varied between very low (10.13) in pure blue gum stands on lowest-quality sites with shrub regeneration by seed, and low (29.85) in mixed stands with a dominance of pine, on best-quality sites with shrub regeneration by resprouting. Site quality and shrub regeneration type significantly affected all biodiversity scores in mixed stands dominated by pine and pure chestnut stands, while less affected pure blue gum stands and mixed stands dominated by blue gum.

Research highlights: The considered biodiversity indicators cover the major biodiversity aspects and allow biodiversity assessment over time. The findings are relevant for biodiversity conservation and fire protection management.

Additional key words: biodiversity indicators; forest function; forest structure; tree species composition; inventory variables; site index; shrub regeneration

Abbreviations used: DBH (diameter at breast height); FMM (forest management model); SI (site index).

Authors' contributions: Conceptualization, methodology, analysis and interpretation of data, statistical analysis and writing the manuscript: MC.

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Introduction

Forest plantations are often criticised due to a low compositional and structural biodiversity (Newbold *et al.,* 2015; Yamaura *et al.,* 2019), particularly mono-specific stands with exotic tree species (Hunter, 1990; Hartley, 2002; Carnus *et al.,* 2006). Forest biodiversity is the diversity of all forms of life and its organisation within the forest area (Hunter, 1990; Winter *et al.,* 2011). Forestry plantations are formed by planting or seeding for purposes that could be economical (*e.g.,* timber and fibre production) or protection (*e.g.,* soil conservation and carbon sequestration) (Carnus *et al.,* 2006; Stephens & Wagner, 2007). Indeed, natural forests typically host higher biodiversity than plantations, although the latter may support higher biodiversity than other intensive land uses, *e.g.* agriculture (Stephens & Wagner, 2007). The concern is that plantation forests with low biodiversity are considered more susceptible to disturbances and environmental changes than natural forests (Lugo, 1992; Carnus *et al.,* 2006; Bassi *et al.,* 2008; Proença *et al.,* 2010). Biodiversity contributes to the resilience of forest ecosystems and the delivery of different ecosystem services (https://millenniumassessment.org/en/Frameworkhtml; Proença *et al.,* 2010). It is necessary, therefore, to include biodiversity conservation aims in forest management plans (Ezquerro *et al.,* 2016). As most of the management operations in plantations are performed at a stand level, it is critical to ensure that biodiversity conservation is addressed at this scale (Similä *et al.,* 2006; Smith *et al.,* 2007). Moreover, assessing biodiversity at larger scales (*e.g.,* Botequim *et al.,* 2021) implies building from a smaller-scale approach such as the stand level.

Thus, forest managers need to include indicators for biodiversity optimization in forest management planning (Biber *et al.,* 2020) that may contribute to the design of resilient and sustainable landscape mosaics (Marto *et al.,* 2018; Botequim *et al.,* 2021). The definition and proper applications of biodiversity indicators are topics under permanent discussion. However, many scholars agree that biodiversity indicators need to be practical (*e.g.,* Ferris & Humphrey, 1999; Angelstam & Donz-Bruss, 2004; Smith *et al.,* 2007). Practical indicators are easy to apply, repeatable, cost-efficient and ecologically meaningful (Ferris & Humphrey, 1999; Smith *et al.,* 2007). Therefore, biodiversity indicators should be based on variables readily available in forest inventory datasets such as tree species composition (number of tree species per unit area) or diameter at breast height (DBH, *i.e,* diameter over bark measured at a height of 1.3 m above ground level) and tree height, for structural biodiversity (Ćosović *et al.,* 2020).

Trees are the dominating elements of forest ecosystems, and thus tree species composition is the most significant indicator of forest biodiversity which also contributes to structure definition (Stapanian *et al.,* 1997) and affects the composition of other forest communities (Martín-Queller *et al.,* 2011). For example, studies from North-Western Iberia have shown that plant and bird species composition are greater in native oak (*Quercus* spp.), maritime pine (*Pinus pinaster* A.), chestnut (*Castanea sativa* M.), and birch stands (*Betula alba* L.), than in non-native blue gum stands (*Eucalyptus globulus* L.) (Proença *et al.,* 2010; Goded *et al.,* 2019). Regarding structure variables that may influence biodiversity, such as tree height, biomass, diameter heterogeneity, and shrub volume, most are available in forest inventories and thus well known to forest managers, and also frequently available for public use (Ćosović *et al.,* 2020). In plantation forests, the understory layer is particularly important component of habitat structure and provides cover and food for wildlife (Smith *et al.,* 2007). Additionally, the understory is an indicator of ecological processes such as carbon storage, nutrient cycling and fire hazard risks (Botequim *et al.,* 2015). However, the structural properties of shrubs are species-dependent. Namely, the shrub species that regenerate by seeds typically develop lower bulk density than the shrubs that regenerate by resprouting, which is relevant for wildlife and the risk of wildfire predicting (Botequim *et al.,* 2015). Other variables, such as DBH, may also indicate stand structure as it relates to tree height, biomass growth or crown development. A particularly important aspect of forest biodiversity structure is trees with large diameters, as these are of great significance for the survival of numerous insects and birds (Badalamenti *et al.,* 2017). Additionally, forests that host large trees from diverse species are more resistant to disturbances than those with low tree species richness (Musavi *et al.,* 2017; Lutz *et al.,* 2018). Mature temperate forests, which are usually high in biodiversity, have higher large tree densities, mean diameters and total living biomass, in comparison to young stands (Burrascano *et al.,* 2013; Badalamenti *et al.,* 2017).

The present study aimed to demonstrate a practical and simple way to estimate stand-level biodiversity in plantation forests of Northwest Portugal. More precisely, for biodiversity assessment, the focus is on structural indicators: mean diameter (DBH) (cm) and shrub biomass (Mg ha^{-1}), but also compositional aspects such as tree species composition and species origin (native or exotic), and functional aspects such as shrub regeneration type and site index. To my knowledge, this research is the first to consider site index and shrub regeneration type in forest biodiversity assessment. Moreover, despite the large body of literature, the estimation of practical and quantitative indicators for application in the framework of managed forest management is still scarce.

Material and methods

Case study and collected data

The case study area was Vale do Sousa in Northwest Portugal, an area that extends over 14, 840 ha, out of which 97% is forest cover (for more information on the study area see Rodrigues *et al.,* 2020). Vale do Sousa can be considered representative of the forest landscape and forest management practices of this part of the country. The topography is very irregular, with a maximum elevation of 700 m (Marto *et al.,* 2018). The mean annual temperature is between 10 °C and 15 °C and the mean annual precipitation is quite high (1240 mm), though summer is typically dry, while autumn is very wet. Forest stands are managed according to four forest management models (FMMs), where each FMM has a different field management regime (silvicultural management). Two forest models (FMM1 and FMM2) assemble mixed stands of blue gum *(E. globulus)* and maritime pine *(P. pinaster)* where in FMM1, maritime pine is dominant (73%), while blue gum dominates in FMM2 (67%). FMM3 harbours pure chestnut (C. sativa) stands and FMM4 harbours pure blue gum stands. FMM1 and FMM2 are even-aged only at the beginning of the rotation period, while after the first eucalypt harvest, the stands became uneven-aged. Eucalypt is harvested three or more times before the first pine harvest (Table 1). FMM3 and FMM4 are even-aged stands.

There is a difference in tree species growth on various sites in Vale de Sousa, therefore, we considered tree site indexes (SI): SI1-low, SI3-medium and SI5-high. Site indexes were derived using data from the forest inventory such as the height and age of dominant trees for each stand (Rodrigues *et al.,* 2020). Blue gum and maritime pine stands are relatively evenly distributed across areas with all three site indexes, while chestnut covers mainly the high-quality sites (SI5).

The forest inventory data used here was collected within the ALTERFOR project (https://alterfor-project.eu/). These data were simulated along a 90-year forest planning horizon. The growth of maritime pine was simulated by the model PINASTER (Nunes *et al.,* 2011), and blue gum stands (FMM1, FMM2, and FMM4) were simulated by model GLOBULUS (Tomé *et al.,* 2006), where both models are implemented into the StandsSIM-MD module (Barreiro *et al.,* 2016). Chestnut stands growth (FMM3) was simulated by CASTANEA yield tables (Patrício, 2006). All growth models are empirical and such models "seek principally to describe the statistical relationships among data with limited regard to an object's internal structure, rules, or behaviour" (Korzukhin *et al.,* 1996). Shrub biomass accumulation (Mg ha^{-1}), was simulated according to Botequim *et al.* (2015) and considers the following management-related biometric variables: (i) stand basal area (m^2 ha^{-1}, with values obtained from growth and yield models described above); (ii) resprouter cover percentage (which considers fully seed and fully resprouting regenerative type strategies); (iii) shrub age (elapsed time since the last shrub clearing); and (iv) mean annual temperature (T = 14.5ºC).

Biodiversity indicators

The following variables were considered biodiversity indicators in this study: (1) tree species composition (tree species richness + species origin), (2) mean diameter (DBH, cm), and (3) shrub biomass (Mg ha^{-1}) with two levels of regeneration (by seed or by resprouting). A tree species composition indicator was created by combining tree species richness and the integer reflecting the number of native/non-native tree species in the stand. Thus, a value of '1' was assigned to one exotic species present in the stand and a value of '2' to one native species present in the stand. Correspondingly, a value of '3' was assigned to mixed pine/blue gum stands since maritime pine is a native species and blue gum exotic, '2' to pure chestnut (native species) stands and '1' to pure blue gum stands. The next step was to define the reference value of each indicator. This was based on the literature review, consulting peers and based on my own experience as a forest ecologist. In a report by Forest Europe (2020), tree species biodiversity is estimated across Europe in four categories: 1, 2-3, 4-5 and 6+ tree species. Forests with 6+ tree species are the rarest, and cover only 4.6% of European forests, while the most dominant are forests

Table 1. Four forest management models (FMMs) inventory variables and management practices.

FMMs	Tree density (trees ha^{-1})	% of study area	Thinning operation frequency	Fuel treatments	Harvesting
FMM1. Mixed maritime pine and blue gum forest system (*P. pinaster* + *E. globulus*) dominance of maritime pine 73%	Maritime pine 2200 Blue gum 1400	16.0	For pine—thinning every five years between 20 and 45 years	Every 5 years	Clear cutting systems for pine (45 years) / Coppice systems for blue gum (11 years)
FMM2. Mixed maritime pine and blue gum forest system (*E. globulus* + *P. pinaster*) dominance of blue gum 66%	Maritime pine 2200 Blue gum 1400	17.0	For blue gum —leaving two shoots at every stool on the 3rd year after the harvest		
FMM3. Chestnut *(C. sativa)* forest systems for the production of chestnut sawlogs	1250	1.0	Thinning every 5 or 10 years starting at age 15		Clear cutting systems (50 years)
FMM4. Blue gum *(E. globulus)* forest system for pulpwood production	1400	66.0	Leaving two shoots at every stool on the 3rd year after the harvest		Coppice systems (11 years)

with 2-3 species that cover half of all the European forests (Forest Europe, 2020). Regarding mainland Portugal, five native *Quercus* species comprise 93% of potential zonal native forests (Capelo *et al.,* 2007; Monteiro-Henriques & Fernandes, 2018). Therefore, the reference value of the tree species composition variable was considered to be 10. This means that it could represent the stand with, *e.g.,* ten exotic tree species, or five native species, or three native and four exotic species, and similar.

Regarding shrub biomass, two sources for the reference value were considered: the data used in the study and the literature. The highest value of shrub volume encroachment simulated in Vale Sousa with no shrub clearings scenario was 26 Mg ha^{-1}. Similarly, a study from northern Portugal reported 28.88 Mg ha^{-1} as the greatest shrub encroachment during 15 years of the post-fire period (Enes *et al.,* 2020). Then, in this work, the reference value utilized was the mean value between the data used in this study and the example found in Enes *et al.* (2020), which is 27 Mg ha^{-1}.

Regarding a reference value for mean diameter, according to national, European and global levels, about 60 cm of diameter is a suitable reference value. Hence, the average diameter of mature trees in Portugal is about 55 cm for 83 years old maritime pine (Pinto, 2004) and about 54 cm for 73 years old chestnut (Patrício & Nunes 2017). European forests are dominated by trees whose diameters are 21-40 cm, but about 8% of trees reaching 60 cm of DBH are found in uneven-aged forests (Forest Europe, 2020). Also, Lutz *et al.* (2018) recommended 60 cm as the fixed diameter threshold for large-diameter trees 'reached by at least some trees in almost all plots' in their study related to global forests. Therefore, the standard reference value of 60 cm was considered an appropriate value for a tree that contributes to biodiversity significantly.

Data analysis

The data of each indicator were normalized as percentages using the indicator's actual and reference values and the following formula: $x = (a/b)\cdot 100$, where x is the indicator's normalized percentage value, a is the actual indicator value and b is the reference value.

The normalized value was calculated for all three indicators. The average of these three indicator values was calculated to estimate the biodiversity value (index), ranging from 0 to 100. Further, five biodiversity categories were created as quintiles, where values between 0 and 20 corresponded to a very low value of biodiversity, 20-40 to low biodiversity, 40-60 to medium biodiversity, 60-80 to high biodiversity, and 80-100 to very high biodiversity. Normalization calculations were performed and the results were visualized in Excel (MS Office 2016). Also, a statistical summary, coefficient of variance, was carried out in the R programming language (R Core Team, 2020). Further, statistics were compared and biodiversity data was first assayed for normality with Shapiro-Wilks Normality Test. Since the data did not follow the normal distribution, the Kruskal-Wallis test was performed to check if there were differences in biodiversity data between groups with different site indexes and shrub regeneration types. After, multiple pairwise-comparison was carried out with Wilcoxon signed-rank test to see which groups were different. All comparative statistics were performed in the R programming language (R Core Team, 2020).

Results

Biodiversity in different forest management models (FMMs), shrub regeneration types and site indexes

The variations of biodiversity values of the four stand types over 90 years of management horizon, on three site quality conditions and two shrub regeneration types, are presented in Figs. 1 and 2. The summary of biodiversity data is presented in box plots (Fig. 3), Table S1 [suppl.] and Fig. S1 [suppl.] where it is shown that the highest mean value (29.85, low biodiversity) was recorded in mixed blue gum and maritime pine stands, with maritime pine dominance (FMM1) on the site index 5 (SI5) with shrub resprouting regeneration. The lowest mean value (10.13, very low) was recorded in pure blue gum stands (FMM4) on the SI1 with shrub seed regeneration. However, the highest maximum value (45.7, medium biodiversity) was recorded in pure chestnut stands (FMM3) with shrub resprouting regeneration and SI5, in the 49th year, right before the clear cut (Fig. 2). Medium category biodiversity score maximum values (41.72) were also recorded in FMM3 stands on shrub resprouting regeneration type and SI3 in the last year before the clear cut (49 years), and in FMM1 (41.24) in the year 40, which is 5 years before pine's clear cut. Regarding FMM2, all maximum values were low category biodiversity, while in FMM4 these were very low and low. Generally, across all site indexes, the mean biodiversity value was lower in shrub seed regeneration stands than in the resprouting stands (Fig. 3, Fig. S1 [suppl.]). In all stands with shrubs regenerating by resprouting, shrubs recovered faster (Figs. 1 and 2) and biomass was larger on average (Fig. 3). However, only in FMM3, there were clear differences in variations of biodiversity data between seed and resprouting stands (Fig. 4). Although, the coefficient of variations of all biodiversity data was high in all FMMs (>17). The highest variations were recorded in FMM4 (34-36), followed by FMM3 with variations between 26 and 33, while FMM1 and FMM2 had the lowest variations (18-21).

Regarding the site index, lower values generally reflected lower biodiversity within the same regeneration type.

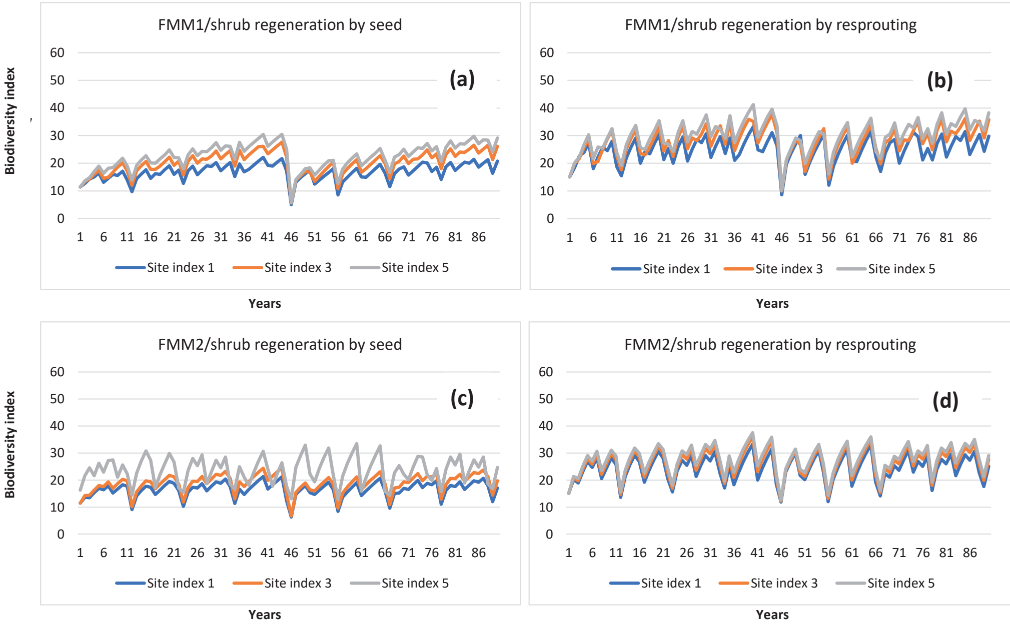

Figure 1. Biodiversity values for mixed blue gum and maritime pine stands, with a dominance of maritime pine (FMM1) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (a) and shrub resprouting regeneration (b); mixed blue gum and maritime pine stands, with a dominance of blue gum (FMM2) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (c) and shrub resprouting regeneration (d). Y axes show biodiversity values and X axes, 90 years of management horizon.

i.e., biodiversity values were always highest in SI5 and lowest in SI1 (Fig. 3). When biodiversity values between regeneration types were compared, SI1 with seed regeneration shrub type typically had the lowest biodiversity, and SI5 with resprouting regeneration type typically had the highest biodiversity (Fig. S2 [suppl.]). The majority of maximum values of all FMMs were higher in shrub resprouting regeneration type than in seed regeneration type, except in the case of FMM2 and FMM3, where SI1 shrub resprouting regeneration was lower than SI5 seed regeneration. There was a significant difference in biodiversity score between most of the site indexes within the same shrub

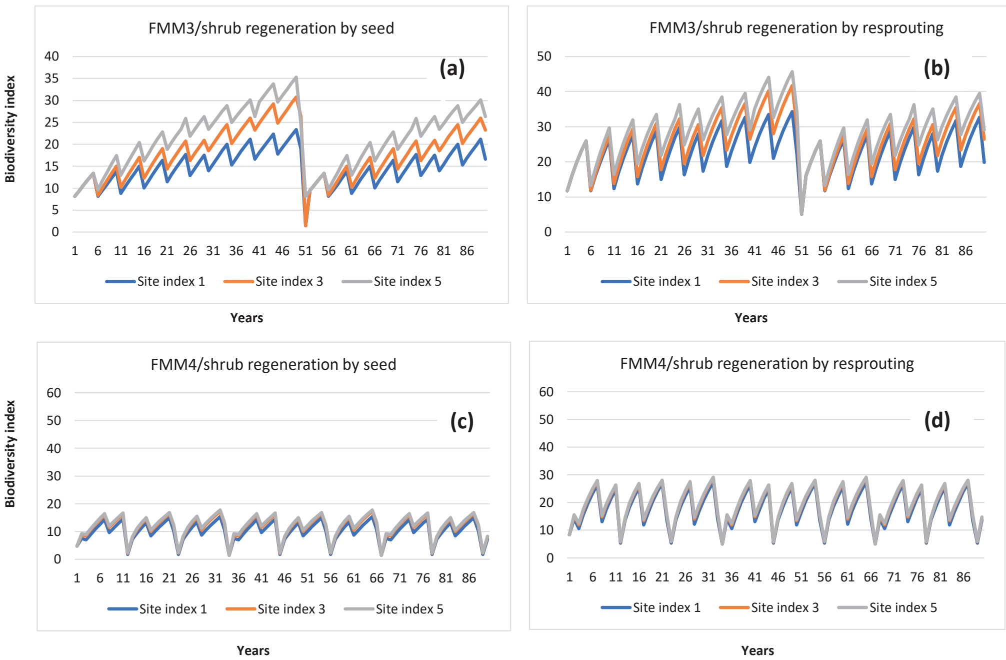

Figure 2. Biodiversity values for pure chestnut stands (FMM3) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (a) and shrub resprouting regeneration (b); pure blue gum stands (FMM4) with three site indexes (SI1, SI3 and SI5), with shrub seed regeneration (c) and shrub resprouting regeneration (d). Y axes show biodiversity values and X axes, 90 years of management horizon.

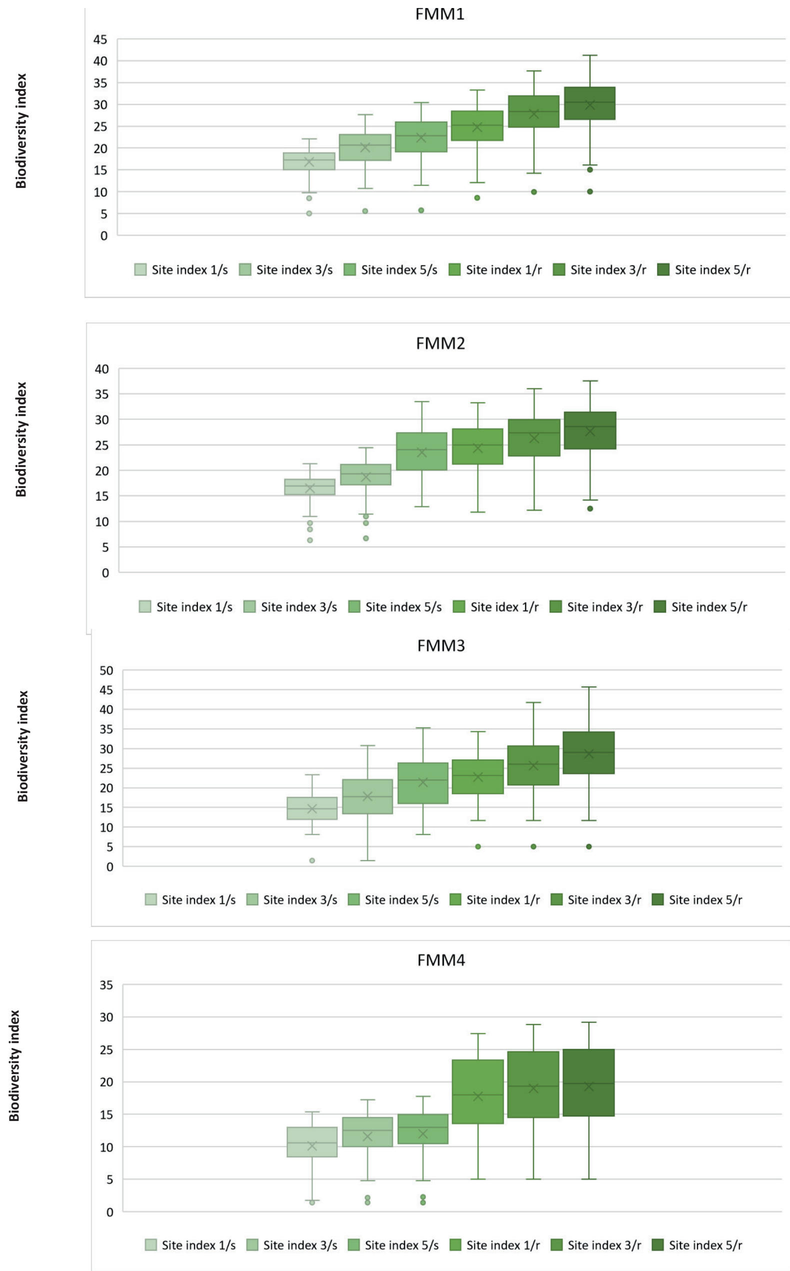

Figure 3. Boxplots of biodiversity values of FMM1, FMM2, FMM3 and FMM4 with three site indexes (SI1, SI3 and SI5), shrub seed regeneration (s) and shrub resprouting regeneration (r).

regeneration type, and between regeneration types (Table S2 [suppl.]). There was no significant difference (p>0.05) in FMM4 between SI3 and SI5 seed regeneration (p=0.30); between SI3 and SI5 resprouting regeneration (p=0.58); between SI1 and SI3 resprouting regeneration (p=0.13), and SI1 and SI5 resprouting regeneration (0.07) (95% of significance). Also, there was no significant difference in FMM2 between SI3 and SI5 resprouting (p=0.07), and between SI5 seed and SI1 resprouting (p=0.22).

Discussion

Biodiversity in plantation forests

In this study, biodiversity was assessed in four types of plantation stands (FMMs) in Northwest Portugal over the 90-year forest management planning horizon, using a method that combines biodiversity indicators and derives biodiversity scores varying from 0 to 100 (very low to very high). The

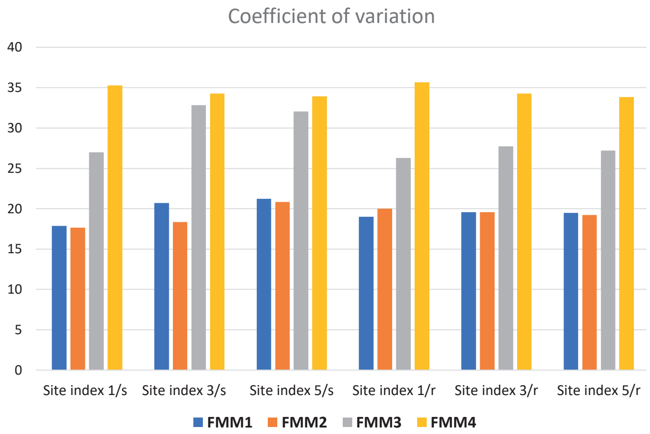

Figure 4. Coefficient of variation of biodiversity values in FMM1, FMM2, FMM3 and FMM4 with three site indexes (1, 3 and 5) and shrub seed regeneration (s) and shrub resprouting regeneration (r)

mean values of biodiversity, considering all FMMs, shrub regeneration category and site quality, varied between very low (10.13) and low (29.85), which confirms the statement that biodiversity in plantation forests, at a stand level, is typically low (Koh & Gardner, 2010; Newbold *et al.,* 2015; Yamaura *et al.,* 2019). However, maximum values were medium in the case of pure chestnut stands (45.71) and mixed stands with a dominance of maritime pine (41), and these values were reached only in mature years. Therefore, the state of biodiversity in plantations depends on the maturity stage. Pure exotic blue gum stands had the lowest mean biodiversity over the entire 90-year horizon and, in general, when compared to the other stands (mixed maritime pine and blue gum stands, and pure chestnut stands). This is in concordance with studies that argue that forests with pure stands of exotic species have the lowest biodiversity (Hunter, 1990; Hartley, 2002; Carnus *et al.,* 2006; Mikulová *et al.,* 2019). However, these low values in our case resulted from management practices such are frequent clear cuts (every 11 years). Plantation forests have very intense dynamics imposed by clear cuts (high coefficient of variation in the present study). Nevertheless, there is potential to develop higher biodiversity, if the time between clear cuts is extended. Therefore, in future research, it may be worth comparing the biodiversity value of exotic species plantations that have not had any silvicultural interventions, with unmanaged forests. Indeed, most plantations of exotic species have shorter rotation periods than native species plantations, and that is one of the main reasons why exotic species are introduced in the first place. For example, blue gum rotation in Portugal is typically 10-12 years (Deus *et al.,* 2019), while maritime pine is around 35 years (Oliveira, 1999; Dias & Arroja, 2012). The shorter the rotation period, the shorter the time necessary for biodiversity recovery and establishing of species interconnections and ecosystem stabilisation. These might be the main reasons for low levels of biodiversity in exotic mono-specific plantations.

Impact of shrub resprouting type and site index on biodiversity

There are differences in mean biodiversity values in the case of the shrub regeneration category in this study. Maximum values of biodiversity were also affected in the case of all FMMs except pure blue gum stands (FMM4). This can be explained by the short rotation of FMM4 (11 years), while other FMMs have nearly four times longer rotation periods such as maritime pine (45 years) in FMM1 and FMM2, and chestnut (50 years) in FMM3. There were also differences in the speed of shrub biomass development among stands between shrub regeneration by seed and by resprouting, where shrubs that regenerate by resprouting developed faster than shrubs that regenerated entirely by seed. Nevertheless, Botequim *et al.* (2015) reported opposite results, where shrubs that regenerate by resprouting developed lower biomass than shrubs that regenerate by seeds and particularly if the basal area was larger. Also, they reported shrubs that regenerated by seed recovered faster particularly if the basal area was lower. The reason for the difference might be that the study area was Mainland Portugal, covering managed, unmanaged forests and plantations, while the study area of this paper, had only plantations. Similarly, a study from central Argentina reported that, after a fire, shrub sprouting vigour was faster if wood density was low and the shrub was tall before the fire, while for small shrubs, wood density had no influence (Gurvich *et al.,* 2005). Also, a study that researched postfire shrub regeneration in heathlands from Australia (Pate *et al.,* 1990) reported slower growth of juvenile resprouters (<6 years), than non-resprouters. However, Pausas *et al.* (2004) found that resprouter traits can hardly be predicted on a global scale, but rather local, due to different responses of species in various areas. Therefore, there are indications that forest structure might influence the speed of shrub regeneration; however, more research is needed to examine locally regenerating traits of certain shrub species and their interaction with biodiversity.

Though numerous studies have reported better forest productivity in mixed stands than in monocultures (*e.g.,* Zhang *et al.,* 2012; Bielak *et al.,* 2014), such a case does not apply to clonal *Eucalyptus* plantations which are the world's fastest-growing plantations (Forrester & Bauhus, 2016). In this study, site index slightly affected mean and maximum biodiversity values in pure blue gum *(E. globulus)* while mixed stands with the dominance of maritime pine (FMM1), and pure chestnut stands (FMM3) were significantly affected by site quality. It can be concluded that the site index does not have a major effect on the biodiversity of short-rotation plantations. However, more research is needed on this topic.

Implications and suggestions for future management

Since the case study of this paper belongs to the Mediterranean geographic region, forest fires are widespread. In the past decade, forest fires became widespread all over the globe due to climate change. Frequently, maintaining a high conservation value habitat such as shrub formation may also imply a higher risk of wildfire (Silva *et al.,* 2020). Therefore, it is not advisable to increase shrub encroachment volume all over the area, but only where it is associated with high biodiversity importance (Botequim *et al.,* 2015). This will necessarily create trade-offs that need to be carefully considered. Additionally, knowledge about shrubs regenerating type may be used in fire prevention management, since shrubs that regenerate by resprouting develop much faster than shrubs that regenerate by seeds, in this case study area.

Extending the rotation period, increasing tree species composition with native species and leaving some big trees after clear-cutting may benefit biodiversity at the stand level. For example, Lafond *et al.* (2015) researched French Alps and found that retention measures of large trees, non-dominant species, and deadwood can compensate for the negative effect of intensive management practices. Initially, it might affect the income from timber production, but in the long run, it might decrease losses induced by pest and disease outbreaks. Additionally, payments for biodiversity conservation management may compensate for losses due to wood production. In Portugal and other Mediterranean countries, introducing species well adapted to forest fire, such as cork oak, may not only improve habitat quality for wildlife but, if well managed, even reduce fire risk.

Plantation forests can be useful in efforts for biodiversity conservation (Koh & Gardner, 2010), even plantations with exotic species can host native species. However, it would be more ecologically acceptable if native forests are restored and protected than to manage eucalypt plantations for biodiversity (Calviño-Cancela *et al.,* 2012).

Even though deadwood or old trees are fundamental indicators of forest biodiversity, those were not applied in this research due to the absence of such aspects in the case study area. However, since the results were mainly anticipated and demonstrated very low and low mean values of biodiversity, these indicators seem explicit and meaningful. The approach used in the present study is not intended for detailed scientific biodiversity assessments. It is rather suitable for initial estimation that will give the forest managers sense of the biodiversity state in their forests and help in management decisions. Also, it can serve as a base for detailed scientific estimations.

The present study demonstrates that forest inventory variables can be used as practical biodiversity indicators, and their combination can provide an overview of biodiversity at stand level over time. Site index and regeneration strategy are important aspects as they influenced the biodiversity of plantations with longer rotations such are those with maritime pine and chestnut, while less affected plantations with short rotations such are those with blue gum. Those findings are important for forest biodiversity conservation and fire prevention management.

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References

Angelstam P, Dönz-Breuss M, 2004. Measuring forest biodiversity at the stand scale: An evaluation of indicators in European forest history gradients. Ecol Bull 51: 305-332.

Badalamenti E, La Mantia T, La Mantia G, Cairone A, La Mela Veca DS, 2017. Living and dead aboveground biomass in Mediterranean forests: Evidence of old-growth traits in a *Quercus* pubescens Willd. sl stand. Forests 8(6): 187. https://doi.org/10.3390/f8060187

Barreiro S, Rua J, Tomé M, 2016. StandsSIM-MD: A management driven forest SIMulator. Forest Syst 25(2): eRC07. https://doi.org/10.5424/fs/2016252-08916

Bassi S, Kettunen M, Kampa E, Cavalieri S, 2008. Forest fires: causes and contributing factors to forest fire events in Europe. Study for the European Parliament Committee on Environment, Public Health and Food Safety under contract IP/A/ENVI/FWC/2006-172/LOT1/C1/SC10.

Biber P, Felton A, Nieuwenhuis M, Lindbladh M, Black K, Bahýľ J *et al.,* 2020. Forest biodiversity, carbon sequestration, and wood production: modelling synergies and trade-offs for ten forest landscapes across Europe. Front Ecol Evol 8: 291. https://doi.org/10.3389/fevo.2020.547696

Bielak K, Dudzińska M, Pretzsch H, 2014. Mixed stands of Scots pine (*Pinus sylvestris* L.) and Norway spruce [Picea abies (L.) Karst] can be more productive than monocultures. Evidence from over 100 years of observation of long-term experiments. Forest Syst 23: 573-589. https://doi.org/10.5424/fs/2014233-06195

Botequim B, Zubizarreta-Gerendiain A, Garcia-Gonzalo J, Silva A, Marques S, Fernandes PM *et al.,* 2015. A model of shrub biomass accumulation as a tool to support management of Portuguese forests. iForest 8: 114-125. https://doi.org/10.3832/ifor0931-008

Botequim B, Bugalho MN, Rodrigues AR, Marques S, Marto M, Borges JG, 2021. Combining tree species composition and understory coverage indicators with optimization techniques to address concerns with landscape-level biodiversity. Land 10: 126. https://doi.org/10.3390/land10020126

Burrascano S, Keeton WS, Sabatini FM, Blasi C, 2013. Commonality and variability in the structural attributes of moist temperate old-growth forests: A global review. For Ecol Manag 291: 458-479. https://doi.org/10.1016/j.foreco.2012.11.020

Calviño-Cancela M, Rubido-Bará M, van Etten EJ, 2012. Do eucalypt plantations provide habitat for native forest biodiversity? For Ecol Manag 270: 153-162. https://doi.org/10.1016/j.foreco.2012.01.019

Capelo J, Mesquita S, Costa JC, Ribeiro S, Arsénio P, Neto C. *et al.,* 2007. A methodological approach to potential vegetation modeling using GIS techniques and phytosociological expert-knowledge: Application to mainland Portugal. Phytocoenologia 37: 399-415. https://doi.org/10.1127/0340-269X/2007/0037-0399

Carnus JM, Parrotta J, Brockerhoff E, Arbez M, Jactel H, Kremer A *et al.,* 2006. Planted forests and biodiversity. J For 104(2): 65-77.

Ćosović M, Bugalho MN, Thom D, Borges JG, 2020. Stand structural characteristics are the most practical biodiversity indicators for forest management planning in Europe. Forests 11(3): 343. https://doi.org/10.3390/f11030343

Deus E, Silva JS, Larcombe MJ, Catry FX, Queirós L, dos Santos P *et al.,* 2019. Investigating the invasiveness of *Eucalyptus globulus* in Portugal: site-scale drivers, reproductive capacity and dispersal potential. Biol Invasions (6): 2027-2044. https://doi.org/10.1007/s10530-019-01954-6

Dias AC, Arroja L, 2012. Environmental impacts of eucalypt and maritime pine wood production in Portugal. J Clean Prod 37: 368-376. https://doi.org/10.1016/j.jclepro.2012.07.056

Enes T, Lousada J, Fonseca T, Viana H, Calvão A, Aranha J, 2020. Large scale shrub biomass estimates for multiple purposes. Life 10(4): 33. https://doi.org/10.3390/life10040033

Ezquerro M, Pardos M, Diaz-Balteiro L, 2016. Operational research techniques used for addressing biodiversity objectives into forest management: an overview. Forests 7(10): 229. https://doi.org/10.3390/f7100229

Ferris R, Humphrey JW, 1999. A review of potential biodiversity indicators for application in British forests. Forestry 72: 313-328. https://doi.org/10.1093/forestry/72.4.313

Forest Europe, 2020. State of Europe's forests. Ministerial Conference on the Protection of Forests in Europe, Madrid. https://foresteurope.org/

Forrester DI, Bauhus J, 2016. A review of processes behind diversity-productivity relationships in forests. Curr For Rep 2: 45-61. https://doi.org/10.1007/s40725-016-0031-2

Goded S, Ekroos J, Domínguez J, Azcárate JG, Guitián JA, Smith HG, 2019. Effects of *Eucalyptus* plantations on avian and herb species richness and composition in North-West Spain. Glob Ecol Conserv 19: e00690. https://doi.org/10.1016/j.gecco.2019.e00690

Gurvich DE, Enrico L, Cingolani AM, 2005. Linking plant functional traits with post-fire sprouting vigour in woody species in central Argentina. Austral Ecol 30 (8): 868-875. https://doi.org/10.1111/j.1442-9993.2005.01529.x

Hartley MJ, 2002. Rationale and methods for conserving biodiversity in plantation forests. For Ecol Manage 155: 81-95. https://doi.org/10.1016/S0378-1127(01)00549-7

Hunter ML, 1990. Wildlife, forests, and forestry: Principles of managing forests for biological diversity. Prentice-Hall, Englewood Cliffs, NJ. 370 pp.

Koh LP, Gardner TA, 2010. Conservation in human-modified landscapes. In: Conservation biology for all; Sodhi NS & Ehrlich PR (Eds). Oxford University Press, Oxford, pp: 236-261. https://doi.org/10.1093/acprof:oso/9780199554232.003.0014

Korzukhin MD, Ter-Mikaelian MT, Wagner RG, 1996. Process versus empirical models: which approach for forest ecosystem management? Can J For Res 26(5): 879-887. https://doi.org/10.1139/x26-096

Lafond V, Cordonnier T, Courbaud B, 2015. Reconciling biodiversity conservation and timber production in mixed uneven-aged mountain forests: identification of ecological intensification pathways. Environ Manage 56(5): 1118-1133. https://doi.org/10.1007/s00267-015-0557-2

Lugo AE, 1992. Comparison of tropical tree plantations with secondary forests of similarage. Ecol Monogr 62: 1-41. https://doi.org/10.2307/2937169

Lutz JA, Furniss TJ, Johnson DJ, Davies SJ, Allen D, Alonso A *et al.,* 2018. Global importance of large-diameter trees. Glob Ecol Biogeogr 27(7): 849-864.

Martín-Queller E, Gil-Tena A, Saura S, 2011. Species richness of woody plants in the landscapes of Central Spain: The role of management disturbances, environment and non-stationarity. J Veg Sci 22(2): 238-250. https://doi.org/10.1111/j.1654-1103.2010.01242.x

Marto M, Reynolds KM, Borges JG, Bushenkov VA, Marques S, 2018. Combining decision support approaches for optimizing the selection of bundles of ecosystem services. Forests 9: 438. https://doi.org/10.3390/f9070438

Mikulová K, Jarolímek I, Bacigál T, Hegedüšová K, Májeková J, Medvecká J *et al.,* 2019. The effect of non-native black pine (*Pinus nigra* JF Arnold) plantations on environmental conditions and undergrowth diversity. Forests 10(7): 548. https://doi.org/10.3390/f10070548

Monteiro-Henriques T, Fernandes PM, 2018. Regeneration of native forest species in mainland Portugal: identifying main drivers. Forests 9(11): 694. https://doi.org/10.3390/f9110694

Musavi T, Migliavacca M, Reichstein M, Kattge J, Wirth C, Black TA *et al.,* 2017. Stand age and species richness dampen interannual variation of ecosystem-level photosynthetic capacity. Nat Ecol Evol 1(2): 1-7. https://doi.org/10.1038/s41559-016-0048

Newbold T, Hudson LN, Hill SLL, Contu S, Lysenko I, Senior RA *et al.,* 2015. Global effects of land use on local terrestrial biodiversity. Nature 520: 45-50. https://doi.org/10.1038/nature14324

Nunes L, Tomé J, Tomé M, 2011. Prediction of annual tree growth and survival for thinned and unthinned even-aged maritime pine stands in Portugal from data with different time measurement intervals. For Ecol Manag 262: 1491-1499. https://doi.org/10.1016/j.foreco.2011.06.050

Oliveira A, 1999. Boas práticas florestais para o pinheiro bravo. Centro Pinus, Porto, Portugal.

Pate JS, Froend RH, Bowen BJ, Hansen A, Kuo J, 1990. Seedling growth and storage characteristics of seeder and resprouter species of Mediterranean-type ecosystems of SW Australia. Ann Bot 65: 585-601. https://doi.org/10.1093/oxfordjournals.aob.a087976

Patrício MS, 2006. Análise da potencialidade produtiva do castanheiro em Portugal. PhD Thesis. Universidade Técnica de Lisboa.

Patrício MS, Nunes L, 2017. Density management diagrams for sweet chestnut high-forest stands in Portugal. iForest-Biogeosci Forest 10(6): 865. https://doi.org/10.3832/ifor2411-010

Pausas JG, Bradstock RA, Keith DA, Keeley JE, 2004. Plant functional traits in relation to fire in crown-fire ecosystems. Ecology 85(4): 1085-1100. https://doi.org/10.1890/02-4094

Pinto I, 2004. Raw material characteristics of maritime pine (*Pinus pinaster* Ait.) and their influence on simulated sawing yield. VTT Technical Research Centre of Finland.

Proença VM, Pereira HM, Guilherme J, Vicente L, 2010. Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal. Acta Oecol 36(2): 219-226. https://doi.org/10.1016/j.actao.2010.01.002

R Core Team, 2020. R: A language and environment for statistical computing R Foundation for Statistical Computing, Vienna, Austria. https://wwwR-projectorg/

Roberge JM, Angelstam P, Villard MA, 2008. Specialised woodpeckers and naturalness in hemiboreal forests - Deriving quantitative targets for conservation planning. Biol Conserv 141(4): 997-1012. https://doi.org/10.1016/j.biocon.2008.01.010

Rodrigues AR, Botequim B, Tavares C, Pécurto P, Borges JG, 2020. Addressing soil protection concerns in forest ecosystem management under climate change. For Ecosyst 7: 1-11. https://doi.org/10.1186/s40663-020-00247-y

Silva V, Catry FX, Fernandes PM, Rego FC, Bugalho MN, 2020. Trade-offs between fire hazard reduction and conservation in a Natura 2000 shrub-grassland mosaic. Appl Veg Sci 23(1): 39-52. https://doi.org/10.1111/avsc.12463

Similä M, Kouki J, Mönkkönen M, Sippola AL, Huhta E, 2006. Co-variation and indicators of species diversity: can richness of forest-dwelling species be predicted in northern boreal forests? Ecol Indic 6: 686-700. https://doi.org/10.1016/j.ecolind.2005.08.028

Smith GF, Gittings T, Wilson M, French L, Oxbrough A, O'Donoghue S *et al.,* 2007. Identifying practical indicators of biodiversity for stand-level management of plantation forests. In: Plantation forests and biodiversity: oxymoron or opportunity? Springer, pp: 67-91. https://doi.org/10.1007/978-90-481-2807-5_4

Stapanian MA, Cassel DL, Cline SP, 1997. Regional patterns of local diversity of trees: associations with anthropogenic disturbance. For Ecol Manag 93: 33-44. https://doi.org/10.1016/S0378-1127(96)03944-8

Stephens SS, Wagner MR, 2007. Forest plantations and biodiversity: a fresh perspective. J For 105(6): 307-313.

Tomé M, Oliveira T, Soares P, 2006. O modelo Globulus 3.0. Publicações GIMREF-RC2/2006; Instituto Superior de Agronomia: Lisboa, Portugal. https://www.repository.utl.pt/handle/10400.5/1760?locale=en

Winter S, McRoberts RE, Bertini R, Chirici G, Bastrup-Birk A, Rondeux J *et al.,* 2011. The need for harmonized estimates of forest biodiversity indicators. In: National forest inventories: Contributions to forest biodiversity assessments. Springer: Dordrecht, The Netherlands. https://doi.org/10.1007/978-94-007-0482-4_1

Yamaura Y, Lindenmayer D, Yamada Y, Gong H, Matsuura T, Mitsuda Y, Masaki T, 2019. A spatially-explicit empirical model for assessing conservation values of conifer plantations. For Ecol Manag 444: 393-404. https://doi.org/10.1016/j.foreco.2019.04.038

Zhang Y, Chen HYH, Reich PB, 2012. Forest productivity increases with evenness, species richness and trait variation: A global meta-analysis. J Ecol 100: 742-749. https://doi.org/10.1111/j.1365-2745.2011.01944.x

Chapter IV - Forest plantations as a potential habitat for the red kite (Milvus milvus) raptor breeding

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Forest plantations as a potential habitat for the red kite (*Milvus milvus*) raptor breeding

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Abstract. To prepare forests better adapt to global change, forest management plans need to include biodiversity targets both at stand and landscape levels. Here we use an "umbrella" raptor bird species, the red kite (*Milvus milvus*), as an indicator to compare the landscape biodiversity performance of different forest plantations in NW Portugal. The plantations comprise pure blue gum (*Eucalyptus globulus* Labill), mixed blue gum and maritime pine (*Pinus pinaster* Aiton), pure chestnut (*Castanea sativa* Mill), pure maritime pine, pure pedunculate oak (*Quercus robur* L.), and pure cork oak (*Quercus suber* L.). We selected four indicators of habitat suitability based on breeding requirements for the red kite from literature: tree height (m), stem density (trees ha^{-1}), diameter at breast height (DBH) (cm), and frequency of silvicultural activities (number of activities year^{-1}). We analyse indicator trends over a forest management planning horizon of 90 years. Results show that the most favourable for breeding of the red kite are mature cork oak woodlands due to the low stem density, and sufficient height for nesting and harvesting without tree felling. Blue gum plantations are the least favourable due to high stem density and low DBH. Common oak forests seem unfavourable due to slow growth and height that never reaches the optimal values for the red kite to nest. Mixed forest plantations are better suited namely because of longer rotation periods of the maritime pine, although not the best because of the higher frequency of silvicultural practices. Chestnut and pure pine plantations are better suited for our conservation target, however, only in the last decade before the clear-cut. The indicators we applied could be useful for the conservation of other raptors with similar habitat requirements.

Keywords: red kite; biodiversity conservation; biodiversity indicators; habitat suitability; plantation forests

1. Introduction

Climate change is a major threat to the forest ecosystem's stability, resilience and provision of ecosystem services. Correspondingly, forests rich in genetic, structural, functional and compositional biodiversity are more likely to adjust to changing environments, than forests with low biodiversity (van Hensbergen 2006; Bauhus et al. 2010; Verheyen et al. 2016). Therefore, biodiversity conservation is relevant in all types of forests, including plantations, and thus should be part of forest management plans (Lindenmayer and Hobbs, 2004; Lindenmayer, 2009). Appropriate management of plantations, specifically at the landscape level, can generate suitable habitats for a set of native and threatened species (Atauri et al., 2004; Proença et al., 2010). To address biodiversity values generated by forest plantations, it is necessary to integrate appropriate biodiversity indicators into the management plans (Cosovic, 2022; Ćosović et al., 2020). Among biodiversity indicators, birds are commonly used for landscape-level biodiversity assessments (Uliczka et al., 2004; Smith et al., 2007; Naumov et al., 2018). Birds are among the most well-studied taxa in habitat suitability research (Angelstam et al., 2004; Naumov et al., 2018). Additionally, selecting “umbrella” species as biodiversity indicators is common, as they inhabit areas that support many other species and require large areas to sustain viable populations (Dale and Beyeler, 2001). Therefore, the conservation efforts directed at these species will also benefit other species with similar overlapping habitat requirements (Lambeck, 1997; Roberge, 2006). Further, avian umbrella species are associated with greater biodiversity than mammalian umbrella species (Branton and Richardson, 2011). Also, selecting “flagship” species is suitable, as they raise awareness for the environment and biodiversity protection in the area often due to their charismatic character (Sergio et al., 2008; Donázar et al., 2016). Avian raptors are often considered an “umbrella” and “flagship” species (Sergio et al., 2008; Barrientos and Arroyo, 2014; Cruz et al. 2021). The red kit (*Milvus milvus)* is such a species (Donázar et al., 2016) and occurs across Europe and in Northern Africa (IUCN, 2020). It regularly winters in Iberia (de Juana and Garcia, 2015) where it often nests in forest plantations and forages on surrounding agricultural areas or grasslands (Olano et al., 2016).

The red kite is listed in Annex I of the EU Birds Directive, and Annex II of the Bern Convention on the Conservation of European Wildlife and Natural Habitats (Sergio et al., 2019). The species is classified as “the least concern” on the IUCN Red List of Threatened Species globally, however, there is a decline in Portugal (BirdLife International, 2021). The red kite used to breed all over continental Portugal in the past (Voous, 1962; Mattsson et al. 2022), and today it can only be seen in the eastern area of the country (IUCN 2020). Poisoning and persecution are leading reasons behind the declining red kite populations in Iberia (Viñuela et al. 1999; Newbery et al. 2009; Mougeot et al. 2011), however, conversion of natural mature forests that are typically used for nesting is also a threat (Evans and Pienkowski, 1991; Maciorowski et al., 2021). Although, there is a lack of evidence on what is the main cause of the decline of breeding populations in Portugal.

Here we explore how different forest management models may contribute to the conservation of the red kite in a region of north-western Portugal dominated by forestry plantations. Since the red kite is an ‘umbrella’ species, we use the habitat requirements of the raptor species to frame biodiversity assessment and conservation at a landscape level. We selected indicators of the red kite nesting habitat suitability that can be readily used by forest managers with various backgrounds. These indicators are usually available in forest management models and the attributes that best relate to the red kite habitat preferences (e.g., dominant height, stem density, mean DBH, frequency of silvicultural activities) can be selected and examined.

The objectives of the present work are (1) to estimate breeding habitat suitability for the red kite in current and alternative forest management plans, and (2) to suggest changes to forest management plans that favour the conservation of the red kite. We hypothesise that current forest management models used in the study area are not suitable because observation of the red kite was last recorded in our study area in 2002. There is evidence that a very small number of migrating kites (1-9) flew all around the study area during the period from 2013 to March 2022, however, avoided flying over the area (see Raab et al. 2022). We also hypothesise that alternative forest models, namely those based on longer rotation periods and lower stem densities, should be more appropriate for red kite conservation.

2. Material and methods

2.1 Study area and Forest Management Models

The study area is known as Vale do Sousa and comprises 14, 840 ha in northwestern Portugal (Lat: 41.1343, Lon: −8.2951). The climate is the Mediterranean, with an Atlantic influence. Average temperatures are 9.5º C in winter and 20.8º C in summer. The average annual precipitation is 1,240 mm, with dry summers and rainy autumns. Soils are mainly shallow, poor and fast draining. The topography is hilly with elevations ranging from 20 m to 700 m. Forest dominates the area.

There are four forest management models (FMMs) currently applied in the study area. We define FMMs as models for predicting forest dynamics (e.g., tree growth) for certain tree species, by considering management practices, tree biometric properties and site conditions. There are FMM1 and FMM2 based on mixed-species plantations of maritime pine (*Pinus pinaster* Aiton) and blue gum (*Eucalyptus globulus* Labill). In FMM1, blue gum is the dominant species (73%) and in FMM2 maritime pine is the dominant species (66%) (Table 1). There are pure chestnut (*Castanea sativa* Mill) plantations (FMM3) and pure blue gum plantations (FMM4). Pure blue gum plantations (FMM4) cover 66% of the study area, mixed maritime pine and blue gum cover 33% (FMM1 and FMM2) and chestnut (FMM3) cover only 1% of the area.

The main product of blue gum plantations is pulpwood for the pulp and paper industry, while pine and chestnut forests are used for sawlog production. During a stakeholder workshop organized within the ALTERFOR project (https://alterfor-project.eu) which involved forest owners, pulp and paper industry agents, municipality representatives, scientists and forest authorities, three alternative FMMs were suggested for application in the study area (Marques et al., 2020). Suggesting alternative FMMs for the study areas was prompted by recent wildfires and the demand for novel ecosystem services beyond pulp and wood. Accordingly, an alternative FMM5 was proposed for pure maritime pine plantations with half the tree density of current models FMM1 and FMM2. Forests with FMM5 would generate resin and timber (provisioning services) as well as recreation (cultural services), while FMM6 was proposed for native pedunculate oak (*Quercus robur* L.) that dominated the landscapes of northern Portugal in the past. The main product of FMM6 would be saw logs of good quality, cultural services and open and more fire-resistant landscapes. Finally, FMM7 was projected for pure cork oak (*Quercus suber* L.) plantations generating cork for wine bottle stoppers, cultural services and also more fire-resistant landscapes.

Table 1: Current and alternative Forest Management Models (FMMs) in Vale do Sousa study area identified by main structural characteristics and silvicultural operations

FMM	Scientific name	Percent age of the study area (%)	Stem density (trees ha^{-1})	Harvesting	Thinning operations	Fuel treatments
1. Mixed maritime pine and eucalyptus forest system, the dominance of maritime pine 73%	*Pinus pinaster* Aiton + *Eucalyptus globulus* Labill	16.0	Maritime pine 2200 Blue gum 1400	Maritime pine-Clear cutting systems (45 years) Blue gum-Coppice systems (11 years) Clear cut --33^{rd} year	For pine—pre-commercial thinning at 10 years of age; thinning every five years between 20 and 45 year For blue gum —leaving two shoots at every stool on the 3^{rd} year after the harvest	Every 5 years
2. Mixed maritime pine and eucalyptus forest system, the dominance of eucalypt 66%	*Eucalyptus globulus* Labill + *Pinus pinaster* Aiton	17.0	Maritime pine 2200 Blue gum 1400			
3. Chestnut forest systems	*Castanea sativa* Mill	1.0	1250	Clear-cutting systems (50 years)	At ages 30 and 40	
4. Pure blue gum forest system	*Eucalyptus globulus* Labill	66.0	1400	Coppice systems (11 years) Clear cut --33^{rd} year	Leaving two shoots at every stool in the 3^{rd} year after the harvest	
5. Pure maritime pine forest systems	*Pinus pinaster* Aiton	-	~1150	Clear-cutting systems (40 years)	Pre-commercial-- 15 years Commercial—25, 35	
6. Pure pedunculate oak forest systems	*Quercus robur* L.	-	1600	Clear-cutting systems (60 years)	At ages 25–31, 35–40 and 43–47	
7. Pure cork oak forest systems	*Quercus suber* L.	-	1600	1st debarking-- 30 years 2nd debarking-- 40 years 3^{rd} and following debarking-- each 9 years	At ages 15, 30, 40, 58, 76	

We obtained the necessary variables for analyzing the current and alternative FMMs from yield tables built with field data collected from 200 inventory plots in Vale de Sousa in a frame of the European Union FP7-funded project ALTERFOR (https://alterfor-project.eu). The data relating to the stand growth and estimation of wood product yields were simulated with standsSIM-MD module (Barreiro et al., 2016) that implemented PINASTER forest model for maritime pine (Nunes et al., 2011), GLOBULUS forest model for blue gum (Tomé et al., 2006), and SUBER forest model for cork oak stands (Paulo et al., 2011, 2015), Gymma model (Barreiro et al., 2004) for uneven-aged eucalyptus stands and PBrirrol (Alegria, 2007) for uneven-aged maritime pine stands. The chestnut wood production was estimated using yield tables (Filipe, 2019; Patrício, 2006), and common oak growth was simulated with an online simulation tool (https://manuelar.shinyapps.io/Quercusrobur_SimGaliza/) (Gómez-García et al., 2015, 2016). The data were simulated over a 90-year management planning horizon.

2.2 The red kite conservation indicators

We selected the red kite as a target species because it can be considered both an “umbrella” and “flagship” species at a landscape scale due to its large territorial area. We searched the literature on the red kite habitat suitability in Iberia and selected appropriate indicators accordingly. First, we searched for all papers related to the red kite, then excluded those addressing direct human-induced mortality such as poisoning and windmill collision. Then, we screened papers related to the red kite habitat suitability, conservation, nesting and breeding.

One of the most important measures for the conservation of endangered species is to protect their breeding habitat (Olano et al., 2016). In the case of birds, safeguarding their nests and nest surroundings is of major importance for ensuring efficient breeding and population productivity (Zuberogoitia et al., 2008; Newton, 2013; Olano et al., 2016).

Red kites typically breed in forest patches surrounded by agricultural landscapes where the species search for food (Cramp and Simmons, 1980; Olano et al., 2016). Habitat selection may also vary between resident or wintering populations. For example, in Doñana National park in Spain, it was found that wintering kites preferred open areas such are marshes, while residential kites preferred forests (Heredia et al., 1991).

In Spain, the main tree nesting species for kites were full-grown stone pines (*Pinus pinea*), cork oaks (*Q. suber*) and large eucalypt trees (*Eucalyptus* spp.) (Heredia et al., 1991). However other studies reported nests only in Monterey pine (*Pinus radiata*) plantations and not in the surrounding native oak forests (Olano et al., 2016). A reason for this may be the height of pine trees being larger than that of native oaks (Olano et al., 2016). A similar phenomenon was noticed in a study from Germany (Nikolai et al., 2017), where, poplar trees (*Populus* sp.), planted as rows in farmlands, were the main nesting sites for the red kites. In addition, the height of nests has been constantly increasing in the past three decades, accompanying the growth of trees, from ~10 m in the eighties to ~18m in the last decade, with kites avoiding nesting in younger and smaller trees (Nikolai, et al., 2017). The height of nesting trees varied among the case studies wherein Corsica the average was 11.8 ± 4.6 m (Mougeot et al., 2011), in Germany average was 18-20 m (Ortlieb, 1989), in England average was 15 m (Carter, 2001) in Spain the average was 20 m (Olano et al., 2016). Tree height, more than the tree species identity, is, therefore, an indicator of habitat suitability for the red kite.

We could not find any information on the average tree diameter of a typical nesting tree, however, given that the red kite's average nest surface is 0.57 m^2 (Zduniak et al. 2021), the diameter must be large to sustain that surface. This is confirmed in a study that researched raptors with a similar mass as the red kite (approximately 1 kg) and their findings indicate that forest patches containing trees with large diameters (>40 cm), and 30-70% forest cover are important attributes for raptors’ breeding habitat conservation (Barrientos and Arroyo, 2014).

Regarding stem density, most of the red kite nests were found in stands with a moderate density of 100-500 trees^{-ha} (8 nests) in a study in Spain, while a lower number of nests were found in stands of higher density (Olano et al., 2016).

Beyond the structural characteristics of plantations, it is necessary to consider how silvicultural operations may affect target species. The red kite, for example, is sensitive to disturbances such as tree planting. A study from Wales (Newton et al., 1996) which examined the red kite population from 1992 to 1996, and particularly the effect of conifer plantation afforestation in the area, reported that tree planting initially disturbed egg-laying of the territorial pairs in the area (Newton et al., 1981; Newton et al., 1996). The egg-laying season is mainly in March-April, but some studies recorded egg-laying in late February or early May in Corsica (Mougeot and Bretagnolle, 2006; Mougeot et al., 2011). Incubation follows the first egg-laying and lasts until hatching takes place in several days while fledging emerges after 45-55 days (Nikolai, 2012), but sometimes the developmental period extends to 60-70 days, depending on brood size and food availability (Cramp and Simmons, 1980; Mougeot et al., 2011).

Following the findings in the literature, we selected four conservation indicators: (1) dominant tree height (m), (2) stem density (trees ha^{-1}), (3) mean diameter at breast height (DBH) (cm), and (4) frequency of silvicultural activities per year. We consider here silvicultural activities such are harvesting, thinning operations and fuel treatments for current and alternative models, and sum up all activities per year. For the harvesting year we assign 5, while for all other activities, we assign 1. We classified the suitability scores for each indicator as follows: 1 (Very low), 2 (Low), 3 (Medium), 4 (Good), and 5 (Very Good) (Table 2).

Table 2: Indicators for the red kite breeding suitability

	Score values for breeding suitability for the red kite				
Indicator	**1** (Very low)	**2** (Low)	**3** (Medium)	**4** (High)	**5** (Very high)
Dominant tree height (m)	<5	5-10	10-15	15-20	>20
Stem density (trees ha^{-1})	>1200	800-1200	500-800	300-500	100-300
Mean DBH (cm)	<15	15-20	20-30	30-40	>40
Frequency of clear cuts (Number of clear cuts per 90-year horizon)	>4	3	2	1	0

2.3 Data analysis

We combined three methods for data analysis: 1) plotting mean values of each variable per year of each FMM, 2) performing descriptive statistics of each variable per FMM, and 3) analysing the differences between each indicator variable in all FMMs. In the second step, we assigned the value of habitat suitability that we created in Table 2 to the means of each variable within FMMs and obtained the mean habitat suitability value for the red kite conservation per each FMM. For the third step, we initially used a Shapiro test for distribution analysis and since data of each FMM followed a normal distribution for all indicators except frequency of silvicultural activities, we applied ANOVA (Analysis of Variance) to check if there were significant differences among FMMs. When the results of ANOVA were significant, we applied the post hoc Tukey's Test for assessing individual differences among FMMs. The studentized range distribution q in Tukey's Test is defined as:

$$qs=(Ymax-Ymin)/SE$$

Here Ymax and Ymin are the larger and smaller means of the two groups that are compared, and SE is defined as the standard error of the entire design. The significance level of all statistical calculations was α =95% and all the calculations were performed in the R programming language (R Core Team, 2020). We visualized the results with letters where the FMMs with no significant differences are followed by a common letter. For such purpose, we used the *glht* function in the *multcomp* package version 1.4-17 in R programming language (R Core Team, 2020).

3. Results

We employed four forest inventory variables (dominant height, stem density, mean DBH, and frequency of silvicultural activities) as indicators to analyse seven types of plantation forests (FMMs) as potential red kite breeding habitats. We created a suitability score for each plantation type by combining the values of those indicators, and we also analysed each indicator separately. The results are presented in Table 3, where the highest overall habitat suitability is recorded in cork oak plantations (FMM7) as high (4), and the lowest suitability is recorded in blue gum plantations (FMM6) as low (1.75). Pure maritime pine plantations (FMM5) and chestnut (FMM3) marked medium suitability (2.5), while maritime pine in mixed stands (FMM1/FMM2) and common oak (FMM6) marked low overall suitability.

Table 3: Data summary of four indicators (DBH, dominant height, stem density and silvicultural activities) used to assess habitat suitability for the red kite. Mean values are indicated in bold. Value in brackets is suitability for red kite breeding. Overall habitat suitability per FMM over 90 years of the planning horizon is presented in the last row.

Indicators	Data summary	*P. pinaster* (FMM1/FMM2)	*E. globulus* (FMM1/FMM2)	*C. sativa* (FMM3)	*E. globulus* (FMM4)	*P. pinaster* (FMM5)	*Q. robur* (FMM6)	*Q. suber* (FMM7)
DBH (cm)	**Mean**	**13.67 (1)**	**7.946 (1)**	**13.50 (1)**	**9.066 (1)**	**11.90 (1)**	**6.997 (1)**	**30.00 (4)**
	Max.	26.40 (3)	13.000 (1)	33.26 (4)	14.500 (1)	24.20 (3)	21.120 (3)	56.42 (5)
Dominant height (m)	**Mean**	**10.82 (3)**	**11.48 (3)**	**10.531 (3)**	**13.94 (3)**	**10.41 (3)**	**2.6811 (1)**	**10.927 (3)**
	Max.	18.80 (4)	19.00 (4)	21.430 (5)	22.10 (5)	19.50 (4)	7.0003 (2)	16.710 (4)
Stem density (no trees ha^{-1})	**Mean**	**1349 (1)**	**1008 (2)**	**864.2 (2)**	**1008 (2)**	**767 (3)**	**897.6 (2)**	**445.4 (4)**

	Max.	1400 (1)	2071 (1)	1250 (1)	2071 (1)	767.0 (3)	897.6 (2)	1600 (1)
Frequency of clear cuts/90 years	**Total number**	**2 (3)**	**8 (1)**	**1 (4)**	**8 (1)**	**2 (3)**	**1 (4)**	**0 (5)**
Overall habitat suitability		**2 (Low)**	**1.75 (Low)**	**2.5 (Medium)**	**1.75 (Low)**	**2.5 (Medium)**	**2 (Low)**	**4 (High)**

3.1 Mean DBH

The greatest mean DBH, and at the same time the most suitable for red kite breeding, was recorded in cork oak forests (FMM7) (30 cm; high), while the lowest mean DBH was in common oak (7 m; very low) (Table 3). All the rest plantation types had a mean DBH lower than 15 cm, which is very low suitability. Such a ratio is in concordance with the result of Tukey's test where there is a significant difference between cork oak (FMM7) mean DBH and mean DBH of all the rest plantation types regarding 90 years of management horizon (Figure 3, Table A1).

Alternative maritime pine plantations (FMM5) have very low suitability (DBH <15 cm) until the age of 22, and current mixed plantations (FMM1 and FMM2) until the age of 24, and medium suitability (20-30cm) from the age of 30 and 32, until clearcut of alternative and current models respectively (Figure 2). The Chestnut range of DBH values (FMM3) is very similar to maritime pine both current and alternative (Figure 3), until age 40 when it outperforms pine's DBH and even reaches high values for the red kite conservation in the last five years of rotation. Regarding current and alternative blue gum (FMM1/FMM2/FMM4), even maximum values of DBH are with a very low value for the red kite conservation (13 cm and 14.5 cm). Common oak (FMM6) reaches medium values for the red kite conservation in the past two years of rotation.

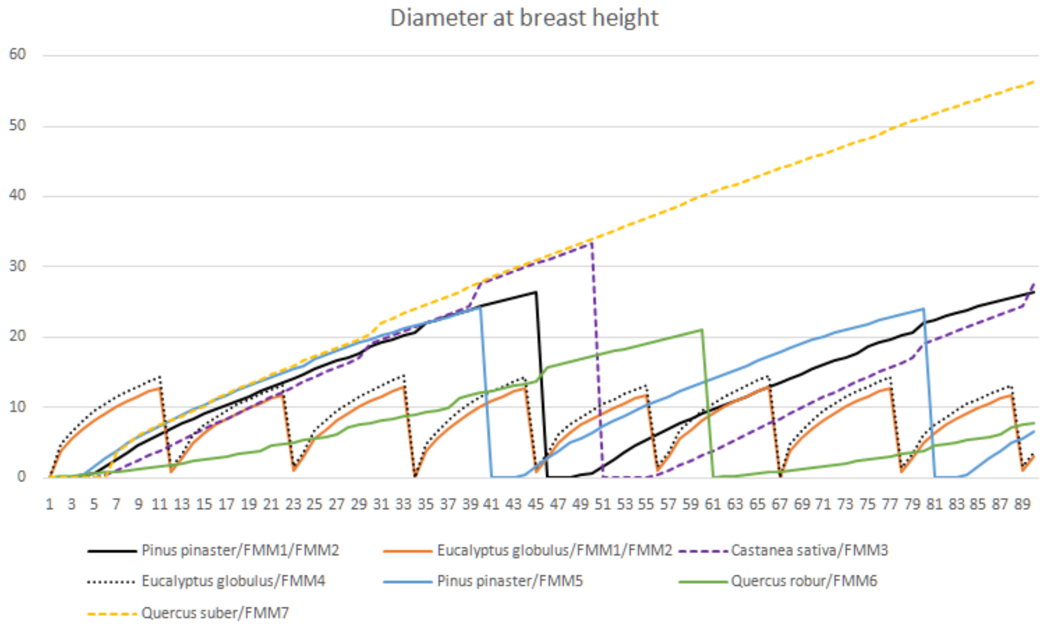

Figure 2: Mean diameter at breast height (DBH) variation of each tree species in current and alternative models (FMM1,2,3,4,5,6 and 7) in the study area over 90 years. Y-axes represent DBH (cm) and X axes represent 90 management horizon years.

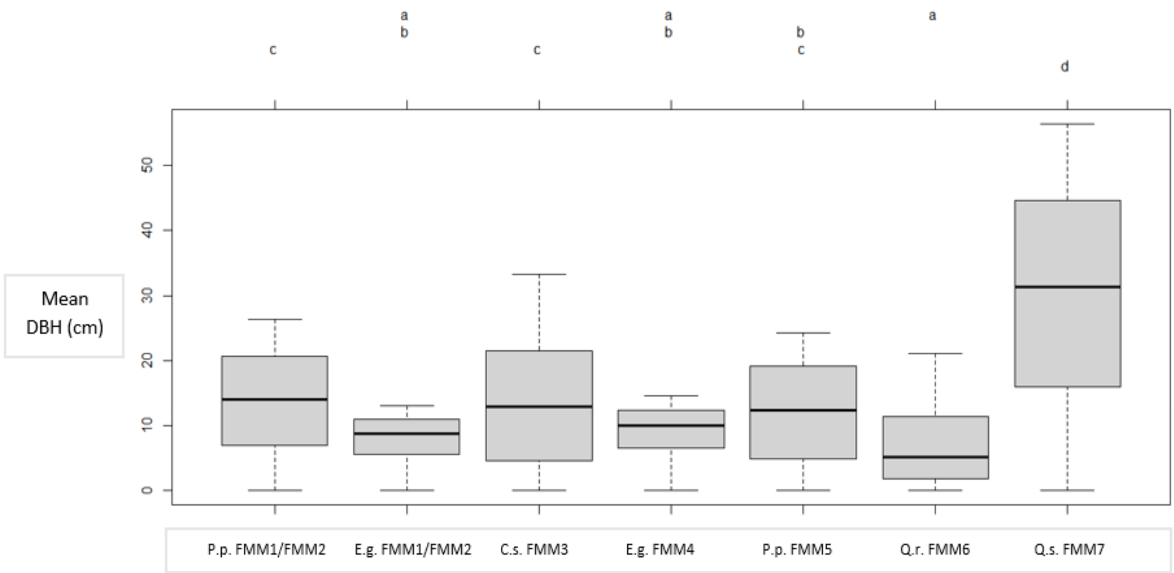

Figure 3: Comparison of mean DBH indicator values over 90 years of management horizon for all tree species in all FMMs. The unit for DBH is centimetres. The common letter shows no significant difference (Tukey's Test). Abbreviations: P.p.-Pinus pinaster, E.g.-Eucalyptus globulus, C.s.-Castanea sativa, Q.r.-Quercus robur, Q.s.- Quercus suber

3.2 Dominant height

Fast-growing blue gum plantations in the current models (FMM4) had the highest mean dominant height (13.9 m) over 90 years, of all other plantation types (Table 3). Only after a few years of growth, FMM4 reached a medium suitable height for the red kite nesting (10-15m) (Figure 4). A steep growth continued in the following 6-7 years until clear cut in the 11th year when the height reached high suitability of height (22.1) m. The lowest mean dominant height was in a common oak plantation (FMM6) with very low suitability for red kite nesting (2.7m). The rest of the plantations' mean dominant height was similar (~10 m), which is medium suitability. This ratio somehow matches with the result of Tukey's test where significant differences ($p<0.05$) in dominant height were only between pure blue gum plantations (FMM4) and all the rest plantation types, and also between common oak (FMM6) and all the rest plantation types (Figure 5; Table A2).

Maritime pine (FMM1, FMM2) takes exactly 31 years to reach high suitability (15-20m) for red kite nesting (Figure 4). The growth continues in the following 15 years until the clear-cut, where pine never reaches very high values (>20m). Chestnut plantation reached very good height (>20m) for red kite nesting, however, 2 years before the clear cut. Cork oak forest systems (FMM7) grow faster than the common oak, but still, takes 30 years to reach the medium height for nesting and 65 years to get a good height, while very good height is never reached before 90 years. The common oak forest systems (FMM6) grow at a very low pace and never reach the medium height for nesting because of the clear-cut in the 60th year when the tree reaches only 7 meters of height (Figure 4).

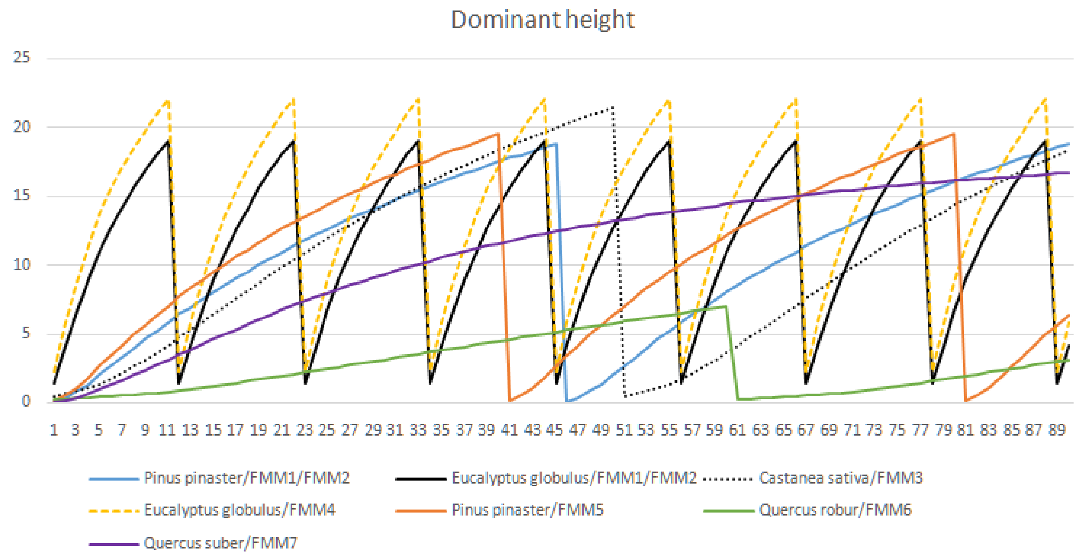

Figure 4: Dominant height variation of each tree species in current and alternative models (FMM1,2,3,4, 5, 6 and 7) in the study area over 90 years Y-axes represent tree height in meters and X axes represent years

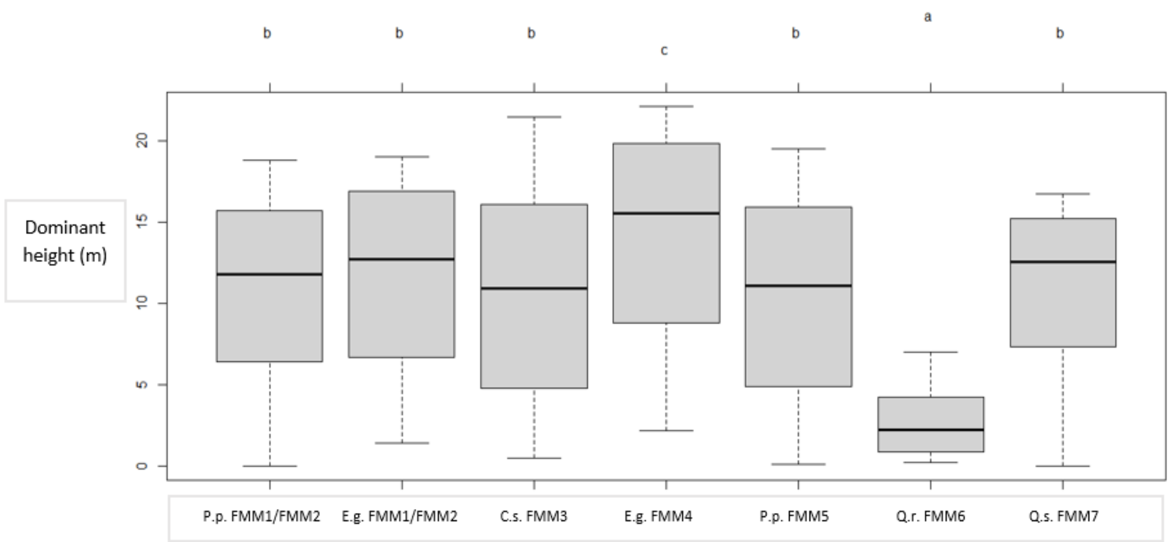

Figure 5: Comparison of dominant tree height indicator values over 90 years of management horizon for all tree species in all FMMs. The unit for dominant tree height is the meters. The common letter shows no significant difference (Tukey's Test). Abbreviations: P.p.-Pinus pinaster, E.g.-Eucalyptus globulus, C.s.-Castanea sativa, Q.r.-Quercus robur, Q.s.- Quercus suber

3.3 Stem density

The most suitable mean stem density over 90 years for red kite breeding was recorded in cork oak forests (FMM8) with high suitability (445.4 trees ha^{-1}), while the lowest suitability (1349 trees $ha^{-}1$; very low) was in current pine plantations (FMM1/FMM2) (Table 3). The rest of the plantations had mainly low suitability of stem density, except alternative pine plantations (FMM5) that had medium suitability (767 trees ha^{-1}). Similarly, the result of Tukey's test showed a significant difference (p< 0.005) between cork oak stem density and stem density of all the rest plantation types and also between current pine plantations and all the rest plantation types (Figure 7; Table A3). Also, there is a significant difference between alternative pine (FMM5) and blue gum plantations (FMM1/2/4).

The stem density of the current maritime pine plantation (FMM1,2) was of medium suitability in the last 5 years before the clear-cut (500-800 trees ha^{-1}) (Figure 6). Alternative maritime pine (FMM5) density was generally twice as low as that of the current pine models all over the 90 years horizon, where it ranges between low and medium suitability for red kite conservation. Chestnut plantation (FMM3) was approaching medium density for 25 years, and from 25 years until the clear cut (50 years) remain very close or within a high suitability range (300-500 trees ha^{-1}) for the red kite breeding (Figure 5). Cork oak (FMM7) had quite drastic changes in density as at the 15th year tree density has reached average values, and after the 30th year — trees density is scored as very good habitat until the end of 90 years management horizon. In common oak (FMM6), stem density reaches medium indicator values at the 26th year and decreases uniformly until the clear cut in the 60th year.

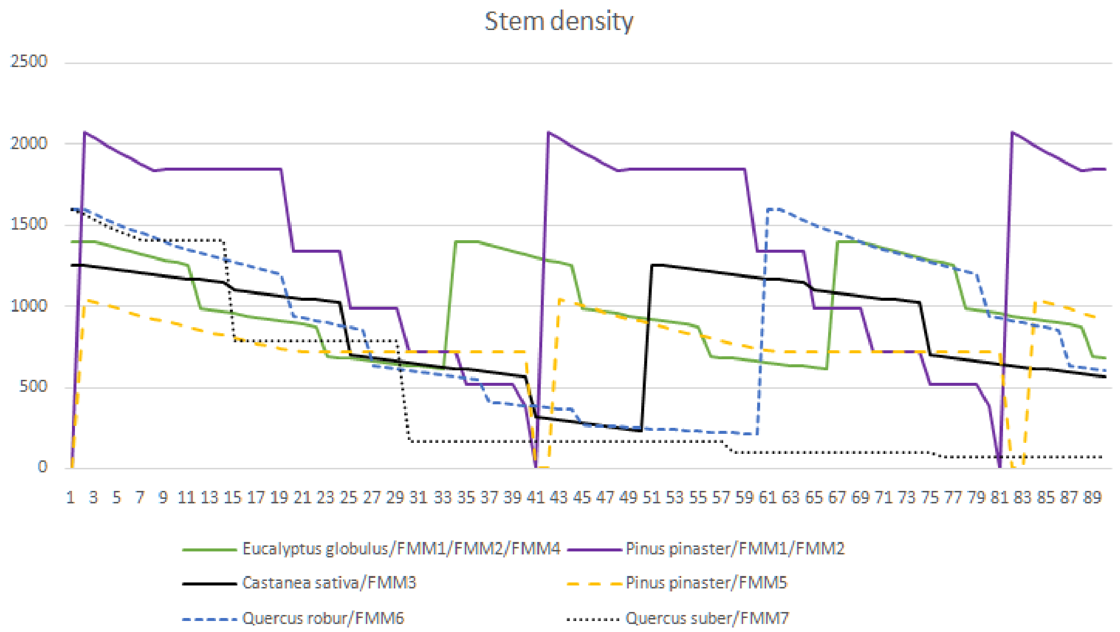

Figure 6: Stem density variation of each tree species in current and alternative models (FMM1,2,3,4,5,6 and 7) in the study area over 90 years Y-axis represents tree density per hectare and X-axis represents 90 management horizon years

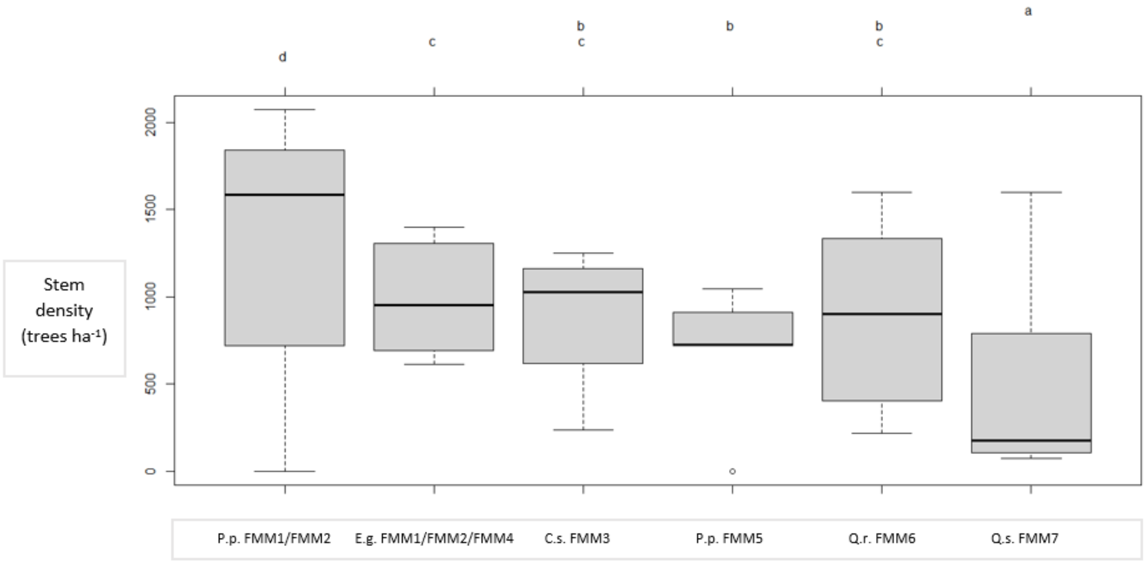

Figure 7: Comparison of stem density indicator values over 90 years of management horizon for all tree species in all FMMs. The unit for stem density is the number of trees^{-ha}. The common letter shows no significant difference (Tukey's Test). Abbreviations: P.p.-Pinus pinaster, E.g.-Eucalyptus globulus, C.s.-Castanea sativa, Q.r.-Quercus robur, Q.s.- Quercus suber

3.4 Frequency of clear-cuts

The most suitable rotation regime over 90 years for red kite breeding was recorded in cork oak forests (FMM8) with very high suitability. The least suitable is blue gum in current mixed and pure stands, due to very frequent clear-cuts, even eight over the 90-year management horizon. High suitability is recorded in chestnut and common oak stands, while medium suitability is in pine stands in current and alternative models.

4. Discussion

4.1 Habitat suitability of Mediterranean plantations for the red kite breeding and management suggestions

When considering average score values for 90 years, the native cork oak (*Q. suber*) plantation (FMM7) demonstrated high suitability for red kite nesting (4) in our study area and is by far the most suitable system for the red kite conservation of all other current and alternative plantations. The main reason for that is the absence of clear-cut over 90 years, and thus there is the possibility to provide tall, mature trees with large diameters. Also, mature cork oak tree management exercises low stem density. This is an optimal situation that needs to be equated to the long-term conservation of the red kites. Cork oak forest systems are known for their conservation value for biodiversity, mainly because of the diversity of the shrub and grassland understorey that provides food and cover for different wildlife species such as threatened raptors such as the imperial eagle (*Aquila adalberti*) (Bugalho et al., 2011). Although it takes longer for cork oak to reach the suitable tree density, and dominant height for red kite conservation (three decades), the trend remains stable and suitable for raptor nesting. Cork oak forests (FMM7) have one very different silvicultural activity from all other plantations, and that is bark removal every 9 years during spring and summer time. It should be examined how this might disturb raptors in Portugal. There is evidence from Spain that silvicultural activities such as cork-oak harvesting, disturbed raptor species cinereous vultures (Margalida et al., 2011; Guerrero-Casado et al., 2013). Also, understory layer clearing, tree thinning, canopy pruning and cork harvesting may affect tree-foraging birds (Ceia and Ramos, 2016). One of the solutions would be to decrease the noise as much as possible during the cork extraction (Margalida et al., 2011). However, the optimal solution is to set aside trees for nesting which would be beneficial for raptor species, but also for tree-foraging birds that are important for pest regulation (see Ceia and Ramos, 2016; Margalida et al., 2011).

Maritime pine (*P. pinaster*) had low (2) suitability for red kite conservation in mixed plantations (FMM1, FMM2) while in alternative pure plantations (FMM5), the suitability was medium (2.5). The difference is mainly due to stem density which is lower in FMM5 (mean 767 trees^{-ha}) and more suitable for the red kite than in the case of FMM1 and FMM2 (mean 1349 trees^{-ha}). However, in the last decade before the clear-cut (30-40 years), values of indicators of current maritime pine plantations have high suitability for red kite conservation, as much as alternative models, due to lower tree density, larger diameters and tree height. Similarly, a study from NW Spain researched the biodiversity of mature pine plantations (>60 years old) with stem density that ranged between 470-870 trees^{-ha}, and the results showed that pine plantations hosted a high abundance of fleshy-fruited species associated with a high abundance of birds, and other understory species composition and functional characteristics comparable to native common oak forests (Calviño-Cancela et al., 2012). Therefore, keeping a lower stem density (<800 trees^{-ha}) would contribute to higher plant species diversity and bird conservation. Alternative pine plantations in our study grow faster than current pine plantations due to lower stem density, where the height difference is as much as 2 m in the 40th year, and the mean DBH is greater (~2 cm). Another advantage of pure maritime pine plantations (FMM5) over mixed plantations (FMM1, FMM2) of maritime pine and blue gum is in less frequent silvicultural measures applied, as each tree species has different treatments (except for the fuel treatment that is the same in all stands). More frequent silvicultural measures impose more disturbance for site fidelity birds and that is the downside of greater tree species diversity in managed forests. However, alternative maritime pine plantations have a shorter rotation period than mixed plantations (approximately five years), with clear cuts occurring twice within the 90-year planning horizon.

Blue gum (*E. globulus*) in mixed (FMM1/FMM2) and pure plantations (FMM4) demonstrated low suitability (1.75) for red kite conservation primarily due to low DBH (<15cm) that never develops enough for red kite nesting because of frequent clear cut (every 11 years). Also, the mean stem density is too high (>1000 trees ha^{-1}). However, the mean dominant height in pure blue gum plantations was the highest of all other plantation types (>22 m). Therefore, blue gum still could serve as a suitable red kite nesting habitat if set aside or extended rotation to develop a large diameter. The red kite was reported to nest on eucalypt trees in Morocco (Radi et al., 2020) and Portugal (Ferreira et al., 2015). Extending the rotation period for >25 years would also contribute to higher biodiversity of common understory species that grow in native forests of NW Iberia (Calviño-Cancela et al., 2012).

Native chestnut (*C. sativa*) plantations (FMM3) had medium (2.5) suitability for the red kite conservation in our study, regarding overall 90 years of the planning horizon, due to low mean DBH (~13 cm), high mean stem density (>800 trees^{-ha}), while the mean height and frequency of clear-cut were medium and high, respectively. Although, native chestnut woodlands in NW Iberia can host high bird richness (39 species) among which is the raptor common buzzard (*Buteo buteo*) with a very similar body physique to the red kite (Guitián et al., 2012). However, regarding the indicators in the last decade before the clearcut (40-50 years), the mean suitability for red kite conservation is high (4), and in the last year before clearcut suitability is very good (4.5). We conclude that suitability for red kite is very low and low in the young stages, while high and very high in the mature stage (>40 years). This is in contrast with a previous study that assessed chestnut woodland biodiversity in NW Spain of various ages and levels of abandonment and reported that the degree of maturity of the woodland only affects the richness of plants, not the richness of birds, beetles and ants, however, birds' richness was related to woodland size (Guitián et al., 2012).

The alternative common oak plantation (*Q. robur*) (FMM6) had low (2) suitability for the red kite conservation in our study, regarding 90 years of the planning horizon, due to very low mean DBH, dominant tree height, and high stem density. Only a few years before the clear-cut (~57th year), the common oak plantation showed medium DBH value, while the optimal height for nesting in our study area is never reached. Similarly, a study from Spain (Olano et al., 2016) reported that even though the common oak was present in their study area, the red kite avoided nesting in oak stands and the nests were only found in Monterey pine plantations where taller trees were located. This enhances the importance of stand structural indicators, rather than species identity (native or exotic) when targeting the conservation of some avian raptors. Common oak forests are frequently reported as the most diverse ecosystems with higher biodiversity than other native and exotic forests and plantations (e.g., Proença et al., 2010; Calviño-Cancela et al., 2012; Goded et al., 2019). Even though common oak performed as unsuitable for the red kite habitat over the 90 years horizon, we would not suggest exempting this tree species from the landscape due to its great importance for biodiversity. Also, the common oak would surely become suitable habitat for the red kite if the planning horizon is larger than 90 years since it is a slow-growing species.

Achieving both production and biodiversity conservation aims is increasingly necessary and is an issue that must be addressed by forest management models. Here we show that FMMs perform differently according to the selected biodiversity target. We use the red kite as an example of an umbrella species of relatively large territorial areas. Other species could be used similarly, to address FMM's suitability for biodiversity conservation. Future research should quantify synergies and trade-offs. For example, increased thinning may increase wildlife habitat suitability simultaneously reducing the risk of wildfire. Conversely, setting aside conservation areas may imply wood or pulp production losses. All these factors need to be assessed and quantified in future research.

4.2 Trade-offs and synergies between red kite conservation and forestry

Implementing open grassland areas to replace the present FMM would favour red kite conservation but also contribute to forest fire prevention. Additionally, FMMs implementing lower tree density would benefit equally. This may be beneficial economically due to the great losses that are imposed by forest fires. According to Portugal's National Forestry Accounting Plan (2019), the cost of forest fires, phytosanitary measures and control of alien invasive species sums to 394M € per year. According to the results of our study, maritime pine plantations were more appropriate for red kite conservation than blue gum plantations. Portuguese government provides subsidies for maritime pine planting by reimbursing 75% of establishment costs for maritime pine, and 40% of pruning and weed control costs (RDP 2014-20 Measure 811 and 816) (Pra et al., 2019). However, for conservation measures, some forest areas should be set-aside. According to Olano et al., (2016), the red kite nests were found in a pine plantation in Spain in the area of ca 44 ha. If forest owners decide to conserve the red kite and set aside 44 ha of maritime pine forest, the annual cost for such a measure would be about 1144€. We estimated this cost by an Annual Equivalent Value (AEV) for maritime pine production in Northern Portugal with annual growth of pine of 14 m3ha-1 (Pra et al., 2019). Nevertheless, bird conservation is increasing the opportunity to introduce novel ecosystem services to the area, such as birdwatching. This branch of ecotourism which only recently became very popular can provide substantial economic and ecological benefits (Şekercioğlu, 2002; Donázar et al., 2016). However, economic revenue data related to birdwatching is rarely available in Europe, while in the USA, there are 47 M of birdwatchers and birdwatching is generating $107 billion in total, and creating more than 650,000 jobs (Carver, 2013; Donázar et al., 2016). Achieving both production and biodiversity conservation aims is increasingly necessary and is an issue that must be addressed by forest management models. Here we show that FMMs perform differently according to the selected biodiversity target. We use the red kite as an example of an umbrella species of relatively large territorial areas. Other species could be used similarly, to address FMM's suitability for biodiversity conservation. Future research should quantify synergies and trade-offs. For example, increased thinning may increase wildlife habitat suitability simultaneously reducing the risk of wildfire. Conversely, setting aside conservation areas may imply wood or pulp production losses. All these factors need to be assessed and quantified in future research.

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References

Alegria, C.M.M., 2007. PBIRROL Modelo de Crescimento e Produção para os Povoamentos de Pinheiro bravo de Estrutura Irregular do Concelho de Oleiros. Instituto Politécnico de Castelo Branco, Castelo Branco, Portugal. 13–18 p. (In Portuguese)

Angelstam, P., Roberge, J.M., Lõhmus, A., Bergmanis, M., Brazaitis, G., Dönz-Breuss, M., Edenius, L., Kosinski, Z., Kurlavicius, P., Lārmanis, V. and Lūkins, M., 2004. Habitat modelling as a tool for landscape-scale conservation: a review of parameters for focal forest birds. Ecological bulletins, pp.427-453.

Atauri, JA, de Pablo, CL, de Agar, PM, Schmitz, MF, Pineda, FD, 2004 Effects of management on understory diversity in the forest ecosystems of Northern Spain Environ Manage 34, 819e828

ALTERFOR (2018) Deliverable; Portugal Case Study Alternative models and robust decision-making for future forest management project The European Union's Horizon 2020 research and innovation programme

Barrientos, R., Arroyo, B., 2014. Nesting habitat selection of Mediterranean raptors in managed pinewoods: searching for common patterns to derive conservation recommendations. Bird Conservation International, 24(2), pp.138-151.

Barreiro, S., Rua, J. and Tomé, M., 2016. StandsSIM-MD: A management driven forest SIMulator. Forest systems, 25(2), pp.eRC07-eRC07.

Barreiro, S., M. Tomé, and J. Tomé. 2004. Modeling growth of unknown age even-aged eucalyptus stands. P. 34–43 in Modeling forest production. Scientific tools—data needs and sources.Validation and application, Hasenauer, H., and A. Mäkelä (eds.). Proceedings of the International IUFRO Congress, April 19–22, Vienna, Austria.

Bauhus, J., P. van der Meer, and M. Kanninen. 2010. Ecosystem goods and services from plantation forests. London: Earthscan.

Branton, M. and Richardson, J.S., 2011. Assessing the value of the umbrella-species concept for conservation planning with meta-analysis. Conservation Biology, 25(1), pp.9-20.

Bugalho, M.N., Caldeira, M.C., Pereira, J.S., Aronson, J. and Pausas, J.G., 2011. Mediterranean cork oak savannas require human use to sustain biodiversity and ecosystem services. Frontiers in Ecology and the Environment, 9(5), pp.278-286.

Calviño-Cancela, M., Rubido-Bará, M. and van Etten, E.J., 2012. Do eucalypt plantations provide habitat for native forest biodiversity? Forest Ecology and Management, 270, pp.153-162.

Carter, I., 2001. The Red Kite Arlequin Press, Essex, UK

Carver, E., 2013. Birding in the United States: a Demographic and Economic Analysis Report 2011-1 US Fish and Wildlife Service Arlington

Ceia, R.S. and Ramos, J.A., 2016. Effects of dominant tree species on insectivorous birds breeding in Mediterranean oak woodlands. Bird Study, 63(1), pp.115-127.

Cosovic, M (2022). Using inventory variables for practical biodiversity assessment in plantation stands. Forest Systems, Volume 31, Issue 2, e016. https://doi.org/10.5424/fs/2022312-18856

Ćosović, M., Bugalho, M.N., Thom, D. and Borges, J.G., 2020. Stand structural characteristics are the most practical biodiversity indicators for forest management planning in Europe. Forests, 11(3), p.343.

Cruz, C., Santulli-Sanzo, G., & Ceballos, G., 2021. Global patterns of raptor distribution and protected areas optimal selection to reduce the extinction crises. Proceedings of the National Academy of Sciences, 118(37), e2018203118.

Dale, V.H. and Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecological indicators, 1(1), pp.3-10.

De Juana, E., Garcia, E. 2015. The birds of the Iberian Peninsula Bloomsbury Publishing

Donázar, J.A., Cortés-Avizanda, A., Fargallo, J.A., Margalida, A., Moleón, M., Morales-Reyes, Z., Moreno-Opo, R., Pérez-García, J.M., Sánchez-Zapata, J.A., Zuberogoitia, I. and Serrano, D., 2016. Roles of raptors in a changing world: from flagships to providers of key ecosystem services. Ardeola, 63(1), pp.181-234.

Evans, I.M. and Pienkowski, M.W., 1991. World status of the Red Kite. British Birds, 84, pp.171-187.

ICNF (2005) Milvus milvus, Milhafre-real, Milhafre-de-rabo-de-bacalhau [Red kite] Page visited 7112019 http://www2icnfpt/portal/pn/biodiversidade/rn2000/resource/doc/rn-plan-set/aves/mil-milvus

IUCN, 2020. Red kite (*Milvus milvus*) https://www.iucnredlist.org/species/22695072/181651010#assessment-information (Accessed 16/03/2021)

Ferreira, R., Zina, H., Marques, A.T., Delgado, A., Venade, D. and Costa, H., 2015. Wintering population of Red kite Milvus milvus in the Castro Verde SPA (Portugal). Airo, 23, pp.55-60.

Filipe, A.F.L., 2019. Implementação de um modelo de crescimento para castanheiro no simulador da floresta StandsSIM.md. Master Thesis, Universidade de Lisboa: Lisboa

Goded, S., Ekroos, J., Domínguez, J., Azcárate, J.G., Guitián, J.A. and Smith, H.G., 2019. Effects of eucalyptus plantations on avian and herb species richness and composition in North-West Spain. Global Ecology and Conservation, 19, p.e00690.

Gómez-García, E., Crecente-Campo, F., Barrio-Anta, M. and Diéguez-Aranda, U., 2015. A disaggregated dynamic model for predicting volume, biomass and carbon stocks in even-aged pedunculate oak stands in Galicia (NW Spain). European Journal of Forest Research, 134(3), pp.569-583.

Gómez-García, E., Diéguez-Aranda, U., Cunha, M. and Rodríguez-Soalleiro, R., 2016. Comparison of harvest-related removal of aboveground biomass, carbon and nutrients in pedunculate oak stands and in fast-growing tree stands in NW Spain. Forest Ecology and Management, 365, pp.119-127.

Guitián, J., Guitián, P., Munilla, I., Guitián, J., Garrido, J., Penín, L., Domínguez, P. and Guitián, L., 2012. Biodiversity in chestnut woodlots: Management regimen vs woodlot size. Open Journal of Forestry, 2(04), p.200.

Guerrero-Casado, J., Arenas, R. and Tortosa, F.S., 2013. Modelling the nesting-habitat of the Cinereous Vulture Aegypius monachus on a fine scale for conservation purposes. Bird Study, 60(4), pp.533-538.

Heredia, B., Alonso, J.C. and Hiraldo, F., 1991. Space and habitat use by Red Kites Milvus milvus during winter in the Guadalquivir marshes: a comparison between resident and wintering populations. Ibis, 133(4), pp.374-381.

Kuuluvainen, T., Tahvonen, O., Aakala, T., 2012. Even-aged and uneven-aged forest management in boreal Fennoscandia: A review Ambio 41: 720–737

Lambeck, R.J., 1997. Focal Species: A Multi-Species Umbrella for Nature Conservation: Especies Focales: Una Sombrilla Multiespecífica para Conservar la Naturaleza. Conservation biology, 11(4), pp.849-856.

Lindenmayer, D.B., 2009. Forest wildlife management and conservation. Annals of the New York Academy of Sciences, 1162(1), pp.284-310.

Lindenmayer, D.B. & R.J. Hobbs. 2004. Fauna conservation in Australian plantation forests—A review. Biol.Conserv. 119: 151–168.

Marques, M., Juerges, N. and Borges, J.G., 2020. Appraisal framework for actor interest and power analysis in forest management-Insights from Northern Portugal. Forest Policy and Economics, 111, p.102049.

Maciorowski, G., Zduniak, P., Bocheński, M., Urbańska, M., Król, P. and Polakowski, M., 2021. Breeding habitats and long-term population numbers of two sympatric raptors—Red Kite Milvus milvus and Black Kite M. migrans—in the mosaic-like landscape of western Poland. Journal of Ornithology, 162(1), pp.125-134.

Margalida, A., Moreno-Opo, R., Arroyo, B.E. and Arredondo, A., 2011. Reconciling the conservation of endangered species with economically important anthropogenic activities: interactions between cork exploitation and the cinereous vulture in Spain. Animal Conservation, 14(2), pp.167-174.

Mougeot, F., Garcia, J.T. and Viñuela, J., 2011. Breeding biology, behaviour, diet and conservation of the red kite (Milvus milvus), with particular emphasis on Mediterranean populations. Ecology and conservation of European dwelling forest raptors and owls, pp.190-204.

Mougeot, F., Bretagnolle, V., 2006. Breeding biology of the Red Kite Milvus milvus in Corsica Ibis, 148(3), 436-448

Maciorowski, G., Zduniak, P., Bocheński, M., Urbańska, M., Król, P., Polakowski, M., 2021. Breeding habitats and long-term population numbers of two sympatric raptors—Red Kite *Milvus milvus* and Black Kite *M. migrans*—in the mosaic-like landscape of western Poland. Journal of Ornithology, 162(1), 125-134.

National Forestry Accounting Plan Portugal, 2019. National Forestry Accounting Plan Portugal 2021-2025 Final version, 1 March 2019 Agencia Portuguesa do Ambiente Ambiente e transicao energetica, Republica Portuguesa

Naumov, V., Manton, M., Elbakidze, M., Rendenieks, Z., Priednieks, J., Uhlianets, S., Yamelynets, T., Zhivotov, A. and 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, pp.1-13.

Newton, I., Davis, P.E. and Moss, D., 1994. Philopatry and population growth of red kites, *Milvus milvus*, in Wales. Proceedings of the Royal Society of London. Series B: Biological Sciences, 257(1350), pp.317-323.

Newton, I., Davis, P.E., Moss, D., 1996. Distribution and breeding of red kites *Milvus milvus* in relation to afforestation and other land-use in Wales Journal of Applied Ecology, 210-224

Newton, I., Davis, P.E. and Moss, D., 1981. Distribution and breeding of red kites in relation to land-use in Wales. Journal of Applied Ecology, pp.173-186.

Newton, I., 2013. Bird populations Collins New Naturalist Library London

Nicolai, B., Mammen, U. and Kolbe, M., 2017. Long-term changes in population and habitat selection of Red Kite *Milvus milvus* in the region with the highest population density. Vogelwelt, 137, pp.194-197.

Nikolai, B., 2012. Rotmilan Red Kite Roter Drachen Katalog zur gleichnamigen Ausstellung des Museums Heineanum in Halberstadt Museum Heineanum · Domplatz 36 · D-38820 Halberstad [This series is included in the abstracting and indexing coverage of Zoological Record] ISSN 0947-1057

Nunes, L., Tomé, J. and Tomé, M., 2011. Prediction of annual tree growth and survival for thinned and unthinned even-aged maritime pine stands in Portugal from data with different time measurement intervals. Forest ecology and management, 262(8), pp.1491-1499.

Olano, M., Beñaran, H., Laso, M. and Arizaga, J., 2016. Exotic pine plantations and the conservation of the threatened Red Kite *Milvus milvus* in Gipuzkoa, Northern Iberia. Ardeola, 63(2), pp.369-374.

Ortlieb, R.,1989. Der Rotmilan Neue Brehm Bu¨cherei Bd 532, Westarp-Wissenschaften, Hohenwarsleben

Patrício, M.S., 2006. Análise da Potencialidade Produtiva do Castanheiro em Portugal. PhD Thesis, Universidade Técnica de Lisboa: Lisboa

Paulo, J.A., Tomé, J., Tomé, M., 2011. Nonlinear fixed and random generalized height–diameter models for Portuguese cork oak stands. Annals of Forest Science, 68, 295–309, doi:10.1007/s13595-011-0041-y.

Paulo, J.A., Palma, J.H., Gomes, A.A., Faias, S.P., Tomé, J. and Tomé, M., 2015. Predicting site index from climate and soil variables for cork oak (*Quercus suber* L.) stands in Portugal. New Forests, 46(2), pp.293-307.

Perhans , K., Glöde , D., Gilbertsson, J., Persson, A., Gustafsson, L., 2011. Fine-scale conservation planning outside of reserves: Cost-effective selection of retention patches at final harvest Ecol Econ 70: 771–777

Pra, A., Masiero, M., Barreiro, S., Tomé, M., De Arano, I.M., Orradre, G., Onaindia, A., Brotto, L. and Pettenella, D., 2019. Forest plantations in Southwestern Europe: A comparative trend analysis on investment returns, markets and policies. Forest Policy and Economics, 109, p.102000.

Proença, V.M., Pereira, H.M., Guilherme, J. and Vicente, L., 2010. Plant and bird diversity in natural forests and in native and exotic plantations in NW Portugal. Acta Oecologica, 36(2), pp.219-226.

Raab, R., Aebischer, A., Kovacs, F. J., Böing, H. & Aberle, S. (2022). Der Rotmilan in Europa - Erfolgreicher Schutz auf internationaler Ebene. Austrian Power Grid (APG), Wien.

Radi, M., Iglesias, J.J., El Khalimchi, R., Martín, J., Rousselon, K., Lorcher, F., Belamine, M., Torralvo, C., Fajardo, B., Farjardo, I. and Garrido, J.R., 2020. New roosting site of Red Kite *Milvus milvus* in the province of Ifrane (Morocco). Go-South Bulletin, 17, pp.4-7.

R Core Team, 2020. R: A language and environment for statistical computing R Foundation for Statistical Computing, Vienna, Austria URL https://wwwR-projectorg/

Roberge, J.M., Lämås, T., Lundmark, T., Ranius, T., Felton, A. and Nordin, A., 2015. Relative contributions of set-asides and tree retention to the long-term availability of key forest biodiversity structures at the landscape scale. Journal of Environmental Management, 154, pp.284-292.

Roberge, J.M., 2006. Umbrella species as a conservation planning tool (Vol. 2006, No. 2006: 84).

Seoane, J., Viñuela, J., Díaz-Delgado, R., Bustamante, J., 2003. The effects of land use and climate on red kite distribution in the Iberian peninsula Biological Conservation, 111(3), 401–414 https://doiorg/101016/S0006-3207(02)00309-9

Sergio, F., Tanferna, A., Chicano, J., Blas, J., Tavecchia, G. and Hiraldo, F., 2019. Protected areas under pressure: Decline, redistribution, local eradication and projected extinction of a threatened predator, the red kite, in Doñana National Park, Spain. Endangered Species Research, 38, pp.189-204.

Sergio, F., Newton , I., Marchesi , L.,Pedrini , P., 2006. Ecologically justified charisma: preservation of top predators delivers biodiversity conservation. J Appl Ecol 43:
1049–1055

Sergio, F., Caro, T., Brown, D., Clucas, B., Hunter, J., Ketchum, J., McHugh, K. and Hiraldo, F., 2008. Top predators as conservation tools: ecological rationale, assumptions, and efficacy. Annual review of ecology, evolution, and systematics, 39, pp.1-19.

Smith, G.F., Gittings, T., Wilson, M., French, L., Oxbrough, A., O'Donoghue, S., O'Halloran, J., Kelly, D.L., Mitchell, F.J., Kelly, T. and Iremonger, S., 2007. Identifying practical indicators of biodiversity for stand-level management of plantation forests. In Plantation Forests and Biodiversity: Oxymoron or Opportunity? (pp. 67-91). Springer, Dordrecht.

Sekercioglu, C.H., 2002. Impacts of birdwatching on human and avian communities. Environmental conservation, 29(3), pp.282-289.

Tomé, M., Oliveira, T. and Soares, P., 2006. O modelo GLOBULUS 3.0-dados e equações. Publicações do GIMREF, pp.1-26.

Uliczka, H., Angelstam, P., Roberge, J.M., 2004. Indicator species and biodiversity monitoring systems for non-industrial private forest owners: is there a communication problem? Ecol. Bull. 379e384.

Verheyen, K., Vanhellemont, M., Auge, H., Baeten, L., Baraloto, C., Barsoum, N., … SchererLorenzen, M. (2016). Contributions of a global network of tree diversity experiments to sustainable forest plantations. Ambio, 45(1), 29–41. https://doi.org/10.1007/s13280-015-0685-1

van Hensbergen, H.J. 2006. Plantaciones, sustentabilidad y certificación. Revista Ambiente y Desarollo 22: 21–28.

Zduniak, P., Bocheński, M. and Maciorowski, G., 2021. How littered are birds' of prey nests? Study of two sympatric species. Science of The Total Environment, 790, p.148079.

Zuberogoitia, I., Zabala, J., Martínez, J.A., Martínez, J.E. and Azkona, A., 2008. Effect of human activities on Egyptian vulture breeding success. Animal Conservation, 11(4), pp.313-320.

Appendices

Table A1: Multiple comparisons of means (Tukey test) of DBH indicator between all tree species in all FMMs; *indicates p is significant in the range 0.05-0.01; ** p =0.01-0.001; *** p < 0.001

Comparison of all species in all FMMs	p adjusted
E. globulus FMM1/FMM2—P. pinaster FMM1/FMM2	0.0006392**
C.sativa FMM3--P.pinaster FMM1/FMM2	0.9999997
E.globulus FMM4--P.pinaster FMM1/FMM2	0.0140972*
P.pinaster FMM5--P.pinaster FMM1//FMM2	0.8548009
Q.robur FMM6--P.pinaster FMM1/FMM2	0.0000275***
Q.suber FMM7--P.pinaster FMM1/FMM2	0.0000000***
C.sativaFMM3--E.globulus FMM1/FMM2	0.0010635**
E.globulus FMM4--E.globulus FMM1/FMM2	0.9829296
P.pinaster FMM5--E.globulus FMM1/FMM2	0.0599460
Q.robur FMM6--E.globulus FMM1/FMM2	0.9928971
Q.suber FMM7--E.globulus FMM1/FMM2	0.0000000***
E.globulus FMM4--C.sativa FMM3	0.0210614*
P.pinaster FMM5--C.sativa FMM3	0.9049499
Q.robur FMM6--C.sativa FMM3	0.0000497***
Q.suber FMM7--C.sativa FMM3	0.0000000***
P.pinaster FMM5--E.globulus FMM4	0.3697263
Q.robur FMM6--E.globulus FMM4	0.7363375
Q.suber FMM7--E.globulus FMM4	0.0000000***
Q.robur FMM6--P.pinaster FMM5	0.0065847**
Q.suber FMM7--P.pinaster FMM5	0.0000000***
Q.suber FMM7--Q.robur FMM6	0.0000000***

Table A2: Multiple comparisons of means (Tukey test) of dominant height indicator between all tree species in all FMMs; *indicates p is significant in the range 0.05-0.01; ** p =0.01-0.001; *** p < 0.001

Comparison of all species in all FMMs	p adjusted
E.globulus FMM1/FMM2--P.pinaster FMM1/FMM2	0.9850195
C.sativa FMM3--P.pinaster FMM1/FMM2	0.9998386
E.globulus FMM4--P.pinaster FMM1/FMM2	0.0032843**
P.pinaster FMM5--P.pinaster FMM1/FMM2	0.9988351
Q.robur FMM6--P.pinaster FMM1/FMM2	0.0000000***
Q.suber FMM7--P.pinaster FMM1/FMM2	0.9999997
C.sativa FMM3--E.globulus FMM1/FMM2	0.9103459
E.globulus FMM4--E.globulus FMM1/FMM2	0.0472790*
P.pinaster FMM5--E.globulus FMM1/FMM2	0.8515960
Q.robur FMM6--E.globulus FMM1/FMM2	0.0000000***
Q.suber FMM7--E.globulus FMM1/FMM2	0.9939132
E.globulus FMM4--C.sativa FMM3	0.0008016***
P.pinaster FMM5--C.sativa FMM3	0.9999992
Q.robur FMM6--C.sativa FMM3	0.0000000***
Q.suber FMM7--C.sativa FMM3	0.9990878
P.pinaster FMM5---E.globulus FMM4	0.0004342***
Q.robur FMM6--E.globulus FMM4	0.0000000***
Q.suber FMM7--E.globulus FMM4	0.0052126
Q.robur FMM6--P.pinaster FMM5	0.0000000***
Q.suber FMM7--P.pinaster FMM5	0.9959712
Q.suber FMM7--Q.robur FMM6	0.0000000***

Table A3: Multiple comparisons of means (Tukey test) of stem density indicator between all tree specie s in all FMMs; *indicates p is significant in the range 0.05-0.01; ** p =0.01-0.001; *** p < 0.001

Comparison of all species in all FMMs	p adjusted
E.globulus FMM1/FMM2/FMM4--C.sativa FMM3	0.2041598
P.pinaster FMM1/FMM2--C.sativa FMM3	0.0000000***
P.pinaster FMM5--C.sativa FMM3	0.6372728
Q.robur FMM6--C.sativa FMM3	0.9950132
Q.suber FMM7--C.sativa FMM3	0.0000000***
P.pinaster FMM1/FMM2--E.globulus FMM1/FMM2/FMM4	0.0000014***
P.pinaster FMM5--E.globulus FMM1/FMM2	0.0020461**
Q.robur FMM6--E.globulus FMM1/FMM2/FMM4	0.4985749
Q.suber FMM7--E.globulus FMM1/FMM2/FMM4	0.0000000***
P.pinaster FMM5--P.pinaster FMM1	0.0000000***
Q.robur FMM6--P.pinaster FMM1/FMM2	0.0000000***
Q.suber FMM7--P.pinaster FMM1/FMM2	0.0000000***
Q.robur FMM6--P.pinaster FMM5	0.3044139
Q.suber FMM7--P.pinaster FMM5	0.0000071***
Q.suber FMM7--Q.robur FMM6	0.0000000***

Chapter V – Final considerations

Final considerations

In this dissertation, I conducted research to address biodiversity in forest management of plantation forests of northwestern Portugal. Specific open questions to address: (1) which biodiversity indicators could be considered in forest management planning, (2) how to assess biodiversity at a stand level in plantation forests in northwestern Portugal and what is the state of biodiversity in those stands, (3) how to assess biodiversity at a landscape level in plantation forests in northwestern Portugal and what is the state of biodiversity in those landscapes.

1. Which biodiversity indicators could be considered in forest management planning?

Biodiversity indicators need to be practical to be used by both professionals and non-professionals (Ferris and Humphrey 1999; Smith et al. 2008). In this sense, practical means simple, cost and time-effective, but still ecologically meaningful. According to the results of our literature review (chapter two), forest biodiversity indicators that can be extracted from forest inventories are practical; such indicators are tree species composition, diameter heterogeneity, height, crown openness and similar biometric variables. Compositional variables such as herbal species, fungi (including mushrooms), insects, amphibia, and other taxa, are hard to recognize by non-professionals and thus are unpractical. However, some scholars (Feest 2011; Müller-Buser 2002; Mosimann 1987) argue that bird abundance and richness are more correlated with forest structural variables rather than tree species compositional ones. This is an advantageous feature of structural indicators since birds are one of the most researched taxa (Angelstam et al. 2004; Naumov et al. 2018) and are often used as biodiversity indicators (e.g., Mikusiński et al. 2001; Roberge et al. 2008; Vangansbeke et al. 2017). The main finding on functional indicators in our review was that these are not sufficiently represented in the literature, despite their important role in the assessment of forest ecosystems' sustainability.

Once decided which biodiversity indicators to use, it is necessary to collect relevant data related to these indicators. We identified in our review (chapter two) that the most convenientfor forest managers is to search existing data in national forest inventories as this is timely and cost-efficient. Also, relevant data can be efficiently found in bird atlases. We also found that remotely sensed data is considered more cost-efficient and timely than ground-based measurements (Müller and Brandl 2009; Ozdemir et al. 2018; Thers et al. 2017), particularly for larger areas of assessment. Also, remote sensing data is accurate when structural variables such are tree biomass and height are estimated (Bottalico et al. 2017). However, more research is needed to provide accurate data on other variables relevant to biodiversity estimation. Smartphone applications are promising tools for biodiversity assessments, however, more research is needed to provide better accuracy in the future.

It was beyond the scope of this review to identify indicators in forest ecosystems smaller than stand or those that relate to landscape structural features (e.g., edges, interior space, patch size, corridors). Also, the indicators in the review are not addressing genetic biodiversity. However, we suggest for future research to consider genetic biodiversity addressing in forest management planning due to great importance for biodiversity conservation.

2. How to assess and what is the state of biodiversity at a stand level in plantation forestsof northwest Portugal?

Following the findings in the systematic review from the second chapter, the starting point in assessing biodiversity at the stand level in plantation forests was to select practical indicators. Thus, we selected the indicators: tree species indicator (number of different species and species origin—depending on if it is native or exotic), mean diameter (cm) and shrub biomass (Mg ha^{-1}), as these are available in forest inventories, simple and yet ecologically relevant. Tree species are a major aspect of forests and therefore forest biodiversity (Stapanian et al. 1997), and native tree species are even more contributing to biodiversity than exotic ones (Goded et al. 2019; Proença et al. 2010). Understory layer such as shrubs, provide food and shelter to numerous forest animals (Smith et al. 2007) and has an important role in nutrient cycling and carbon storage (Botequim et al. 2015). The mean diameter is an indicator of crown development, tree height and biomass growth in managed forests, while in natural forests, a large tree diameter indicates high biodiversity (e.g., Burrascano et al. 2013; Badalamenti et al. 2017). We went further in the stand-level assessment by comparing biodiversity value in stands with shrubs reproducing by seeds and shrubs reproducing by resprouting. Shrub reproducing type is a functional attribute relevant for assessing vegetation responses to fires. Also, we considered three site quality conditions. We normalized the data of each indicator as percentages using the indicator's actual and reference values, combined values of those normalized indicators and obtained a final biodiversity score.

The highest mean values of biodiversity were estimated in mixed stands dominated with pine on the superior quality sites and fully regenerating with resprouting (29.85--low), while the lowest mean biodiversity scores were in pure blue gum stands (10.13—very low) on lowest-quality sites with shrub regeneration by seed. Site quality and shrub regeneration type significantly affected all biodiversity scores in mixed stands dominated with pine and pure chestnut stands,while less affected in pure blue gum stands and mixed stands dominated with blue gum. We found that shrubs that regenerate by resprouting develop faster than shrubs that regenerate by seeds. The results are relevant for management planning and biodiversity conservation as the fast development of shrub biomass might impose a higher risk of wildfire. However, our results are not in concordance with the study from the Portuguese Mainland (Botequim et al. 2015), which reported results opposite to ours, however from various types of forests. Similar resultsare reported in a study from central Argentina (Gurvich et al. 2005) and Australia (Pate et al. 1990). However, Pausas et al. (2004) argue that shrub functional traits can be predicted with high probability, only locally and not globally, due to different responses of species in different sites. Therefore, more research is needed to assess the effect of various shrub species resprouting categories on biodiversity in forests with various management intensities. Site quality also had an impact on mean biodiversity value in our study, and the greatest values are recorded on superior sites, as anticipated. There is a lack of studies that investigate the effect of site quality on biodiversity and therefore, future research is required to focus on such a problem.

This research did not consider deadwood as a biodiversity indicator due to the absence of data on such aspects in our case study. Therefore, comparing our results with other studies that considered it when addressing biodiversity might be difficult. Since deadwood is very important for forest biodiversity, considering leaving some deadwood in set-aside areas along with sparing some trees from a cut in our study area, could compensate for the negative effect of intensive silviculture measures on biodiversity (e.g., Lafond et al. 2015). Some incentives provided by the government or the EU funds could compensate for timber losses as a trade-off with biodiversity management. Introducing native species well adapted to fire, such as cork oak, might benefit biodiversity but also fire prevention which would also compensate for timber lossesdue to fire hazards. Regarding the exotic eucalypt plantations that dominate in our study area,there is evidence that exotic plantations can host native flora and fauna and thus contribute to biodiversity conservation (Koh and Gardner 2010). However, restoring native forests would bemore beneficial for biodiversity conservation than managing eucalypt forests for biodiversity (Calviño-Cancela et al. 2012).

3. How to use stand-level indicators to assess landscape biodiversity aspects in plantation forests in northwest Portugal?

We selected the red kite as a case study species and attempted to assess the potential for conserving the red kite in NW Portugal. We selected the red kite since it is an "umbrella" and "flagship" species dependent on large territories such are landscapes. The study area assembles four forest management models that are typically part of plantation forests in NW Portugal, as well as three other models that may be considered for that purpose. We focused on suitability for nesting, since these raptors often use plantation trees during breeding. For this purpose, we used the indicators that represent the breeding habitat suitability of the raptor: tree height (m), mean DBH (cm), tree density (number of trees^{-ha}) and frequency of clear-cut.

Results showed that pure blue gum plantations are not favourable to red kite conservation, mainly because of high stem density and low DBH, which negatively affect habitat suitability for the red kite. The major problem here is the short rotation period of blue gum (11 years) and therefore, DBH cannot develop enough. However, if rotation would be extended or some trees set aside, blue gum could be suitable for red kite nesting. Mature blue gum plantations (>25 years of age) with low management (no pruning and thinning), can host many common species that grow in maritime pine plantations and native oak forests (Calviño-Cancela et al. 2012). Mixed forest plantations are better suited for red kite species because of longer rotation periods of the maritime pine, although a higher frequency of silvicultural practices also negatively affects habitat suitability. Chestnut woodlands had the same overall suitability as mixed stands for red kite suitability, however, in the last decade before clear-cut the suitability was high. Therefore, suitability for red kite in this study was very low and low in the young stages, while it was high and very high in the mature stage (>40 years). This is in contrast with a previous study that assessed chestnut woodland biodiversity in NW Spain of various ages and levels of abandonment and reported that the degree of maturity of the woodland only affects the richness of plants, and not the richness of birds, beetles and ants, however, birds' richness was related to woodland size (Guitián et al. 2012).

Regarding alternative models, pure pine plantations are more suitable and marked medium overall suitability mainly due to only half of the stem density of the current models, even though the rotation is shorter. Low density in pine plantations is also linked to high biodiversity in a study from Spain (Calviño-Cancela et al. 2012). Common oak plantations showed unexpected results and seem the least favourable of all models due to slow growth and height that never reaches the optimal values for the kite to nest. However, native common oak forests are very diverse ecosystems and are very important for biodiversity conservation (e.g., Proença et al. 2010; Calviño-Cancela et al. 2012; Goded et al. 2019). Thus, keeping these trees in the landscape would have great importance for the native biodiversity. Also, common oak would become suitable for red kites in planning horizon longer than 90 years, since it is a slow-growing species. The most favourable of all current and alternative models are older cork oak (>30 years of age) woodlands due to low stem density, sufficient height for nesting, and absence ofclear cuts as cork is harvested without tree felling. This was an expected result since cork oakforest systems are known for their high conservation value for biodiversity (Bugalho et al. 2011).

Overall, the landscape of plantations of species with a longer rotation period, or set-aside trees would benefit red kite conservation. The longer rotation should be one of the goals in forest management planning as it benefits forest specialist species and biodiversity conservation in general (Oxbrough et al. 2006; Jukes et al. 2001). Apart from clear-cut, other silvicultural activities such are fuel treatment and thinning, can also disturb birds during egg-laying and birding. Therefore, silvicultural activities should be performed outside of egg-lying and fledging season. Diversifying landscapes with different tree species is possible, however by combining species with similar silvicultural activities to decrease disturbance. Mosaics of open areas andplantations in the landscape would also benefit red kite conservation and overall biodiversity. This is in concordance with Borges and Hoganson (2000) who emphasized the need to acknowledge the relationship between the forested landscape spatial structure and its ecological characteristics when developing multifunctional forest management planning. It is in concordance further with Hunter (1990) who emphasized that biodiversity in a forested landscape would be best preserved in a land mosaic characterized by a diverse array of stands.

The indicators we applied here could be useful for the conservation of other raptors with similar habitat requirements as the red kite. However, the focus of this study is on suitability for red kite nesting and doesn’t include general suitability. Therefore, future studies can investigate the suitability for red kite hunting and roosting in the landscape. Also, there is a lack of studies that included a DBH of red kite nesting trees. We used the proxies for DBH from otherstudies that researched raptors nests with similar size as the red kite, however, it should be further investigated and confirmed what is the optimal DBH for red kite nests.

4. Original contributions to science

The review (Chapter II) is the first paper to review the practicality, namely efficiency, ease and efficacy, of biodiversity indicators. As such, it is a good initial point to deal with biodiversity assessment. Hence, if biodiversity indicators are not practical, that would affect the assessment and therefore biodiversity conservation. Hence, such a review was very much needed to clarify the gaps and point at impractical indicators and indicators that can be readily used by forest managers with various backgrounds.

The research paper (Chapter III) is intended for the stand-level biodiversity assessment. Here, our original contribution (a novelty in research) is firstly by following the results of the second chapter and thus choosing practical biodiversity indicators that can be extracted from forest inventories. Secondly, for the tree species indicator, instead of just numbering the tree species, we created a score that gives value to each tree species according to its richness and its contribution to biodiversity. Namely, after conducting the literature review, we found out that native tree species host much larger biodiversity than the exotic ones in Portugal; meaning that there are numerous insects, birds, mammals and other taxa, that depend on certain tree species, and there more species dependant on native species than the exotic. Further, we included site indexes (indexes estimating site productivity) in the evaluation of biodiversity and estimated if biodiversity is affected by different site indexes. To our knowledge, such an aspect has not been included in biodiversity assessment in forestry studies so far. Finally, we estimated the impact of shrub regeneration type on plantation biodiversity, to our knowledge, that is a novelty in research.

Regarding the third research article (Chapter IV), we selected the indicators for raptor species nesting that are also extractable from forest inventories, and even yield tables. These indicators coincide with the indicators for fire prevention, which is crucial in Portuguese forestry. Also, the indicators are important for overall biodiversity.

References:

Angelstam, Per, D Monika, and Jean-michel Roberge. 2004. “Oikos Editorial Office Targets and Tools for the Maintenance of ForestBiodiversity: An Introduction Author (s): Per Angelstam, Monika Dönz-Breuss and Jean-Michel Roberge Published by: Oikos Editorial Office Linked References Are Available on JSTOR F.” (51): 11–24.

Badalamenti E, La Mantia T, La Mantia G, Cairone A, La Mela Veca DS. 2017. Living and dead aboveground biomass in Mediterranean forests: Evidence of old-growth traits in a *Quercus pubescens* Willd. sl stand. Forests 8(6):187.

Borges, J. G. and H. M. Hoganson. 2000. Structuring a landscape by forestland classification and harvest scheduling spatial constraints. Forest Ecology and Management 130: 269-275

Bottalico, Francesca et al. 2017. “Modeling Mediterranean Forest Structure Using Airborne Laser Scanning Data.” *International Journal of Applied Earth Observation and Geoinformation* 57(May): 145–53. http://dx.doi.org/10.1016/j.jag.2016.12.013.

Botequim B, Zubizarreta-Gerendiain A, Garcia-Gonzalo J, Silva A, Marques S, Fernandes PM, Pereira JMC, Tomé M. 2015. A model of shrub biomass accumulation as a tool tosupport management of Portuguese forests iForest 8: 114-125 -doi: 103832/ifor0931- 008

Bugalho MN, Caldeira MC, Pereira JS, Aronson JA, Pausas J. 2011. Mediterranean oak savannas require human use to sustain biodiversity and ecosystem services. Frontiers in Ecology and the Environment 5: 278-286 (doi:10.1890/100084).

Burrascano S, Keeton WS, Sabatini FM, Blasi C. 2013. Commonality and variability in the structural attributes of moist temperate old-growth forests: A global review. For Ecol Manag 291: 458–479.

Calviño-Cancela M, Rubido-Bará M, van Etten EJ. 2012. Do eucalypt plantations provide habitat for native forest biodiversity? Forest Ecology and Management. Apr 15;270:153-62.

Feest, Alan, Chris Van Swaay, Timothy D. Aldred, and Katrin Jedamzik. 2011. “The Biodiversity Quality of Butterfly Sites: A Metadata Assessment.” *Ecological Indicators* 11(2): 669–75. http://dx.doi.org/10.1016/j.ecolind.2010.09.006.

Goded, Sandra et al. 2019. “Effects of Eucalyptus Plantations on Avian and Herb Species Richness and Composition in North-West Spain.” *Global Ecology and Conservation* 19.

Guitián J, Guitián P, Munilla I, Guitián J, Garrido J, Penín L, Domínguez P, Guitián L. 2012. Biodiversity in Chestnut Woodlots: Management Regimen vs Woodlot Size. OpenJournal of Forestry. Oct 30;2(04):200.

Gurvich DE, Enrico L, Cingolani AM. 2005. Linking plant functional traits with post-fire sprouting vigour in woody species in central Argentina. Austral Ecology.30(8):868-75.

Hunter, M.L., 1990. Wildlife, Forests and Forestry. Principles of Managing Forests for Biological Diversity. Prentice-Hall, New Jersey, 370 p.

Jukes, M.R., Peace, A.J., Ferris, R., 2001. Carabid beetle communities associated with coniferous plantations in Britain: the influence of site, ground vegetation and stand structure. For. Ecol. Manage. 148, 271–286

Koh LP, Gardner TA. 2010 Conservation in human-modified landscapes. In: Sodhi NS,Ehrlich PR (Eds) Conservation biology for all Oxford University Press, Oxford, pp236–261

Lafond V, Cordonnier T, Courbaud B. 2015. Reconciling Biodiversity Conservation and Timber Production in Mixed Uneven-Aged Mountain Forests: Identification of Ecological Intensification Pathways. Environmental Management 56(5): 1118–33

Mikusiński, Grzegorz, Maciej Gromadzki, and Przemyslaw Chylarecki. 2001. "Woodpeckers as Indicators of Forest Bird Diversity." *Conservation Biology* 15(1): 208–17.

Mosimann, P.; Naef-Daenzer, B.; Blattner, M. 1987. Die Zusammensetzung der Avifauna in typischen, Waldgesellschaften der Schweiz. D. Ornitholog. Beobachter, 84,275–299

Müller, Jörg, and Roland Brandl. 2009. "Assessing Biodiversity by Remote Sensing in Mountainous Terrain: The Potential of LiDAR to Predict Forest Beetle Assemblages." *Journal of Applied Ecology* 46(4): 897–905.

Naumov, V. et al. 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.

Oxbrough, Anne G. et al. 2006. "The Influence of Open Space on Ground-Dwelling Spider Assemblages within Plantation Forests." *Forest Ecology and Management* 237(1–3): 404–17.

Ozdemir, Ibrahim et al. 2018. "Predicting Bird Species Richness and Micro-Habitat Diversity Using Satellite Data." *Forest Ecology and Management* 424(May): 483–93. https://doi.org/10.1016/j.foreco.2018.05.030.

Pate JS, Froend R H, Bowen BJ, Hansen A, and Kuo J. 1990. Seedling growth and storage characteristics of seeder and resprouter species of Mediterranean-type ecosystems of SW Australia. Annals of Botany 65:585–601.

Pausas JG, Bradstock RA, Keith DA, Keeley JE. 2004. Plant functional traits in relation to fire in crown-fire ecosystems. Ecology. 85(4):1085-100.

Proença, Vânia M., Henrique M. Pereira, João Guilherme, and Luís Vicente. 2010. "Plant and Bird Diversity in Natural Forests and in Native and Exotic Plantations in NW Portugal." *Acta Oecologica* 36(2): 219–26.

Roberge, Jean Michel, Per Angelstam, and Marc André Villard. 2008. "Specialised Woodpeckers and Naturalness in Hemiboreal Forests - Deriving Quantitative Targets for Conservation Planning." *Biological Conservation* 141(4): 997–1012.

Smith, George F. et al. 2008. "Identifying Practical Indicators of Biodiversity for Stand-Level Management of Plantation Forests." *Biodiversity and Conservation* 17(5): 991–1015.

Stapanian, MA, Cassel, DL, Cline, SP, 1997. Regional patterns of local diversity of trees: associations with anthropogenic disturbance. For Ecol Manag 93: 33–44

Thers, Henrik et al. 2017. "Lidar-Derived Variables as a Proxy for Fungal Species Richness and Composition in Temperate Northern Europe." *Remote Sensing of Environment*

200(January): 102–13. http://dx.doi.org/10.1016/j.rse.2017.08.011.

Vangansbeke, P. et al. 2017. “Spatially Combining Wood Production and Recreation with Biodiversity Conservation.” *Biodiversity and Conservation* 26(13): 3213–39.