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Assessing European forest biodiversity: a multi-dimensional approach

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Abstract

Assessing forest diversity is still a critical task at local, national and global level. Monitor biodiversity nationwide is required for countries participating to the Convention of Biological Diversity, however, measuring forest diversity is a challenge due to the complexity and the functionality of forest and to the difficult of harmonizing sampling design at continental scale.

The present research is aimed to analyze a large forest diversity data to assess forest diversity at the pan-continental level, to investigate how the tree layer diversity can be used as a proxy of the community diversity at different scale levels, and to demonstrate the importance of forest monitoring for forest management.

We used a multi-dimensional pan-European data acquired in the framework of BioSoil project (Li-BioDiv database), as part of ICP forest Level I network. The database contains forest structure and vegetation data collected in 19 European countries in the period 2005-2008. In addition, for one of the purpose we used also local-scale diversity data as well. The dissertation is divided into two main parts. The first one includes paper I and II that deepen data stored in the LI-BioDiv database, while the second one includes paper III and IV investigating the relationship between tree and ground vegetation data.

The paper I presents the LI-BioDiv database and compares plot level forest variable with data of National Forest Inventories and demonstrates that the database appears useful mainly for research purpose aimed at studying cross-relationships between multiple forest variables.

Paper II is aimed of evaluating the uncertainty of ground vegetation data related to forest diversity descriptors using a diffusion-based cartogram approach that visually displays how data information change in function to different uncertainty degrees.

The study highlights than an awareness of the negative relationship between the period of the survey and species richness can lead to a better data handling and analysis. In addition it also demonstrate that cartograms are efficient tools for evaluating and managing uncertainty and can strengthen the results of data analysis by providing alternative perspectives and interpretations of spatial phenomena.

In Paper III, we explore the relationship between structural diversity indicators and floristic diversity indices, across bioregions, in different management conditions and in different forest types at pan-European scale. As expected, no relationship was found between the two group of indices. The results confirm that structural diversity indices are complementary to floristic biodiversity indices. A selection of groups of different types of indices is highly encouraged to better explain forest biodiversity.

In Paper IV stand structure and plant diversity changes were tested at local scale. The effects of alternative managements were investigated by comparing the structure and plant diversity along a four-stages temporal gradient of stand development under similar site conditions. The study underlines species turnover along a temporal gradient in order to promote conservation-oriented management. The results underline that the frequency of silvicultural interventions seems to influence the turnover in species composition and structural condition variation.

In conclusion, understanding heterogeneity related to biological data is a complex task that can lead to good data use and analysis. The relationship between tree layer and ground vegetation layer change across scales and local conditions. National Forest inventories should be taken into consideration as a platform to include multiple forest variable in order to properly monitoring and assessing forest biodiversity status and trends, and to promote a targeted management.

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Chapter 1

Introduction

1.1 Background

1.1.1 Forest monitoring

Forests are fundamental in environmental functions, human wellbeing, for fighting climate change, conserving biological diversity, protecting soils and preserving water resources (MEA, 2005; FOREST EUROPE, 2015). Understanding forest resources and their changes is required by several international agreements, including the United Nations Framework Convention on Climate Change (UNFCCC), the Convention on Biological Diversity (CBD) and the UN Forest Instrument and the Sustainable Development Goals (SDGs) (MCPFE, 2002; FAO, 2017). National Forest Inventories (NFIs) are the primary source of data on the status of forest (Alberdi et al., 2010; Chirici et al., 2011). Traditionally, NFIs have carried out data collections to conduct forest resource assessment, to assess Sustainable Forest Management (SFM) and to provide a basis for proper planning and to optimize forest management (Chirici et al., 2011; FAO, 2017). NFIs do not assess a common set of variables and do not use common sampling design or measurement protocols among countries. These disparities contribute to the lack of comparability of data at international level (Chirici et al., 2011). NFIs are in fact designed to support forest-related decision at national level. In the last 20 years forest managment shifted from wood production objective to wider purposes. As consequence NFIs also started to acquire larger set of variables moving from traditional to multipurpose inventories. However, data collection on species diversity including ground vegetation is limited in a few countries and it has been implemented in different ways, with consequent limitation in harmonization possibilities. (Chirici et al., 2012).

The BioSoil project, developed as part of the International Cooperative Programme on Assessment and Monitoring of Air pollution effects on Forests (ICP Forests; `www.icp-forests.org`), was designed to evaluate if ICP Level I plots may be used for a large-scale European study providing harmonized soil and forest biodiversity data (Hiederer and Durrant, 2010). The BioSoil project approach was focused on stand structure, deadwood, and ground vegetation information to increase the knowledge of European forest complexity and its ecological significance for forest biodiversity. Specifically, the objectives of the project were to contribute to: i) issues related to a better comprehension of forest ecology, and ii) supporting forest policies, including sustainable forest management (Hiederer and Durrant, 2010). Field measurements were collected in 3311 plots across Europe from 2005 to 2008, concerning tree diameters, status and species of all woody plants taller than 1.30 m, top height of at least three largest trees, deadwood, vascular species list of ground vegetation and assessment of canopy closure and tree layering. All the data are strored in the Li-BioDiv database. The LI-BioDiv dataset is an example of a possible “new approach” for a pan-European forest monitoring approach for trends assessment at European scale since it provides a potential basis to support a permanent multi-dimensional monitoring program of forest biodiversity.

Forests are complex system (Levin, 2005; Messier et al., 2013) and their monitoring requires approaches that reflect this (FAO, 2017). Preserving biodiversity, mitigating climate change, safeguarding the ecosystem functions are the multiple goals of sustainable forest management. The relevance to up-to-date multiple forest information has been increasing in recent years (FAO, 2017). Comparability and consistency are key elements to provide timely and reliable forest information at different scales (FAO, 2017). To integrate conservation purpose in forest management, monitoring systems should be planned to be able to respond to a dynamic contest which can change depending on the scale and involved stakeholders.

In this context, a National Forest Monitoring System (NFMS) is a new concept of forest assessment into the future. In particular, the NFMS, proposed by FAO (2017), represents an institutional challenge and specifically a process of gathering, collecting, analyzing information from diverse source to assess forest at national and international level. The aim is to build a comprehensive database helpful containing multiple information for decision-making basing on scientific ground. FAO (2017), provides guidelines to optimize inventories, statistical modeling and estimation and also contributes to developing basic standards (i.e. terminology) to facilitate data comparison in space and time. FAO has provided both institutional and scientific challenge basing on lack of a comprehensive national monitor system design.

1.1.2 Data quality

Data quality is closely linked to policy implications and quality decision. Rigorous vegetation science and efficient environmental and forest planning require careful assessment and management of data uncertainty (Galluzzi et al., 2018). Multiple sources can lead to data uncertainty starting from differences among observers (Hall and Okali, 1978; Mason et al., 2015), non-random sampling (Lepš and Hadincová, 1992), and the dynamic nature of species distributions (Rocchini et al., 2011).

Understanding data uncertainty and quality constitute a powerful tool to avoid biased ecological inferences (Meyer et al., 2016; FAO, 2017). The first step to understand how data can be managed is to explicitly describe and quantify the uncertainty of data themselves (Galluzzi et al., 2018). However, uncertainty information is often not shown explicitly (Mason et al., 2015), although it has been recently demonstrated that it is possible to reduce the negative effects of data coming from different sources by considering different levels of uncertainty (Galluzzi et al., 2018). It is difficult to reduce uncertainty related to observer but it is possible to reduce uncertainty due to the adoption of different monitoring procedures.

Different challenges and objectives characterize biodiversity monitoring at different scales (Chirici et al., 2011). This is due to the variability of forest ecosystems and to the different information required by forest managers and forest ecologists. In this context, NFIs data have been assessed inside an international cooperation initiated by COST Action E43 with the aim of understanding how data coming from different countries with different data collection can be harmonized and used as a complete source of forest information by Chirici et al. (2011). Data harmonization is possible albeit with limitations and common variables relevant for monitoring community biodiversity, should be considered and implemented in a standard field protocol (Chirici et al., 2011).

1.1.3 Forest biodiversity evaluation

Forest biological diversity is a multi-dimensional concept that spans different levels of ecosystems, landscapes, species, populations, and genetics (CBD, 2007). The interaction between and within these levels contributes to forest complexity and allows organisms to adapt to continually changing environmental conditions and to maintain ecosystem functions (CBD, 2007).

For several years, tree species composition and structure have been considered as information for a biodiversity index (MCPFE, 2002; Barbier et al., 2008), and the importance of tree structure to preserve forest diversity and complexity is recognized at international level (Chirici et al., 2011; MCPFE, 2011), despite the higher species richness of herbaceous layer (Gilliam, 2007). Structural diversity is the most used indicator to study forest biodiversity (Winter et al., 2008; Chirici et al., 2011; Bottalico et al., 2017) and the most used indicators are generally based on common forest inventory variables: tree height and DBH (Corona et al., 2011; Bottalico et al., 2017). Ground vegetation has a species richness usually higher than other forest strata, yet, the survey of herb layer is often omitted in traditional forest inventory protocols (Gilliam, 2007). Traditionally NFIs have been designed to assess land coverage and the production value of forests rather than forest biodiversity, however in the last decades, in some countries, ground layer vegetation was included in the field survey albeit in different ways (see Chirici et al. (2012) for a review). Overstory and understory are linked by reciprocal interaction (Gilliam and Roberts, 2003). The overstory composition influences the dynamic of understory by altering light regime and influencing the heterogeneity of soil fertility (Gilliam, 2007). Conversely, the understory community can influence overstory seedling dynamics and overstory composition (Gilliam, 2007). Several studies focused on the relationship between overstory and understory layer with results that range widely (Burrascano et al., 2011) and Barbier et al. (2008), reviewed the influence of tree species composition on understory vegetation diversity, pointed out the difficulties to make generalization due to the conflicting results. Furthermore, structural diversity is not the only factor influencing understory composition which is in fact other environmental parameters such as soil and pH (Lenière and Houle, 2006). Understory diversity sometimes could be more closely related to silviculture operations (Barbier et al., 2008), sometimes to environmental conditions (Lenière and Houle, 2006) and sometimes to tree species. The use of overstory diversity indices as descriptors of forest diversity explains only part of the status and trends of forests.

The need to expand the variables surveyed in the NFIs for a comprehensive biodiversity monitoring, was considered only in the few years (CBD, 2007; Chirici et al., 2011). The initiative requested by FAO about the integration of biodiversity across agricultural sectors at national, regional and international level have been endorsed in the 40th Session of the FAO Conference in 2017 (FAO, 2018). The need to mainstream biodiversity across relevant policies, plans, and programs for ensuring the effective implementation of the Convention on Biological Diversity (the Strategic Plan for Biodiversity 2011-2020), has been now recognized at international level (FAO, 2018). Interconnection and an integrated vision and approach between agriculture, forestry, and fisheries sectors are now required (FAO, 2018). In this regard, the scientific community can provide a basis for new opportunity to integrate biodiversity in a management context. However, multi-scale evidence-based about the interconnection of biodiversity across the different component of ecosystem still need to be investigated.

1.2 List of papers

This dissertation is based on the following four articles, which are referred to by their Roman numerals in the text. The articles are reprinted with the kind permission of the publishers.

Paper I

Galluzzi M., Giannetti F., Puletti N., Canullo R., Rocchini D., Bastrup-Birk A., Chirici G., 2018. A plot-level exploratory analysis of European forests based on the results from the BioSoil project. Submitted to *European Journal of forest research.*

Paper II

Galluzzi M., Rocchini D., Canullo R., Mc Roberts R.E., Chirici G., 2018. Mapping uncertainty of ICP-Forest biodiversity data: from standard treatment of diffusion to density-equalizing cartograms. *Ecological Informatic.* `https://doi.org/10.1016/j.ecoinf.2018.06.005`.

Paper III
Galluzzi M., Rocchini D., Canullo R., Nocentini S., Chirici G., 2018. Linking overstory and understory diversity indicators through different European forest diversity pattern. Submitted to *Ecological indicators*.

Paper IV
Galluzzi M., Selvi F., Carrari E., Chirici G., Paffetti D., Nocentini S., Travaglini D., 2018. Temporal and dynamic plant diversity change in thermophilous deciduous forests. *Draft.*

1.3 Aims

Analyzing, reporting forests monitoring-related information is an opportunity for researchers to support the claim of long-term ecological monitoring. Provide evidence-based results and uncertainty information is a way to strengthen forest monitoring programs and to better focus future effort. Furthermore, forest monitoring supports forest management, encourages collaborative efforts between scientific, institutional and private sectors. Biodiversity characterization concern different components at several levels of organization and identify measurable indicators that can explain the overall status of biodiversity. The driving force behind the present dissertation, was to respond to these needs. The aims of the papers included in this dissertation were as follows:

- To provide knoledge about complete data on European forest deepening data stored in the Li-BioDiv database (Paper I);
- To offer a strategic key to integrate multiple information of forest component and to develop a method to assess data uncertainty (Paper I and II);

- To investigate relationships between the tree layer diversity and vegetation community diversity at European scale and at local scale (Paper III and IV) and to demonstrate how forest monitoring can support forest management at local scale (Paper IV).

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Chapter 2

Paper I - A plot-level exploratory analysis of European forest based on the results from the BioSoil project

Marta Galluzzi[1] *, Francesca Giannetti[1], Nicola Puletti[2], Roberto Canullo[3], Duccio Rocchini[4], Annemarie Bastrup-Birk[5], Gherardo Chirici[1]

Abstract

The lack of multi-dimensional data is one of the major gaps which limit the knowledge and the assessment possibilities of European forest. Nowadays, the most extensive and complete data on the European forest status are given by National Forest Inventories (NFIs) which provide information about the extent of forest resources and their composition and structure. Traditionally NFIs collect data related to trees, with a limited consideration of other habitat components, such as ground vegetation. This non-trees information are instead essential for a more complete forest

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biodiversity assessment. This paper is aimed at introducing the ICP Forests LI-BioDiv database which resulted from BIOSOIL-Biodiversity, a large collaborative European project. The database is organized as a multi-dimensional forest geodatabase that contains forest structure and vegetation records collected in 19 European countries in the period 2005-2008. The data were acquired in 3311 geocoded plots gathering different types of data: stand level general information, tree level data, deadwood, canopy closure and floristic composition. The paper is structured in order to: i) give a clear overview of the raw data available in the database, and to ii) present a first tentative elaboration of raw data to calculate simple plot level forest variables (biomass, deadwood volume, alpha diversity). On the basis of the results we achieved, the LI-BioDiv database appears useful mainly for research purposes aimed at studying cross relationships between multiple forest variables and not for an operative use for monitoring and assessing European forest. In particular we hope that this contribution can stimulate scientists to carry out cross analysis of the database for defining future forest biodiversity indicators that could be introduced in the field protocols of the NFIs in Europe.

Keywords: *ICP Forests; BioSoil project; European forest; Data structure; Ground vegetation; Biodiversity.*

2.1 Introduction

2.1.1 European forest monitoring

Since many years, Hunter (1990) has recognized the multi-dimensional character of forest biodiversity, that no longer considers the trees only, but also all other living organisms. The importance of forest biodiversity was addressed in several international agreements: the 1992 Convention on Biological Diversity (CBD) of the United Nations Conference on Environment and Development, the Ministerial Conference on the Protection of Forests of Europe (MCPFE, 2002) and the Montréal Process (2006). In 2003, as part of the Convention on Biological Diversity (1992), world leaders committed to achieve by 2010 a significant reduction of biodiversity loss and the Ministerial Conference on the Protection of Forests in Europe (MCPFE) has addressed all dimensions of sustainable forest management (SFM) in the pan-European region, including the protection of biodiversity (MCPFE, 2002).However the absence of threshold to define trend toward SFM indicators is still a weakness. Despite the rise of the scientific basis and the efforts made at global scale, forests are becoming progressively more simplified and fragmented, leading to biodiversity loss (Noss, 1990; Foley et al., 2005). And in 2010 at European scale the biodiversity loss did not diminished (Butchart et al., 2010). Nowadays, the most extensive and complete data on the European forest status are given by National Forest Inventories (NFIs) which provide information about the extent of forest resources (Chirici et al., 2011). This information concerns trees but usually marginally consider other ecosystem components such as ground vegetation, shrubs, animals, habitat trees, albeit a first step to integrate standardized tree related microhabitat survey into NFIs was recently done by Larrieu et al. (2018). In Europe, a part the NFIs, other networks for forest monitoring exist. The Long-Term Ecosystem Research (LTER) network through a multiscale monitoring system is aimed, since 2003, at improving our knowledge about the structure and functions of ecosystems and their long-term response to environmental, societal and economic drivers (`http://www.lter-europe.net`). FunDivEUROPE, born in 2010, is a research project funded in the 7th Framework Program to quantify the role of forest biodiversity for ecosystem functioning and the delivery of goods and services in major European forest types (`www.fundiveurope.eu`) and The Global Forest Biodiversity Initiative is a compilation of data acquired in the field in 777,126 permanent plots in 44 countries (Liang et al., 2016). The compilation of an exhaustive list of international forest monitoring networks developed for long term activities or just for the duration of specific research projects is out of the scopes of this paper, but for a recent review we refer to Danielewska et al. (2013). Despite European political efforts on halting biodiversity loss, a monitoring system to assess the role of forests in reducing the impact of climate change and providing ecosystem services still does not exist (FAO, 2017). More information need to be acquired, within the framework of NFIs or not, for a more complete assessment of forest ecosystems functionality and biodiversity (Mura et al., 2016). The demand for reliable national forestry data at continental level has grown considerably in recent year (FAO, 2017). European countries should urgently establish a national forest monitoring system in order to provide timely and reliable forest information at multiple scales. In this regards, FAO (2017) has defined guidelines for planning and implementing multi-purpose national forest monitoring systems with the aim to provide a tool to strengthen sustainable forest management at local and global scale. Constitutes National Forest Monitoring System is the new institutional challenge (FAO, 2017) that highlight the importance of data comparability taking into account variation in term of biophysical conditions rather than between countries. Only recently it seems a willingness to define forest monitoring systems that go beyond the countries boundaries and that can provide timely reliable forest information.

2.1.2 ICP Forests network and the BioSoil project

The International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (hereafter ICP Forests) was established in 1985 under the UNECE Convention on Long-range Transboundary Air Pollution (CLRTAP) in order to provide a plots network and a platform to collect and exchange information on European forest (UN-ECE, 1994, 1998; ICP, 2006). The ICP Forests network was structured in two different levels of monitoring: Level I or large scale extensive monitoring, and Level II or intensive monitoring, and for further information, we refer to the basic design principles for the ICP Forests Monitoring (Ferretti et al., 2010) and the information available at `www.icp-forests.org`. As part of ICP Forests, the BioSoil project was designed to evaluate if ICP Level I plots may be used for a large-scale European study providing harmonized soil and biodiversity data (the so called BioSoil project was related to the biodiversity component), thus contributing to: i) issues related to a better comprehension of forest ecology, and ii) supporting forest policies, including sustainable forest management (Hiederer and Durrant, 2010). The BioSoil project approach has focused on stand structure, deadwood, and ground vegetation information to increase the knowledge of European forest complexity and its ecological significance for forest biodiversity. The ICP level I plots used in the BioSoil – Biodiversity project are approximately 5700. Field measurements concern tree diameters, status and species of all woody plants taller than 1.30 m, top height of at least three largest trees, coarse woody debris, vascular species list of ground vegetation and assessment of canopy closure and tree layering. The original BioSoil dataset was unavailable even for research activities due to data policy restrictions. The need of the Level I data measured in the BioSoil project was recognized by ICP Forests during the Joint Expert Panel Meeting on European Level Data Evaluation (Helsinki, FI; 28th Task Force Meeting, Białowieża, PL, 2012) (Canullo, 2016). So, the ICP Forests and the Expert Panel on Biodiversity and Ground Vegetation supported by Camerino University asked the countries involved to voluntary re-submit the data to incorporate into a collaborative ICP Forests dataset named as LI-BioDiv dataset (Canullo, 2016). Notwithstanding the BioSoil project represents a unique pan-European effort for a standardized multi-dimensional forest biological diversity monitoring survey, the LI-BioDiv dataset is still under-utilized by scientists and the data are available by a request form at `www.icp-forests.org` (albeit they are labelled as "BioSoil/BioDiv" data) (Canullo, 2016). As far as we know, despite technical reports such as Hiederer and Durrant (2010), Canullo (2016) and the analysis of the deadwood component recently reported by Puletti et al. (2017), no complete and exaustive exploratory analysis of the dataset was carried out until now. The aim of this paper is to present the LI-BioDiv dataset aiming at: i) providing a first complete overview of the data available for the different variables and the different involved Countries which is an essential pre-requisite to evaluate the consistency and thus the scientific added value of the database; ii) presenting the biomass of living trees, the deadwood volume and the alpha diversity since they represent potential candidates as forest biodiversity indicators following for FOREST EUROPE (2015) and EEA (2010). To do so we structured this contribution in six sections. After this introduction, Section 2 describes in detail the data gathering process, introducing the sampling methodology of ICP Forests Level I, the type of data available and how they have been acquired in the field. Section 3 provides a dataset overview, by describing the structure and spatial distribution of data including diameter at breast height (DBH), deadwood (DWD), canopy closure (CAN), ground vegetation (GVG). Section 4 presents a first analysis of indicators calculated on the basis of raw data related to European forests. Section 5 and 6 include discussion and final remarks about the comparison between LI-BioDiv dataset with other monitoring networks, the relevance of forest monitoring programs and the potentiality of LI-BioDiv dataset and its future development. LI-BioDiv dataset aimed at demonstrating a possible “new approach” for a pan-European forest monitoring system for trends assessment at European scale, since it provides a potential basis to support a permanent multi-dimensional monitoring program of forest biodiversity.

2.2 Materials and methods

In this section, we included a description of the LI-BioDiv database and the methods used for the calculation of structural and compositional forest biodiversity indicators. For a detailed description on the methods used for the field data collection we refer to the BioSoil field manual (Aamlid et al., 2007; WGFB, 2011). It is important to note that here below we present the database as it is, with the data really available following the point of view of a potential user of the LI-BioDiv database. This may sometimes differ from the description available in the field manuals because crews used different protocol options resulting in slightly differences in the database from country to country.

2.2.1 Sampling design

Level I network is made of point locations systematically placed on a 16x16 km grid. The starting coordinate and the orientation of the sampling grid were individually decided by ICP participating country teams (Travaglini et al., 2013). In the BioSoil – Biodiversity project a circular plot with inner nested subplots was created around a subset of Level I point locations. The selection of the ICP level I plots to be included in the BioSoil Biodiversity project was carried out by the Countries, Belgium, Czech Rep., Finland, Latvia, Lithuania and Spain subjectively selected the points in order to homogeneously cover the area, while UK set up an entirely new random network (always 16X16km grid) specifically for the project as reported by Hiederer and Durrant (2010) (Tab. 2.1).

Around the selected ICP level I points, a plot having a radius of 25.24 m (2000m^2) with two concentric subplots was created in Biosoil - Biodiversity: subplot 1 has a radius of 3.09 m (30 m^2), and subplot 2 with a radius of 11.28 m (400 m^2) (Fig. 2.1). Furthermore, four squared sampling units (A, B, C, D, 10x10 each) were optionally installed for specific surveys (WGFB, 2011). The

Country	ICP Level I plots (*)	Li-BioDiv plots	%
Austria	136	135	99
Belgium	29	10	34
Cyprus	15	15	100
Czech Republic	205	146	71
Denmark	25	26	>100
Estonia	97	96	99
Finland	931	630	68
France	553	548	99
Germany	451	425	94
Hungary	74	78	>100
Ireland	32	36	>100
Italy	265	239	90
Latvia	207	95	46
Lithuania	82	62	76
Slovak Republic	111	112	>100
Slovenia	45	45	100
Spain	620	272	44
Sweden	790	795	>100
United Kingdom	89	167	>100

Table 2.1: Percentage of Li-BioDiv plots choosen from ICP Level I network. United Kingdom set up a new network for the BioSoil project. (*) Number of ICP level I plots were derived from Lorenz et al. (2005); Michel and Seidling (2016, 2017) as the maximun number of plots surveyed between 1992 and 2016.

combination of A, B, C and D sampling area is the same of the two subplots 2 (400 m^2) (Fig. 2.1). The LI-BioDiv dataset consists of 3311 plots georeferenced in ETRS89 Lambert Azimuthal Equal Area Coordinate Reference System (ETRS-LAEA) (Fig. 2.2). The plots were surveyed in the field between 2005 and 2008.

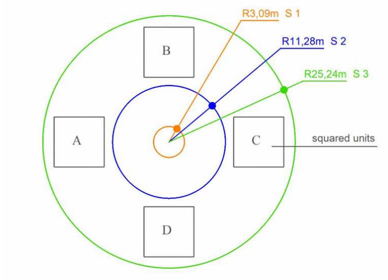


Figure 2.1: The BioSoil plot. WGFB (2011), modified.

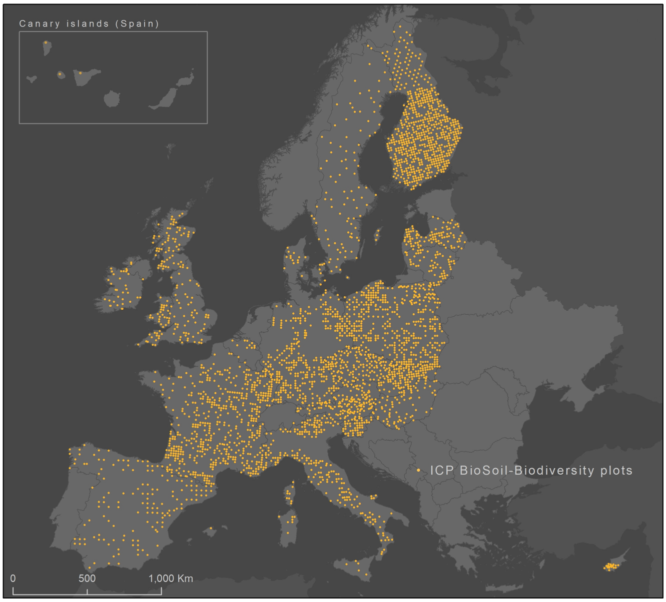


Figure 2.2: Distribution of the ICP forest LI-BioDiv dataset unit used in this study (3311 plots).

2.2.2 General information about plots

For each plot, general information were acquired in the field: geographical coordinates, date of the field measurements, elevation, aspect, slope, previous land use, origin of the stand, management intensity, management type, signs of removal of coarse woody debris, pattern of tree mixture, mean age of the stand, presence of fence, and forest type according to the classification of EEA (2006). The canopy closure (CAN) was visually estimated as the percentage of trees canopy cover projected on the ground, referred to subplot 2, it was expressed in classes; the number of tree layers was also assessed. See Appendix 2.A for a detailed description of the information acquired.

2.2.3 Biomass of living trees

During the Biosoil-Biodiversity project stems belonging to living and dead trees (standing or lying) were measured if taller than 130 cm adopting a minimum DBH of 0 cm in subplot 1, a minimum DBH of 10 cm in subplot 2, and a minimum DBH of 50 cm in the whole plot (Hiederer and Durrant, 2010). For each stem DBH, species and canopy characteristics were recorded. Trees height (THT) was measured for the 3-5 trees with the largest DBH in the whole plot. For the explorative purposes of this study we calculated the plot level above ground biomass (expressed in kg ha-1) of standing living trees surveyed in subplot 2, where all trees with DBH $>$ 10 cm were recorded. To do so we found that a multivariate approach for modelling biomass on the basis of tree DBH and height was not feasible because tree height was collected in the field only for trees with the largest DBH, for this reason it was not possible to predict tree heights for smaller trees. To overcome this limitation we decided to model biomass on the basis of DBH only. To do so we used the European Allometric Equations available at GlobAllomeTree (`http://www.globallometree.org/`), an online platform which provides a consistent and harmonized database of tree allometric equations for volume, biomass and carbon assessment of trees (Henry et al., 2013). Single tree biomass estimates were then aggregated at plot level as per hectares values. Additionally for each plot we calculated the basal area too.

2.2.4 Deadwood

Deadwood was measured in the sampling area of 400 m^2 (subplot 2, or A, B, C, D sampling units) with a minimum DBH of 5 cm and it was classified in fine wood debris (FWD 5 $<$ diameter $<$ 10 cm), coarse woody debris (CWD, diameter $>$ 10 cm), snags (height $>$ 1.3 m and DBH $>$ 10 cm) and stumps (height $<$ 1.3 m and D at cut $>$ 10 cm).Lenght and species of deadwood were recorded when possible and a decay state, described by 5 different classes according to Hunter (1990), was associated to each deadwood element. For standing and lying dead trees with DBH $>$ 10 cm, we calculated the volume with the same allometric equations used for the living trees. For snags we used the truncated cone formula with a taper of 1 cm m^{-1} (eq. 2.1).

$$VS = \frac{1}{3}\pi(r_1^2 + r_1 r_2 + r_2^2)h \tag{2.1}$$

Finally we calculated the volume for CWD, FWD and stump with the Huber's formula (eq. 2.2):

$$V = \frac{L\pi D_m^2}{4} \tag{2.2}$$

where V = CWD, FWD and stumps volume, L= length of the log, d_m= mid-diameter of the log. We calculated the volume of total deadwood for each one of the plots of the LI-BioDiv database summing up the volume of all the deadwood components (lying and standing dead trees, fine and coarse woody debris, snags and stumps) and referred to the hectare (expressed in m^3 ha^{-1}).

2.2.5 Ground vegetation data

Under the ICP Forests BioSoil network the objective of ground vegetation survey was to provide information on species richness and specific abundance at the plot level, following a common and standardized field work method (Aamlid et al., 2007). A mandatory Common Sampling Area (CSA) of 400 m^2 was adopted in order to achieve comparability of results between countries (Aamlid et al., 2007). The ground vegetation assessment was based on a census of the species listed in the Flora Europaeae and identified by a nine-digit code (WGFB, 2011). The layers where vegetation was evaluated are the moss layer (i.e. terricolous bryophytes and lichens), the herb layer (all non-ligneous, and ligneous $\leq$ 0.5m height), the shrub layer (only ligneous and all climbers $>$ 0.5 m height, up to 5 m) and the tree layer (only ligneous and all climbers $>$ 5 m height).

In some plots of France, Luxembourg and Slovenia two layers were used for shrubs: lower shrubs and upper shrubs. In the CSA of 400 m^2 only the vascular plant species list was mandatory assessed. However, as optional variables, some countries have also recorded terricolous lichens and bryophytes, and/or the specific coverage in the various layer using a percentage scale. For the purposes of this work we calculated the alpha diversity as the total number of species recorded in each CSA for each one of the plots of the LI-BioDiv database where this information was available.

2.3 Results

In this section we first present the consistency of the raw data available in the LI-BioDiv database, and then the results of our elaboration for calculating plot level aggregated values for biomass of living trees, deadwood volume and alpha diversity. The complete description of record layout of the different tables of the LI-BioDiv database is instead available in 2.A.

2.3.1 LI-BioDiv database consistency

The DBH, DWD and CAN tables contains records that belong to twenty countries: France, Belgium, Germany, Italy, United Kingdom, Ireland, Denmark, Spain, Sweden, Austria, Finland, Hungary, Poland, Slovak Republic, Norway, Lithuania, Czech Republic, Slovenia, Latvia, Cyprus and Canaries islands. The same for the GVG dataset, with the exclusion of Sweden. The map representing spatial distribution of data across countries are presented in Appendix 2.B. The DBH table counts 3189 plots, but for 59 of them the geographic coordinates are not available (Tab. 2.2, Appendix 2.B, Fig. 2.10). The DWD table has 2885 plots, but 51 of them are not georeferenced (Tab. 2.2). The CAN table has 3214 plots, 176 of them are not georeferenced (Tab. 2.2 and Appendix 2.B Fig. 2.11). GVG dataset counts 3123 plots, 30 of them are not georeferenced

(Tab. 2.2).

		DBH		**DWD**		**CAN**		**GVG**	
Country	**Code**	**TOT**	**GEO**	**TOT**	**GEO**	**TOT**	**GEO**	**TOT**	**GEO**
France	1	539	530	504	504	538	538	547	547
Belgium	2	10	10	10	10	10	10	10	10
Germany	4	225	224	212	212	223	222	312	312
Italy	5	219	218	179	176	220	219	201	199
United Kingdom	6	163	163	121	119	163	163	157	157
Ireland	7	35	34	35	21	35	35	29	29
Denmark	8	22	22	5	5	22	22	22	22
Spain	11	145	142	92	86	151	151	151	147
Sweden	13	100	100	85	85	100	76	-	-
Austria	14	135	135	128	128	133	133	136	136
Finland	15	621	606	577	577	630	625	629	629
Hungary	51	77	75	74	74	78	78	18	18
Poland	53	432	411	408	390	438	438	438	418
Slovak Republic	54	107	106	104	101	108	108	108	107
Lithuania	56	62	61	58	57	62	62	62	61
Czech Republic	58	139	137	142	142	141	-	146	146
Slovenia	60	40	40	40	40	44	40	39	39
Latvia	64	95	95	88	88	95	95	95	95
Cyprus	66	19	18	19	16	19	19	19	18
Canaries islands	95	4	3	4	3	4	4	4	3
	tot	**3189**	**3130**	**2885**	**2834**	**3214**	**3038**	**3123**	**3093**

Table 2.2: Total number of available plots (TOT) and the number of plots with geographic coordinates information (GEO), by country, in the ICP Forests LI-BioDiv dataset. In the Li-BioDiv database Canaries islands are considered separately from the rest of Spain, so we presented data accordingly with the database code.

Regarding GVG dataset France, Belgium, Germany, Italy, United Kingdom, Spain, Finland, Poland, Slovak Rep., Czech Rep., Slovenia and Canaries islands have surveyed the optional percentage coverage of each vascular plant species (Appendix 2.B, Fig. 2.12). In reference to the survey period, the data has been recorded between February and December (Appendix 2.B, Fig. 2.13) and Galluzzi et al. (2018) provide a description of data variability and survey period. As a results of our preliminary analysis on a total of 3311 plots, 74% of them (2446) contain information about all data type (Fig. 2.3). The remaining plots contain one or more data type, but not all of them. 330 plots contain only DBH, CAN and GVG data, 158 plots only DBH, DWD and CAN data, 146 plots gather DBH, DWD and GVG data and 102 plots contained only GVG data. From 2005 to 2008, all data types were collected in each plot one time with the exception of 283 plots in CAN dataset from France that have been visited in both 2006 and 2007.

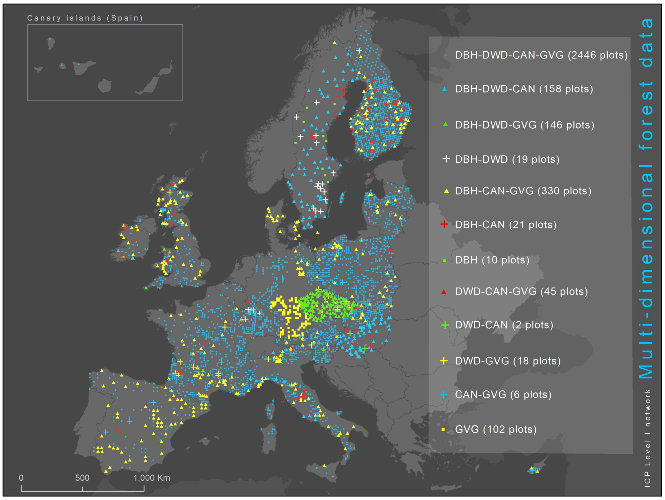


Figure 2.3: Data type availability for each plot from the ICP Forests LI-BioDiv dataset.

2.3.2 Plot-level aggregated variables

The growing stock and deadwood volume and alpha diversity distributions through the 3311 plots is in Fig. 3.4.

Above ground biomass ranges between 40 and 120 t ha-1 for the 50% of plots (Fig. 2.5). In order to assess the characteristics of the plots included in the LI-BioDiv we compared aggregated values by countries with biomass data coming from NFIs (Avitabile and Camia, 2018) (Tab. 2.3). We found a positive significant relationship between the two data sources ($R^2 = 0.35$, $F = 9.6$, $p<0.01$). By removing Denmark as an outlier because of the very limited forest area, the comparison we found has strong positive significant relationship ($R^2 = 0.5$, $F = 16.28$, $p<0.01$) (Fig. 2.6).

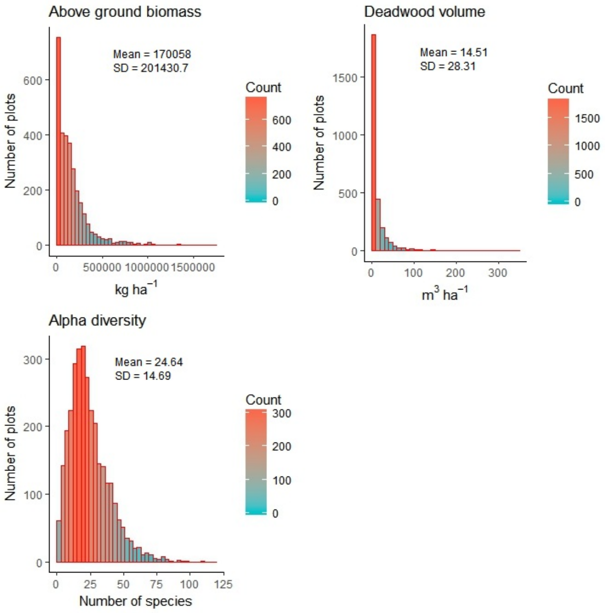


Figure 2.4: Distribution of the number of plot related with above ground biomass (t ha^{-1}) deadwood volume (m^3 ha^{-1}), and alpha diversity across the LI-BioDiv database.

Deadwood volume ranges between 2 and 6 m^3 ha^{-1} in the 50% of plots (Fig. 2.7). We compared our results with standing and lying deadwood data coming from MCPFE (2007) (Tab. 2.4). Between MCPFE (2007) and Li-BioDiv data we found a positive significant relationship ($R^2 = 0.4$, $F = 8$, $p<0.05$) (Fig. 2.8).

Alpha diversity has almost a normal distribution with values ranging between 14 and 33 species per plot for 50% of them (Fig. 2.12). We present data across MCPFE region (Tab. 2.5). The highest values of species richness were found in the South-West Europe region with 1102 species and in the North-West Europe region with 1061 species (Tab. 2.5). 873 species were recorded in the Central Europe region, while in the Nordic/Baltic region we found an average lower number of species (433) and in South-East Europe

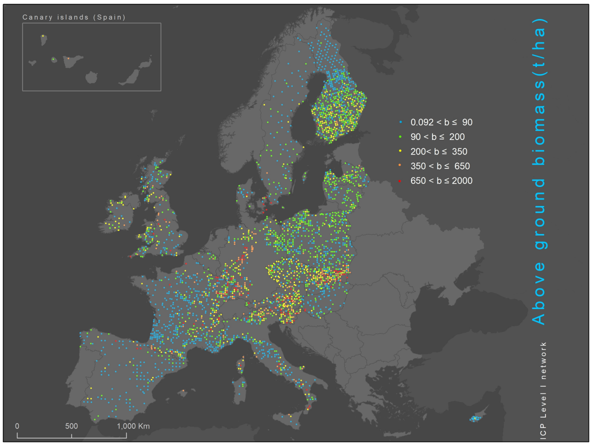


Figure 2.5: Above ground biomass (t ha^{-1}) in the plots from the LI-BioDiv database.

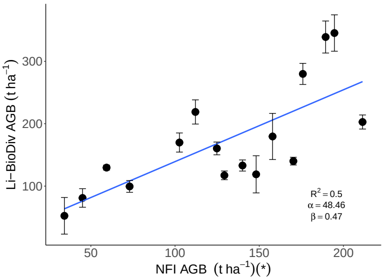


Figure 2.6: Relationship between Li-BioDiv above ground biomass (AGB) data NFIs AGB. (*) data source: Avitabile and Camia (2018) (R2 = 0.5, F = 16.28, $p<0.01$).

region with 473 species (Tab. 2.5). Observing the spatial patter of alpha diversity, it is possible to see that the north-west part of Europe has continues lower values of species (Table 2.6). Furthermore we compared tree and vascular species stored in the Li-BioDiv database with threatened species listed in the IUCN Red List Categories (Bilz et al., 2011). We compared our results with the Forest

Country	Country code	Li-BioDiv AGB (t ha^{-1})	NFI AGB (t ha^{-1}) (*)
France	1	133	140
Belgium	2	180	158
Germany	4	280	176
Italy	5	170	103
Ireland	7	219	112
Denmark	8	343	114
Spain	11	81	45
Sweden	13	100	73
Austria	14	339	189
Finland	15	130	59
Hungary	51	119	148
Poland	53	140	170
Slovak Republic	54	345	195
Lithuania	56	117	129
Czech Republic	58	203	211
Latvia	64	160	125
Cyprus	66	52	34

Table 2.3: Average values of Above ground biomass (AGB) by country, data from Liv-BioDiv database and from National Forest Inventories (NFI). (*) data source: Avitabile and Camia (2018).

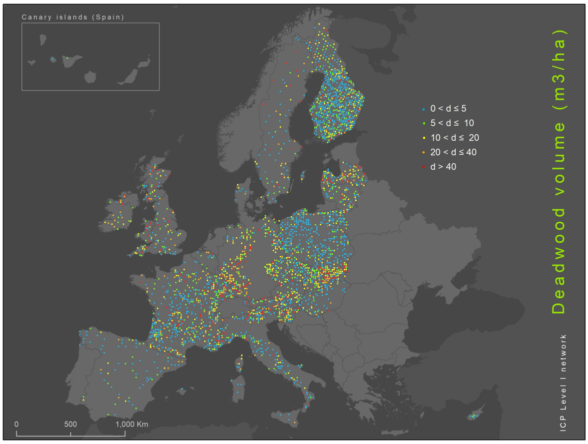


Figure 2.7: Deadwood volume (m^3 ha^{-1}) in in the plots from the LI-BioDiv database.

Europe indicator 4.8 "Threatened forest species" (MCPFE, 2007). Very few threatened vascular species were found in the Li-BioDiv data base (Figure 2.6) compared with data reported by MCPFE

Country	Country code	Li-BioDiv DWD (m^3 ha^{-1})	Eu's forests DWD (m^3 ha^{-1}) (*)
Belgium	2	27	7
Germany	4	22	11
Italy	5	13	12
United Kingdom	6	18	4
Spain	11	5	1
Sweden	13	16	6
Finland	15	17	6
Poland	53	8	2
Czech Republic	58	14	11
Slovenia	60	34	16
Latvia	64	18	16
Cyprus	66	0	1

Table 2.4: Average of standing and lying deadwood volume by country of EU's forests ((*) data from MCPFE (2007)) and from Liv-BioDiv database

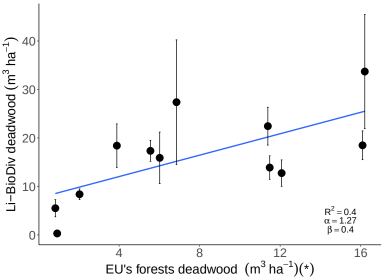


Figure 2.8: Relationship between Li-BioDiv standing and lying deadwood data and EU's forests standing and lying deadwood data ((*) data coming from MCPFE (2007)) (R^2 = 0.4, F = 8, $p<0.05$).

(2007), while no tree threatened species were found (Table 2.6).

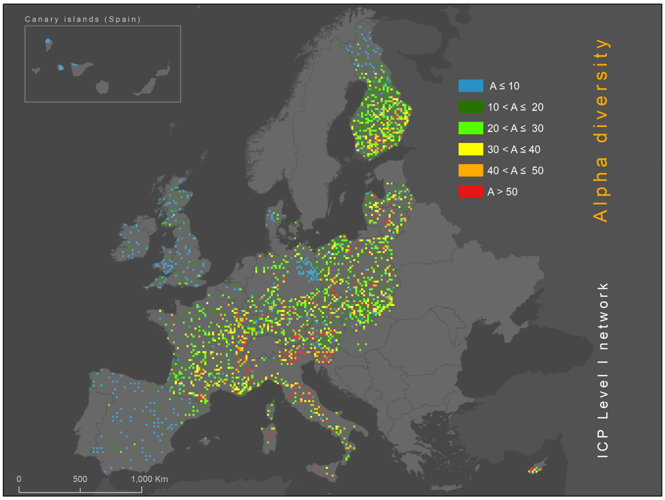


Figure 2.9: Alpha diversity, measured as the number of species, in in the plots from the LI-BioDiv dataset.

Region	Species richness
Central Europe	873
North-West Europe	1061
North/Baltic Europe	433
South-East Europe	473
South-West Europe	1102

Table 2.5: Average number of species from the LI-BioDiv dataset by MCPFE region

Country	Country code	Li-BioDiv threatened	EU threatened (*)
France	1	3	/
Belgium	2	1	14
Germany	4	2	/
Italy	5	5	/
United Kingdom	6	1	32
Ireland	7	0	/
Denmark	8	1	/
Spain	11	1	/
Sweden	13	0	45
Austria	14	5	270
Finland	15	0	35
Hungary	51	2	/
Poland	53	3	/
Slovak Republic	54	2	207
Lithuania	56	0	/
Czech Republic	58	2	771
Slovenia	60	2	/
Latvia	64	1	28
Cyprus	66	0	17

Table 2.6: Numbers of threatened vascular species (IUCN Red List Categories, (Bilz et al., 2011)) founded in the Li-BioDiv database. (*) Data from (MCPFE, 2007), 4.8 indicator "Threatened forest species".

2.4 Discussion

In physical sciences, it is not possible to assess phenomena until they are measured. When dealing with forest monitoring, the collection and aggregation of meaningful forest-related physical, compositional and structural data is needed to guarantee an efficient and effective analysis of forests changes in space and time. A wide variety of monitoring networks have been developed in Europe for reporting purposes or for analyzing ecological functioning of forest habitats and their relations with climate change, air pollution, and human activities. NFIs are the official sources for reporting national and international level statistics on a wide variety of ecosystem services produced by forests (Tomppo et al., 2010). Unfortunately NFIs still suffer for a lack of consistency at international level because in the different countries they adopted different definitions for a large number of forest variables (McRoberts et al., 2009). As a result statistics from the NFIs need to be harmonized before comparison or aggregation at international level. This is relatively simple for some traditional forest variables (Vidal et al., 2008) but it is extremely difficult or totally impossible for variables needed for the computation of biodiversity indicators such as deadwood (Rondeux et al., 2012) or ground vegetation (Winter et al., 2008; Chirici et al., 2011, 2012). As a result, forest biodiversity monitoring based on NFI data was frequently based on the calculation of a limited number of compositional or structural indexes which takes into consideration the tree component of flora only (McRoberts et al., 2008; Corona et al., 2011). In this context, the ICP BioSoil LI-BioDiv database represents the only standardized dataset of multiple forest variables measurements available at pan-European level (Simpson et al., 2006). In this paper we presented the consistency of the different dataset available in the LI-BioDiv database demonstrating its potential relevance for the derivation of multiple indicators on different forest biodiversity components. The consistency of the LI-BioDiv database in terms of number of plots where all the different variables are available is of 2446 plots (the 74% of the total plots investigated in the Biosoil-Biodiversity project). Regarding the above ground biomass, we compared the per-country aggregated values of the LI-BioDiv database with Avitabile and Camia (2018) which are based on NFIs official statistics. We found that the biomass in the plots of the LI-BioDiv database are always higher. It is difficult to find a specific reason for this. Of course the number of plots available in the LI-BioDiv database is much lower than those available from NFIs and thus the uncertainty of our estimations is much greater. Allometric equations are sensitive to sample size (as number of sample unit), and small sample size may lead to an overestimate in biomass (Chave et al., 2014; Duncanson et al., 2015). Furthermore, the Li-BioDiv plots surveyed were a subset of ICP Level I network and the method used to select the Li-BioDiv plots from the ICP level I dataset is still unclear. It is possible that less disturbed plots were selected, which may bring to an overall positive bias in biomass estimation. However, a strong positive relationship was found between Li-BioDiv data and NFIs data, excluding Denmark the coefficient of determination reaches 0.5 (Fig. 2.6). The results regarding deadwood are similar. At country level deadwood volume data from Li-BioDiv are always higher than official statistics from 12 countries. Even if a consistent statistical relationship exist between the two (Fig. 2.8). The reason for this difference may be the same of that one we hypothesized for biomass overestimation. If the Li-BioDiv plots are those ones less disturbed of the ICP-level I set, than these plots probably accumulated larger amounts of deadwood too. In addition this difference could be due to different methods of deadwood measure.

Regarding data of alpha diversity as species richness, the geographic patterns markedly differs across the continent, with the lowest values in the South-West Europe countries e.g. the Spain. Our results are partly in agreement with the highest values of species richness distribution in the Central Europe, in particular for mountainous regions, found by Kalwij et al. (2014). From this comparison, also the United Kingdom seems to have lower values of species richness. Actually, there isn't any extensive and comprehensive vascular plant distribution datasets useful for comparing our results about alpha diversity. Comparison with Forest Europe indicator 4.8 "threatened forest species" highlighted a weakness and a limit of the ground vegetation dataset. Most of the species contained in the IUCN Red List categories have a very restricted distribution and are subspecies. Hence, the level of experience of the observer is certainly crucial fo the success of the surveys. The value of this collection of data lies in the combination of forest structure with floristic data. This is a unique dataset at European scale leading consider the use of ICP Forests BioSoil data as a possible integrated potential useful dataset for supporting the creation of an integrated system for monitoring forest biodiversity in Europe.

2.5 Conclusion

This work is aimed at presenting the first elaboration of the LI-BioDiv dataset acquired in the framework of the Biosoil-Biodiversity project in a set of 3311 plots belonging to the pan-European systematic grid of the ICP-Level I network. More specifically we considered the information related to living trees, deadwood components, shrub and herbaceous species composition. We clarified the characteristics and consistency of the data available and we completed some basics pre-elaborations calculating plot-level variables potentially useful as forest biodiversity indicators: above ground biomass, total deadwood and alpha biodiversity (in terms of total number of plant species) and we compared our species with species listed in IUCN Red List Categories. Furthermore with this paper we have provided a tool to understand how Li-BioDiv data can be used beyond the countries boundaries and hoe data can be set to explore them across biogegraphical regions. Some conclusions can be derived from the results obtained. The LI-BioDiv database is a standardized source of information on European forests which may have a relevant importance for the scientific analysis of the relationships between multiple characteristics of European forests. In particular some scietific question, relevant for forest biodiversity monitoring, could be answered by analysing Li-BioDiv database:

- to link tree data with functional traits, exploring functional biodiversity of European forest and their adaptation to climate changes;
- how forest structure and composition change according to stand age, management intensity, management type and forest type;
- to understand if forest landscape composition influence forest characteristics;
- to link tree varaibles with Natura 2000 sites, when possible, and to define indices that can support conservation strategies;
- exploring if some biodiversity characteristic (i.e. functional diversity) shows relationship with remote sensing data.

The LI-BioDiv and more in general the ICP Level I network can be used to develop future multidimensional monitoring programs including information on additional variables relevant for a more complete biodiversity assessment. For example including other flora (such as bryophytes and lichens) and fauna components. More important is to take into accout the effect of the selection, made by countries, of the ICP level I plots to be included in the BioSoil Biodiversity project. Basing on our results and discussion, make the Li-BioDiv monitoring a true probabilistic samplig. In addition, the Li-BioDiv database cannot be used for statistical inference for the derivation of estimation at pan European level because of its limited number of plots and the possible subjective selection of the plots. Furthermore, we encourage institutions responsible of the implementation of NFIs in Europe to consider the inclusion of more complete and formal ground vegetation surveys in NFIs field protocols, at least for those countries where this is not yet the case. The multiple repetition of the survey in the same points in the future will make possible to carry out trends analysis of European forests in respect of the obligations set up by the Convention on Biological Diversity and EU Biodiversity Strategy initiatives for halting the loss of biodiversity by 2020 (UE, 2011). The analysis of the relationships between biodiversity trends and forest management across different EU forest types could help in proposing more sustainable practices. Finally we encourage a more consistent integration of ICP level I network with NFIs networks, also following existing technical proposals (Travaglini et al., 2013), toward the implementation of a European forest monitoring system able to support forest policy decision at pan-European level for halting the loss of biodiversity.

2.A Data organization

Information about dataset table is presented below, including detailed descriptions of all attributes within the tables. Each attribute in a table is listed with its abbreviated name followed by a description of the attribute. Attributes that are coded include a list of the codes and their meanings. The attribute called "quality" has been defined for easily filtering the necessary data for each data type.

Common attributes to all data type:

- **id_unique**: progressive number that identified the rows;
- **survey_year**: year when the surveys were carried out;
- **code_country**: number that identifies the country (Tab. 2.1);
- **code_plot**: number that identifies the plot;
- **index**: links "code_country" and "code_plot" to the unique plot record;
- **bd_subplot** (only for level I): subplot number where the diameters were recorded. Code 1 - subplot with a radius of 3.09m (30m^2). Code 2 - subplot with a radius of 11.28m (400m^2). Code 3 - subplot with a radius of 25.24m (2000m^2).

Specific attributes to GPL dataset:

- **Latitude**
- **Longitude**
- **gps_elevation**: values of elevation;
- **code_orientaion**: Code 1- North. Code 2- North-east. Code 3- East. Code 4- South-east. Code 5- South. Code

6 - South-west. Code 7 – West. Code 8- North-west. Code 9- Flat.

- **Slope**
- **Code_preuse**: previous land use. Code 1- Forested more than 300 years. Code 2- Forested more than 100 years. Code 3- Forested for 25 - 100 years ago. Code 4- Forested in the past 25 years. Code 5- No information.
- **Code_stand_actual**: Code 1- Planted. Code 2- Seeded. Code 3- Natural regeneration. Code 4- Mixed. Code 9- Unknown.
- **Code_manage_intensity**: Code 1- Unmanaged (no evidence). Code 2- Management (evidence but for more than 10 years ago). Code 3- Managed (within the last 10 years). Code 9- Unknown.
- **Code_manage_type_bd**: Code 1- High forest - Femelschlag. Code 2- High - Small groups. Code 3- High forest (uneven aged) - Plenterwald. Code 4- High forest (other). Code 5- Young/Medium forest (under development to high forest). Code 6 - Coppice without standards. Code 7 – Coppice with standards. Code 8- Other.
- **Code_dw_rem**: Removal of coarse woody debris. Code 1- Yes, all stems and main branches have been removed. Code 2- Yes, stems and main branches have been removed. Code 3- No, stems and main branches are lying in the forest. Code 4- Partly, some stems and main branches have been removed, others still present. Code 5- Unknown. Code 6 - Introduced. Code 7 – Presence of accumulation (branches have been stacked in piles or in rows).
- **Code_treemix**: Pattern of tree mixture. Code 1- Intimate (different tree species are mixed throughout the stand). Code

2- Non-intimate (different trees occur in clusters). Code 3- No mixture.

- **Code_meanage_bd**: mean age of stand. Code 1- 0-20 years. Code 2- 21-40 years. Code 3- 41-60 years. Code 4- 61-80 years. Code 5- 81-100 years. Code 6 - 101- 120 years. Code 7 – >120 years. Code 8- Irregular stands. Code 9- Unknown.
- **Code_fencing**: fancing of the plot. Code 1- Fenced. Code 2- Not Fenced. Code 3- Fenced in parts.
- **Code_forest_type_bd**: code in accordance with EEA (2006).

Specific attributes to DBH dataset:

- **tree_number**: number that identified the tree inside each plots;
- **dw_dbh**: values of diameters (cm);
- **code_tree_status**: Code 1 - standing living tree. Code 2 - standing dead tree. Code 3 - lying dead tree;
- **code_tree_species**: number that identifies the tree species;
- **quality**: attribute to filter data availability. Code 1 – Fields holding “diameter”, “code_tree_status” and “code_tree_species” (D>10cm). Code 2 – Fields holding “diameter”, “code_tree_species” and does not contain “code_tree_status”(D>10cm). Code 3 - Fields holding “diameter”, “code_tree_status” and does not contain “code_tree_species” (D>10cm). Code 4 Fields holding “diameter”, “code_tree_status” and “code_tree_species” ($3 \leq D \leq 10$cm). Code 5 - Fields holding “diameter”, “code_tree_species” and does not contain

"code_tree_status" ($3 \leq D \leq 10$cm). Code 6 - Fields holding "diameter", "code_tree_status" and does not contain "code_tree_species" ($3 \leq D \leq 10$cm). Code 999 - Unclassified or incorrect data;

- **code_decay**: deadwood decomposition is assigned in 5 decay classes according to Hunter (1990). Code 1 - No evidence of decay. Code 2 - Solid wood. Less than 10 % changed structure due to decomposition. The wood is solid at its surface. The wood is attacked only to a very mall degree by wood decomposing organisms. Code 3 - Slightly decayed. 10-25% of the wood has a changed structure due to decomposition. This can be assessed by sticking the wood with a harp object. Code 4 - Decomposed wood 26-75% of the wood is soft to very soft. Code 5 - Very decomposed wood. 76% - 100% of the wood is soft.

Specific attributes to deadwood (DWD) dataset:

- **deadwood_id**: number that identified the wood inside each plots;

- **dw_type**: existing classification based on type of deadwood detected. Code 1 - Coarse woody debris ($D>10$ cm). Includes stems, limbs, branches lying on the ground. Code 2 - Fine woody debris (5 cm $<D<$10 cm). Includes small wood pieces. Code 3 - Snag (height $>$ 1.3 m and DBH $>$ 10 cm). Standing deadwood without branches. Code 4 - Stump (height $<$1.3 m and D at cut $>$ 10 cm). Stump is a snag below breast height. Code 5 - Other. All cases with values falling outside the above mentioned definitions or undefined. (e.g.: values $<$ 5 cm for diameter with code=2 or unsolvable code or definition conflicts). Code 9 - Special cases. Threshold values, erroneously not defined in the old manual (i.e.: $D = 10$ cm, $D = 5$ cm, $DBH = 10$ cm, height $= 1.3$ m);

- **code_dw_species**: number that identifies the group of tree species. Code 1 – deciduous, code 2 – conifer, code 3 – unknown;
- **diameter**: values of diameters (cm);
- **dw_length**: values of length of woody debris (m);
- **code_decay**: deadwood decomposition is assigned in 5 decay classes according to Hunter (1990), see DBH dataset code decay;
- **quality**: attribute to filter data availability. Code 1 – Fields holding “diameter”, "dw_length", "code_dw_species" and "code_decay" (CWD). Code 2 - Fields holding “diameter”, "dw_length", "code_dw_species" and does not contain "code_decay" (CWD). Code 3 - Fields holding “diameter”, "code_dw_species", "code_decay" and does not contain "dw_length" (CWD). Code 4 - Fields holding “diameter”, "dw_length", "code_decay" and does not contain "code_dw_species"(CWD). Code 5 - Fields holding “diameter”, "dw_length", "code_dw_species" and "code_decay" (FWD). Code 6 - Fields holding “diameter”, "dw_length", "code_decay" and does not contain "code_dw_species"(FWD). Code 7 - Fields holding “diameter”, "dw_length", "code_dw_species" and does not contain "code_decay" (FWD). Code 8 - Fields holding “diameter”, "code_dw_species", "code_decay" and does not contain "dw_length" (FWD). Code 999 - Unclassified or incorrect data.

Specific attributes to canopy (CAN) dataset:

- **code_canopy**: code that identified the average percentage of canopy cover. Code 1 - open sky. Code 2 - 1-25%. Code 3 - 25-50%. Code 4 - 50-75%. Code 5 - >75%;

- **n_tree_layer**: number of distinct tree layer. Code 1 - one layer (one dominant tree layer). Code 2 - two layers (dominant tree layer plus 1 sublayer). Code 3 – three layers (dominant plus two sublayers). Code 4 - more than three layers. Code 5 - no tree layer;
- **n_trees**: number of trees in the plot;
- **quality**: attribute to filter data availability. Code 1 – Fields holding “code_canopy”, “n_tree_layer” and “n_trees”. Code 2- Fields holding “code_canopy”, “n_tree_layer” and does not contain the “n_trees”. Code 3- Fields holding “code_canopy” and does not contain the “n_tree_layer” and “n_trees”. Code 4 - Fields holding “code_canopy”, “n_trees” and does not contain the “n_tree_layer”. Code 999 - Unclassified or incorrect data.

Specific attributes to ground vegetation (GVG) dataset:

- **code_species**: number that identifies the species;
- **code_layer_surface**: code of layers where vegetation was evaluated. Code 1 - tree layer (only ligneous and all climbers) $>$ 5 m height, code 2 - shrub layer (only ligneous an all climbers) $>$ 0.5 m height, code 3 - herb layer (all non-ligneous, and ligneous $<$ 0.5m height), code 4 - moss layer (i.e. terricolous bryophytes and lichens), code 5 - lower Shrubs, code 6 - upper Shrubs;
- **cover**: percentage of coverage of the species;
- **quality**: attribute to filter data availability. Code 1 – Fields holding "code_species" and "cover". Code 2- Fields holding "code_species" and does not contain "cover" and "code_layer_surface". Code 3 - Fields holding "code_species", "cover" and "code_layer_surface". Code 4 - Fields holding "code_species", "code_layer_surface" and does not contain "cover";

- **class**: Field refers to “code_species”. Code S, “code_species” identified the species. Code G, “code_species” identified the genus.

2.B Supplementary material

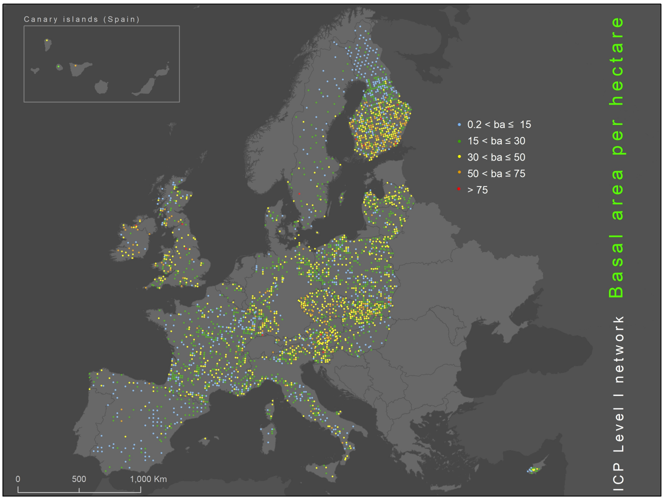


Figure 2.10: Basal area (m^2 ha^{-1}) from the ICP Forests LI-BioDiv dataset.

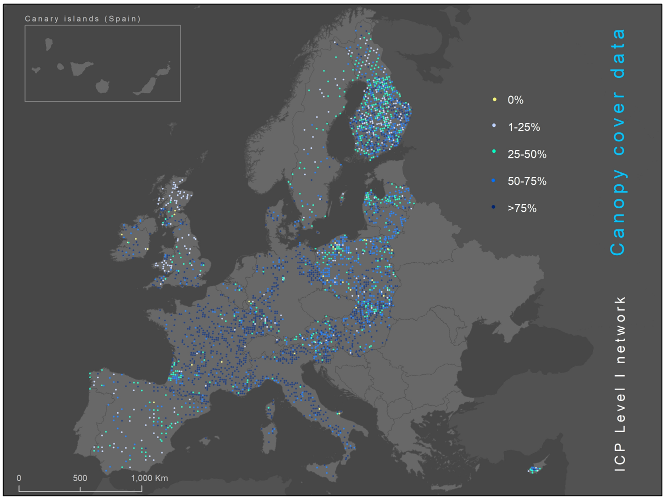


Figure 2.11: CAN data available from the ICP Forests LI-BioDiv dataset and canopy cover variation.

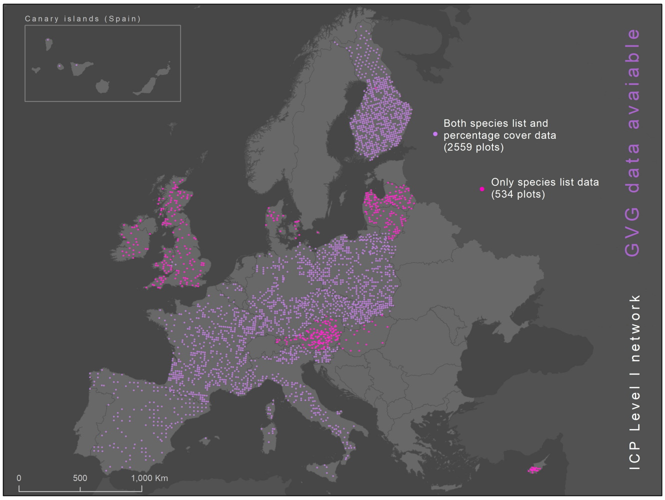
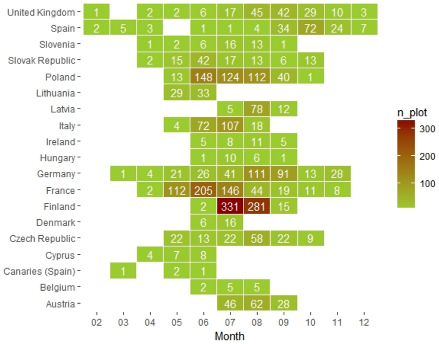


Figure 2.12: Type of GVG data available from the ICP Forests LI-BioDiv dataset.



	02	03	04	05	06	07	08	09	10	11	12
United Kingdom	1		2	2	6	17	45	42	29	10	3
Spain	2	5	3		1	1	4	34	72	24	7
Slovenia			1	2	6	16	13	1			
Slovak Republic			2	15	42	17	13	6	13		
Poland				13	148	124	112	40	1		
Lithuania				29	33						
Latvia						5	78	12			
Italy				4	72	107	18				
Ireland					5	8	11	5			
Hungary					1	10	6	1			
Germany		1	4	21	26	41	111	91	13	28	
France			2	112	205	146	44	19	11	8	
Finland					2	331	281	15			
Denmark					6	16					
Czech Republic				22	13	22	58	22	9		
Cyprus			4	7	8						
Canaries (Spain)		1		2	1						
Belgium					2	5	5				
Austria						46	62	28			

Figure 2.13: Number of plots surveyed for the census of ground vegetation for each month by the different countries (GVG dataset)

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Chapter 3

Paper II - Mapping uncertainty of ICP-Forest biodiversity data: from standard treatment of diffusion to density-equalizing cartograms

Marta Galluzzi[1]*, Duccio Rocchini[2], Roberto Canullo[3], Ronald E. McRoberts[4], Gherardo Chirici[1]

Abstract

Data uncertainty due to spatial gaps and heterogeneity is a fundamental problem in conservation and environmental planning. Thus, investigation of issues related to data uncertainty contributes to more efficient conservation plans. We evaluated the uncertainty of data related to forest diversity descriptors using a diffusion-based cartogram approach that visually displays how data information change in function with respect to degree of uncertainty. We used ground vegetation data for 3093 plots collected as part of the BioSoil project through the ICP Forests

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Level I network and stored in the LI-BioDiv database. For each plot, we assigned an uncertainty value based on the survey season and the mean monthly temperature for the survey period. The density-equalizing map or cartogram highlights that data collected in Spain, the United Kingdom and the German federal states of Berlin and Brandenburg have smaller values of species richness corresponding to larger values of uncertainty. We found than an awareness of the negative relationship between the period of survey and species richness that can lead to a good data handling and analysis. We demonstrated that cartograms are efficient tools for evaluating and managing uncertainty and can strengthen the results of data analysis by providing alternative perspectives and interpretations of spatial phenomena.

Keywords: *density-equalizing cartograms; uncertainty map; plant species richness; European biodiversity inference.*

3.1 Introduction

Rigorous vegetation science and efficient environmental and forest planning require careful assessment and management of data uncertainty from sources (Meyer et al., 2016; Mason et al., 2015) that include systematic observer error (Hall and Okali, 1978), purposive or subjective rather than random sampling (Lepš and Hadincová, 1992), temporal change in plant communities (Kopecký and Macek, 2015), and the dynamic nature of species distributions (Rocchini et al., 2011). The adverse effects of data uncertainty are often underrated (Walther and Moore, 2005) but extend to predictive models, estimates of key biodiversity indicators (Mason et al., 2015) and the reliability of conservation strategy outcomes (Meyer et al., 2016; Hortal et al., 2008). Hence, understanding data uncertainty is necessary to avoid biased ecological inferences (Meyer et al., 2016).

Maps are a central element for describing phenomena and for understanding patterns beyond the data. In particular, the spatial aspects of data uncertainty could be represented by a "map of ignorance" that depicts uncertainty in a spatially explicit manner (Rocchini et al., 2011). For this purpose, density-equalizing maps or cartograms are increasingly used to represent aspects of spatial variation for multiple variables simultaneously (Hennig, 2014). Cartograms are maps that vary the size and the shape of geographic regions proportionally with respect to a selected variable (Gastner and Newman, 2004; Tobler, 2004). Specifically, Gastner and Newman (2004) proposed a diffusion-based method for producing density-equalizing maps by first defining a starting map density $\rho(r)$ (where r represents geographic position) as the basis for constucting the cartogram. Traditionally, cartograms have been used to represent economic and political data but more recently have also been used to display statistical information for purposes of identifying the potential spread of disease (Kronenfeld and Wong, 2017) and quantifying the effects of non-random sampling on species occurrence estimates (Rocchini et al., 2017). Conventional maps use two features, color and geometry, to describe attributes, a practice that may be less well-suited for heterogeneous information and big-data. A cartogram uses three types of information: geographic information (distorted), statistical information (depending on area size) and additional related information (defined by color). Specifically, a cartogram depicts the values of the attribute of interest by changing area sizes and shapes to reflect different levels of importance.

Knowing the spatial distribution of a plant species contributes to understanding the spatio-temporal ecological processes and ecosystem functions that provide benefit for other species, physical phenomena and human well-being (Meyer et al., 2016). Describing and assessing how vegetation biodiversity patterns change in response to anthropogenic pressures is a great challenge whose solution is limited by the lack of biodiversity information (Mihoub et al., 2017). Existing sources of biodiversity information include the Global Biodiversity Information Facility (GBIF; `http:`

`//www.gbif.org/`) that provides world-scale biological data and the European Vegetation Archive (EVA; `http://euroveg.org/`), a centralized database of European vegetation plot data. Although these are important sources of data, they are limited by the different survey designs, operators and sampling protocols used to acquire the data. For Europe, data collected between 2005 and 2008 by the BioSoil-Biodiversity project as part of the International Cooperative Programme on Assessment and Monitoring of Air pollution effects on Forests (ICP Forests) represent a unique example of a pan-European forest and biodiversity database obtained using statistically rigourous systematic surveys (16x16 km grid: Level I network). Specifically, ground vegetation (GVG) data for 3093 plots distributed across the forests of 19 European countries are available through the LI-BioDiv database (Canullo, 2016).

Despite rigourous ICP sample designs, the GVG data are still subject to uncertainty due to the different years and seasons in which the surveys were conducted, to different operators, and to different country or organization protocols. The main GVG concern regarding the LI-BioDiv data relates to differences in the years and months in which the data were collected. Plant species identification is related to morphometric characters such as leaves and flower structure. (Cope et al., 2012). Vegetation development is further affected by a complex interaction between abiotic and biotic factors throughout the year (Larcher, 2003; Rocchini et al., 2017). The optimal condition for plant growth occurs when metabolic and hormonal factors including radiation, temperature, and chemical condition are in a synergistic relationship (Larcher, 2003). A specific thermal range is a prerequisite for the life cycle with the rate of plant germination increasing as temperature increases after reaching the minimum threshold temperature (Larcher, 2003; Hatfield and Prueger, 2015). Temperature is a crucial factor that drives morphometric plant character, and multiple studies have suggested a significant relationship between mean monthly air temperature and flowering (Fitter and Fitter, 2002; Hatfield and Prueger, 2015).

Therefore, the ICP Forests Manual for Ground Vegetation assessment recommends that surveys be conducted when plants have the maximum biomass and when the maximum number of species can be correctly and readily identified (Canullo et al., 2013), although specific periods for specific bioregions are not prescribed (Bastrup-Birk et al., 2007).

Explicitly describing and quantifying the uncertainty inherent in data recorded under different environmental conditions is the first step in clarifying how data uncertainty can be managed. Construction of maps that simultaneously depict observed data and their associated uncertainties is a challenge that has rarely been addressed. The aim of this study was to develop a procedure for mapping data uncertainty resulting from different GVG survey periods as recorded for the ICP Forests Level I network. The underlying purpose was to better manage and compare uncertainties when assessing European forest biodiversity. In particular, the study had three technical objectives (i) to develop a method for characterizing uncertainty based on the theoretical temperature range for a plant's life cycle, (ii) to map uncertainty using a diffusion-based method, and (iii) to develop a method for assessing biodiversity data quality using an uncertainty-biodiversity relationship. The results will be useful for achieving satisfactory accuracy of European forest biodiversity estimates, for correctly interpretating the estimates, and for facilitating comparable use of the data. Finally, we propose an alternative method for investigating and displaying data uncertainty.

3.2 Materials and methods

Based on the LI-BioDiv dataset (UN\ECE ICP Forests PCC Collaborative Database; `www.icp-forests.org`) data from 19 countries (Austria, Belgium, Cyprus, Czech Republic, Denmark, Finland, France, Germany, Hungary, Italy, Ireland, Latvia, Lithuania, Poland, Slovak Republic, Slovenia, Spain, Sweden, United Kingdom), were assessed (Figure 3.1). All vascular plants were sampled in an area of 400- m^2 between 2005 and 2008, albeit in different months with no mandatory sampling period (Bastrup-Birk et al., 2007). The resulting species lists, coming from the data collection, have been used to estimate “plant species richness” as a measure of European-level plant diversity.

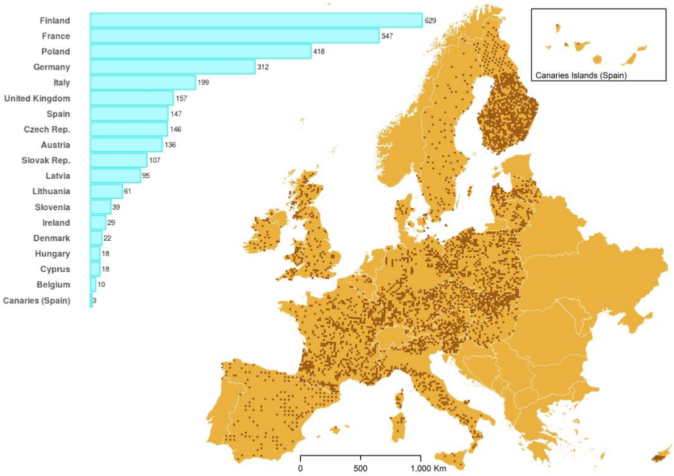
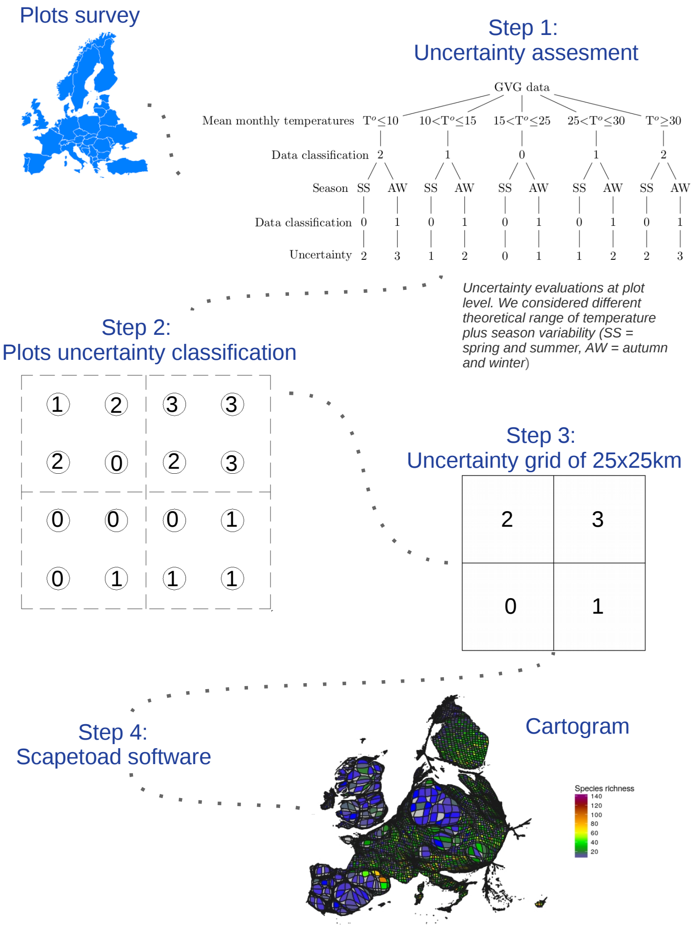


Figure 3.1: Ground vegetation plots (3093) and number of plots by country included in the LI-BioDiv dataset (after BioSoil-Biodiversity project)

Two types of information were used to asses the uncertainty of the data for each plot: the mean monthly temperature and the season of the survey. The mean monthly temperature was estimated from *Climatologies at high resolution for the earth’s land surface areas* (CHELSA) (Karger et al., 2017a,b). We assigned an increasing value for uncertainty as mean monthly temperature deviated from the theoretical optimum of $15 < T^{o} \leq 25$ (Step 1 in Figure 3.2) (Larcher, 2003; Rocchini et al., 2017). In a second step, we assigned an uncertainty value of 0 to plots surveyed between 1 April and 30 September (spring and summer, when plants have the maximum biomass, and the maximum number of species can be readily assessed (Canullo et al., 2013)) and an uncertainty value of 1 to plots surveyed between 1 October and 31 March (Figure 3.2). Taking into account these two information items, an uncertainty value ranging between 0 (no seasonal uncertainty) and 3 (maximum seasonal uncertainty) was assigned to each plot as explained in the tree diagram in Figure 3.2.

Maximum uncertainty was depicted for 25-km x 25-km grid cells encompassing the entire European continent. Cartograms were constructed using the open source software ScapeToad (`https://scapetoad.choros.ch/`). Distortions in the shapes of cells were determined by the uncertainty values, while color represented species richness values. Linear regression models were used to estimate the relationship between species richness and the uncertainty values. Finally an uncertainty versus latitude profile was developed for assessing species richness change across Europe.



Uncertainty evaluations at plot level. We considered different theoretical range of temperature plus season variability (SS = spring and summer, AW = autumn and winter)

Figure 3.2: Flowchart of steps to accomplish uncertainty cartograms. Step 1: Tree diagram explaining the uncertainty classification based i) on mean monthly temperature (increasing value of uncertainty moving from a theoretical optimum) and ii) on month of survey, with 0 uncertainty of plots surveyed between 1 April and 30 September (spring and summer). Step 2: Based on step 1, each plot was classified into different uncertainty values. To implement the cartogram, uncertainty values must be linked to a polygonal object, which in this study was represented by a grid cell with a resolution of 25-km x 25-km. Step 3: Each cell of the grid was associated with the uncertainty value of the plot with the greatest uncertainty record. Step 4: Starting from the 25-km x 25-km grid, the cartogram was produced through the open source software ScapeToad (`https://scapetoad.choros.ch/`).

3.3 Results and Discussion

The number of GVG plots surveyed per month and year by each country was displayed using a heat map whereby values stored in a matrix were represented by colors (Wilkinson and Friendly, 2009) (Figure 3.3). Spain, the United Kingdom and Germany had the greatest variability with resect to survey months followed by the Slovak Republic and France. During the autumn-winter months, Spain surveyed 111 of 147 plots, the United Kingdom surveyed 43 of 157 plots, Germany surveyed 133 of 312 plots, France surveyed 19 of 547 plots, and the Slovak Republic surveyed 13 of 107 plots.

The results of the uncertainty assessments showed that Spain, the United Kingdom and Germany had the most plots with the largest uncertainty value of 3 (Figure 3.4). In particular, Spain had only a few plots with small uncertainty values. France, the Slovak Republic, Finland, Poland and the Czech Republic had both the most plots with small uncertainty values and few plots with large uncertainty values (Figure 3.4). For the other countries, the plot uncertainty values were mostly small or absent (Figure 3.4). Overall, 63% of plots had uncertainty values of 0, 28% of plots had uncertainty values of 1, and 3% and the 4% of plots had uncertainty values of 2 and 3, respectively.

The cartograms depicted the distribution of GVG uncertainty due to survey period. Figure 3.5(a) is a cartogram in which the cell sizes were scaled according to the maximum uncertainty value of 3, while in Figure 3.5(b) cell sizes were scaled according to the second largest uncertainty value of 2. The maps exhibited considerable distortion due to the heterogeneity of the survey periods across Europe. The German federal states of Berlin and Brandenburg, Spain and the United Kingdom showed the greatest cell distortions Figure 3.5(a). In the second cartogram (Figure 3.5(b)) Spain still showed the greatest distortion while distortions for the United Kingdom and Berlin and Brandenburg were reduced. The cartogram showed that greater cell distortion was associated with

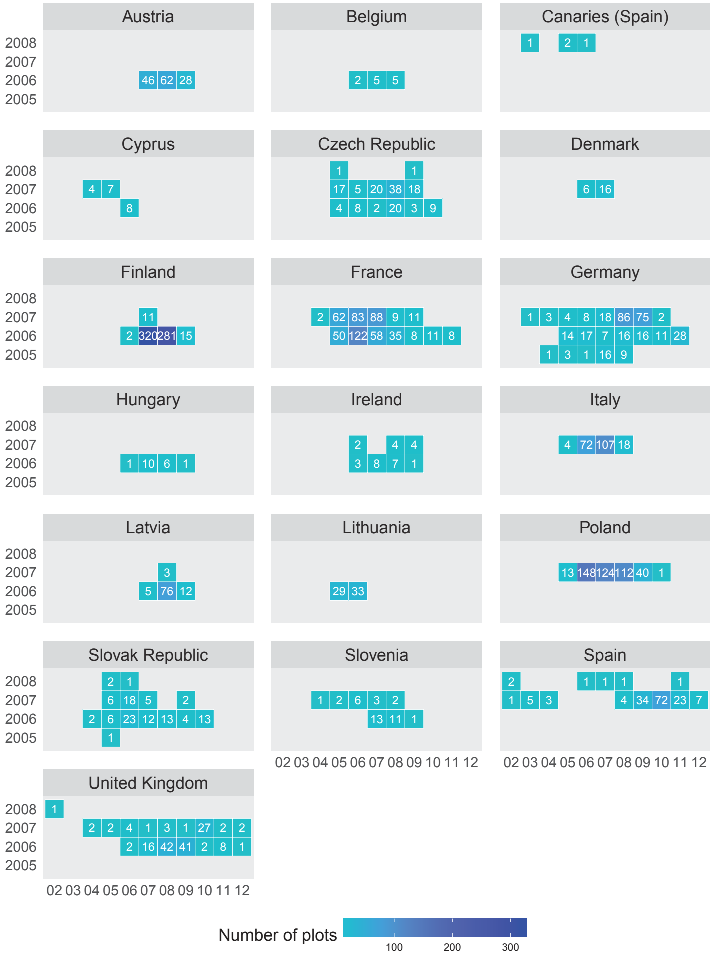


Figure 3.3: Number of plot per month and year surveyed by each country for Ground vegetation assessment across the ICP Forests Level I network.

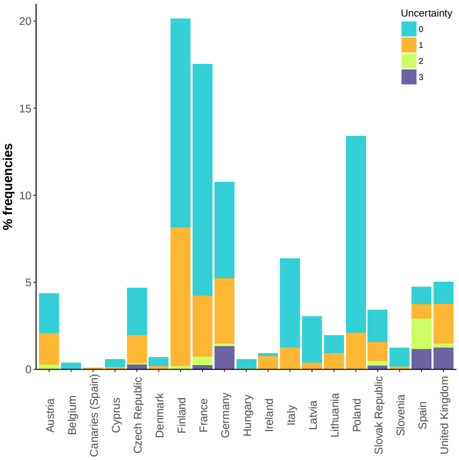
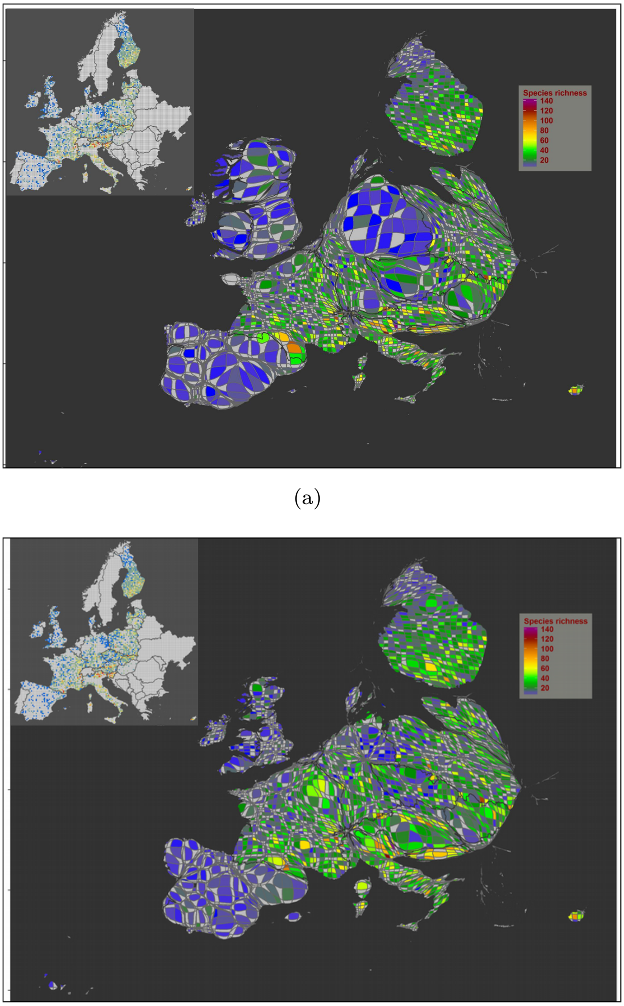


Figure 3.4: Plots uncertainty percentage frequencies by country of GVG data included in the ICP Forests Level I network.

smaller values of species richness, a result that was confirmed by the linear regression analyses ($F = 99.86$, $p < 0.001$) (Figure 3.6).

The uncertainty versus latitude profile highlighted how species richness changed across latitudes by considering different uncertainty values (Figure 3.7). Specifically, the vertical axis shows the latitude, the horizontal axis shows the uncertainty values while the color represents species richness values. Uniform species richness patterns corresponded to greater uncertainty values of 2 and 3, while heterogeneous patterns characterized smaller uncertainty values. The chart shows that when considering only smaller uncertainty values, species richness is greater for the lower latitude Mediterranean bioregion than for the higher latitude Boreal bioregion, (Figure 3.7).

We have provided a different approach for visualizing and evaluating data uncertainty using diffusion-based cartograms. In particular, we have displayed how data information changes with respect to degree of uncertainty. For this study, area size distor-



(a)

(b)

Figure 3.5: Cartograms of uncertainty data-related (cell distortion): (a) proportional to maximum uncertainty value (equal to 3), (b) proportional to the second highest uncertainty value (equal to 2). Cells color range according to plant species richness values. In the upper left a reference undeformed map.

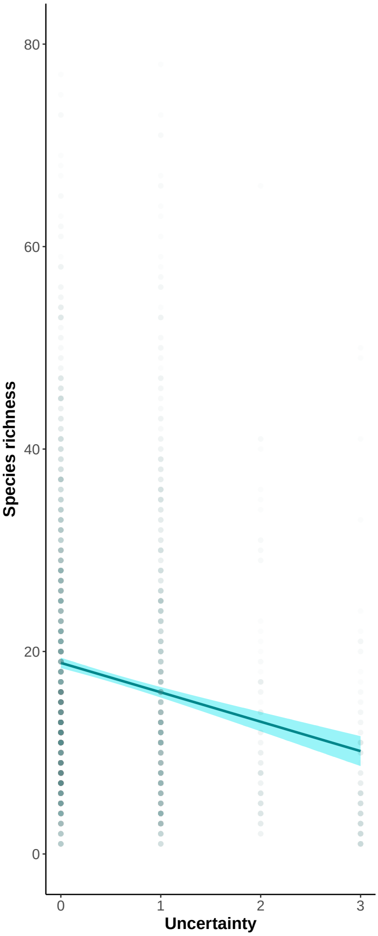


Figure 3.6: Relation between uncertainty data values and plant species richness records (R^2 = 0.4, F = 99.86, p < 0.001, with 95% confidence interval) of GVG data included in the ICP Forests Level I network.

tions highlighted that data collected in Spain, the United Kingdom and the German federal states of Berlin and Brandenburg have uniformly large uncertainty values, basing on survey period. Furthermore, such distortions corresponded to smaller species richness

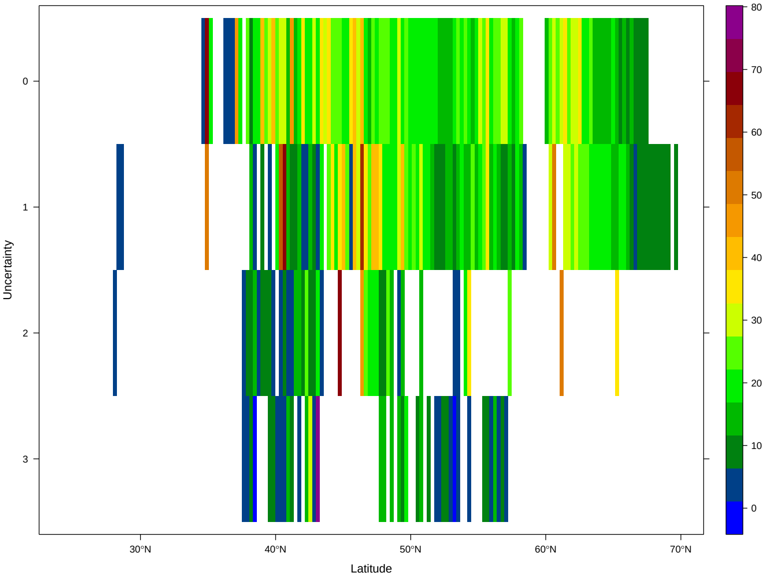


Figure 3.7: Latitude profile of species richness according to the different values of data uncertainty (minimum: 0; maximum: 3) of GVG data included in the LiBioDiv database under the ICP Level I network. The horizontal axis shows latitude degrees, and the vertical axis shows species richness average under different values of data uncertainty while the color represents species richness values. Lower plant species richness values occur at greater uncertainty values (3, on the lower part), while an increase of heterogeneity appears at smaller uncertainty values (0, at the bottom).

values as highlighted by grid color. The cartogram revealed the effects of lack of harmonized data collection protocols due to survey period and its effects on data quality. Plots surveyed in months with mean temperatures less than 10^oC and during autumn and winter had very small numbers of species recorded. The cartogram visualizations gave different perspectives on phenomena, visually depicted spatial relations, and emphasized trends that occur in isolated areas (i.e. the German federal states of Berlin and Brandenburg). The two cartograms showed different distortions for the two greatest uncertainty levels. The United Kingdom and the German federal states of Berlin and Brandenburg were greatly distorted by considering only uncertainty level 3, while Spain is largely expanded in both maps. These patterns were not easy to be read in Figure 3.4 that paradoxically contained more and complete information compared to a cartogram, but the use of both representations permits a clearer picture of the data pattern.

Understanding heterogeneity related to botanical data is a complex task that can change plans for managing data (Meyer et al., 2016). We found clear evidence of distinct species richness patterns across Europe, mainly due to non-random sampling (Rocchini et al., 2017) resulting from extreme differences in survey periods. Figure 3.7 tracks data limitations and suggests positive prospects for using GVG data. The use of all GVG data can lead to underestimates of vascular plant biodiversity in term of species richness across Europe. The underestimation penalizes the Mediterranean bioregion (due to the uncertainty of the Spanish data) which is one of the most complex and biodiverse regions (Blondel and Aroson, 1999). We have demonstrated that it is possible to reduce the effects of underestimation by considering different levels of uncertainty. However, our results may be sensitive to the choice of uncertainty evaluation (Figure 3.2). For example, different algorithms could be developed to assess uncertainty related to survey period (Rocchini et al., 2011), and it is possible that other solutions might produce additional information. Botanical uncertainty evaluation is complex because of environmental conditions and seasonal variability. In addition, because multiple factors affect both plant development and management of big data, defining common criteria for assessing uncertainty is not easy. We think our solution leads to improved data interpretation as demonstrated by the results and the connections between uncertainty and vascular plant species richness records. In particular, the negative relationship between survey season and species richness is a clear indication of data misinterpretation of the field manual. GVG data from the systematic ICP Forest Level I network (ex BioSoil-Biodiversity project) are a unique example of floristic data based on a rigourous, representative sampling design survey across Europe. Because data uncertainty could affect conservation strategies, our findings have important implications for assessing European forest biodiversity.

3.4 Conclusion

Density-equalizing maps or cartograms were used to depict data uncertainty. The cartograms allowed us to display geographic relationships among attributes and to highlight the effects of data uncertainty on data patterns. We highlighted data limitations and provided a method for understanding how different field data collection strategies may influence data analysis and statistical inference. In this case, data uncertainty led to underestimation of species richness, especially across Europe and for the Mediterranean bioregion. Botanical data are often heterogeneous depending on the observers, sampling designs and environment conditions. We demonstrated that cartograms are a useful tool for exploring and handling data, particularly when used together with other graphs and representations. Density-equalizing maps or cartograms give different representations of spatial phenomena that can be visualized and understood from different perspectives. Thus, effective alternative methods for evaluating and managing data uncertainty should be encouraged as a means of strengthening data analyses and their interpretation.

Acknowledgement

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Chapter 4

Paper III - Linking overstory and understory diversity indicators using European forest diversity patterns

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Abstract

Evidence-based policies are necessary to support the role of forests in the conservation of biodiversity and provisioning of ecosystem services. Assessing forest resources and their trends over time is required by many international agreements, and is a key step in developing policy strategies. Measures of tree species composition and structure are considered as possible biodiversity indices, despite ground vegetation usually having greater species richness than other forest strata. This study is aimed at evaluating relationships and connections between the overstory and understory

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by considering both stand structure and diversity indices, through a network analysis in which the system takes the form of a network or "graph". We used tree-related and ground vegetation data from 2789 plots collected as part of the BioSoil project through the ICP Forests Level I network and stored in the LI-BioDiv database. Additional information levels such as biogeographical region, forest management type and European Forest Type were also considered. In our study, differences among and within the different strata influence the correlation among overstory and understory diversity indices. The relationships among the structural and diversity measures showed few and weak correlations between overstory and understory diversity over the additional information levels. We demonstrated that at European level the overstory and understory indices are complementary, and they explain forest diversity components. Based on these results, overstory diversity cannot be considered as a proxy for the diversity of the entire forest plant community.

Keywords: *Stand structure; Biodiversity; Vegetation; Monitoring; Network analysis.*

4.1 Introduction

For many years, diversity conservation has been a key objective defined in international agreements such as the 1992 Convention on Biological Diversity (CBD) of the United Nations Conference on Environment and Development, the Ministerial Conference on the Protection of Forests of Europe (MCPFE, 2002) and the Montréal Process (2006). Agencies traditionally enforce regulation based on quantitative criteria (Noss, 1990) and for many years, the use of indicators has received increasing attention for supporting policy decisions and strategies, to set targets, to monitor their achievement and to define sustainable forest management practices (Lindenmayer, 1999; Eurostat, 2014). Indicators are summary measures of observed facts or phenomenon and do not necessarily contain all their information but hugely contribute to explain it (Noss, 1990; Eurostat, 2014). Biodiversity characterization concerns different components at several levels of organization (Noss, 1990) and identifying measurable indicators that can explain the overall status of biodiversity is an open challenge.

Since their inception early in the 20th century, National Forest Inventories (NFIs) have been the primary source of forest information. Most NFIs were designed primarily to assess status, trends and forest production values (Alberdi et al., 2010; Chirici et al., 2012), but in recent years have shifted their objectives to include wider multipurpose objectives. As reported by Chirici et al. (2012), in NFIs the data availability of ground vegetation, is limited to a few countries and additionally these data are collected in different ways, with consequent difficulties of harmonization. As a consequence, measures of tree species composition and stand structure have been considered as biodiversity indicators (MCPFE, 2002; Barbier et al., 2008), despite the fact that ground vegetation usually has greater species richness than other forest strata (Gilliam, 2007).

Overstory and understory layers are linked by reciprocal interactions (Gilliam and Roberts, 2003). Overstory composition influences understory dynamics by altering light regime and influencing the heterogeneity of soil fertility (Gilliam, 2007). An example of this ecological role is linked to foliar concentration of nutrients and rapid decomposition processes of the herbaceous species, ensuring an efficient recycling of nutrients (Gilliam, 2007). Other study demonstrate that understory species have a direct effect on the soil microbial and microfauna composition and substrate utilization (Mitchell et al., 2012). Conversely, the understory community can influence tree demography (seedlings, resprouts) and overstory composition (Gilliam, 2007). Studies from northern Sweden showed that forest understory components can drive forest regeneration, belowground properties, and long-term forest succession (Nilsson and Wardle, 2005) while studies conducted on *Pinus* stands demonstrated that understory vegetation which invaded during the process of secondary succession had negative physiological effects on the overstory pine needles through the competition of roots for water and nutrients (Kume et al., 2003).

Several studies have focused on the relationship between overstory and understory layers with results that range widely (Burrascano et al., 2011). Neumann and Starlinger (2001), compared different indices of stand structure and species diversity and highlighted that only a few and weak correlations exist between species diversity and the diversity of stand structure. Barbier et al. (2008), reviewed the influence of tree species composition on understory vegetation diversity and pointed out the difficulties to make generalizations due to the conflicting results. Conversely, Giorgini et al. (2015) confirmed the role of woody species observed by forest inventories to predict compositional species diversity in temperate forests. Some studies found weak relations between an increase in stand structure diversity and total plant diversity (Lindgren et al., 2006). More recently, Gao et al. (2014), while analyzing whether stand structural types could be indicators of plant species diversity, found that mature stands with a complex structure had the greatest plant species diversity. Despite these results, the use of overstory diversity as a proxy for forest biodiversity is still an unresolved issue (Barbier et al., 2008; Giorgini et al., 2015).

A wide range of methods have been developed to understand the relationship between overstory and understory layers: from simple correlation (Neumann and Starlinger, 2001) to more complex models (Barbier et al., 2009; Gao et al., 2014). Network analysis is used to study a wide range of systems, especially social networks (Newman, 2002), although its use by ecologists is limited to a few cases (Fuller et al., 2008). Network analysis quantifies the interrelations of a system by exploring associations between objects and as result the system takes the form of a network or "graph" (Newman, 2002).

We used data from the Li-BioDiv database collected in the BioSoil project from 2005 to 2008, under the ICP Forests Level I network. This database represents a unique example of pan-European data containing tree layer and ground vegetation information about European forests and additional information levels such as biogeographical region (EEA, 2016), forest management (WGFB, 2011), and forest type (EEA, 2006). Although several studies comparing overstory and understory diversity indices have been reported, to our knowledge no study has been conducted over a pan-continental scale and over a wide range of ecological conditions. The aims of this study are fourfold: (i) to explore the connections between overstory and understory diversity indices across European forests, (ii) to understand if this connection ranges across different levels (biogeographical region, forest management, and forest type) and (iii) to assess whether overstory diversity and structural indices could be a proxy for overall plant biodiversity and (iv) to provide a basis for the knowledge necessary for policy-makers, and to suppport future forest biodiversity systems.

4.2 Material and methods

4.2.1 Data

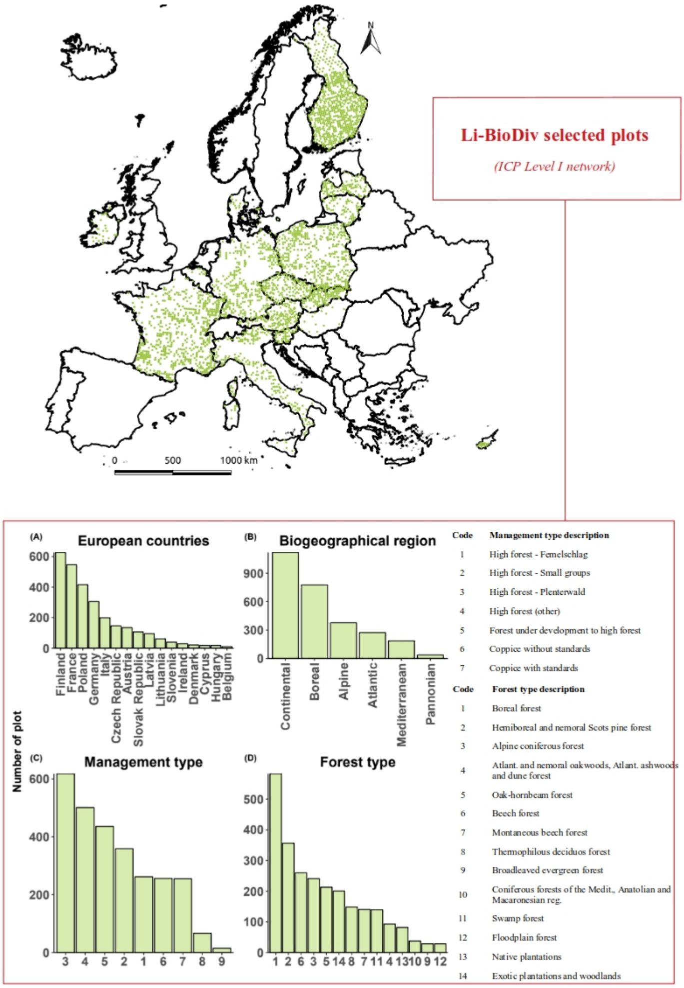
We used data collected in the BioSoil-Biodiversity project (LI-BioDiv database), a part of the International Cooperative Programme on Assessment and Monitoring of Air pollution effects on Forests (ICP Forests; `www.icp-forests.org`). Diameter at breast height (DBH > 10 cm), deadwood (DWD with diameter > 10 cm), canopy closure (expressed in percentage classes: 1 (open sky), 2 (1-25%), 3 (25-50%), 4 (50-75%), 5(>75%)) and ground vegetation data (as species list and species cover) surveyed in a 400 m^2 plot, were chosen as variables. Detailed information about data collection are reported in the BioSoil field manual (Aamlid et al., 2007; WGFB, 2011). From a total of 3311 georeferenced plots, systematically distributed across Europe on a 16 x 16-km grid, 2789 plots were selected according to (Galluzzi et al., 2018) because they included records for all the chosen variables. Additional plot information levels such as biogeographical region (EEA, 2016), forest management (WGFB, 2011) and forest type (according to EEA (2006)) were also available for each plot (Figure 4.1).

4.2.2 Data analysis

Computation and statistical analyses were conducted using the R-statistical framework (R-Core-Team, 2017). Diversity patterns across the information level about the plot were firstly assessed, then the overstory and understory layer were studied considering both structural and diversity-related variables separately. Species richness pattern variation within each information level was examined through a sample-based standard (exact) rarefaction curve that is commonly used to evaluate species variation in data with different sampling intensities (Eq. 4.1) (Gotelli and Colwell; Clarke et al., 2011):

$$S_n = K - \binom{N}{n} \sum_{i=1}^{K} \binom{N - N_i}{n} \tag{4.1}$$

where $\mathrm{S_n}$ is the expeted number of species (with n = 1,...,N), K is the total number of observed species, N is the number of sampling units and $\mathrm{N_i}$ is the number of sampling units containing at least one individual of observed species. Continuous number of species of sample relationships were fitted by using a log-log transformation (Koellner et al., 2004). The overstory was characterized through dimensional, dendrometric and species diversity indeces (Tab. 4.1). Indices commonly used in NFIs (Corona et al., 2011) were calculated, such as aboveground biomass (kg $\mathrm{ha^{-1}}$)(AGB; by using the European allometric models; `http://www.globallometree.org/`), volume of deadwood ($\mathrm{m^3}$ $\mathrm{ha^{-1}}$; DWD; cylinder formula and truncated cone formula for snag),



Code	Management type description
1	High forest - Femelschlag
2	High forest - Small groups
3	High forest - Plenterwald
4	High forest (other)
5	Forest under development to high forest
6	Coppice without standards
7	Coppice with standards

Code	Forest type description
1	Boreal forest
2	Hemiboreal and nemoral Scots pine forest
3	Alpine coniferous forest
4	Atlant. and nemoral oakwoods, Atlant. ashwoods and dune forest
5	Oak-hornbeam forest
6	Beech forest
7	Montaneous beech forest
8	Thermophilous deciduos forest
9	Broadleaved evergreen forest
10	Coniferous forests of the Medit., Anatolian and Macaronesian reg.
11	Swamp forest
12	Floodplain forest
13	Native plantations
14	Exotic plantations and woodlands

Figure 4.1: Plots selected from the Li-BioDiv database and their distribution across the following information levels: biogeographical region (EEA, 2016), forest management (WGFB, 2011), forest type (EEA, 2006).

arithmetic mean of DBH (m.D), standard deviation of DBH (sd.d), coefficient of variation of DBH (Cv.d), Shannon index of DBH (H'.d) (Shannon, 1948), Simpson index of DBH (D.d;) (Simpson, 1949), Gini coefficient of DBH distribution based on basal area (Gin.d) (Gini, 1921). Stand density per hectare (N.ha), basal area (BA) (m^2 ha^{-1}; m.B), Shannon index of BA (H'.B) (Shannon, 1948), Simpson index of BA (D.B;), Pielou index of BA (Pielou, 1969), Margalef index on BA (Mi.B) (Clifford and Stephenson, 1975) and Berger-Parker index on BA (Bpi.B) (Berger and Parker, 1970) were calculated as well according as measures of structural diversity. The species richness of the tree layer (SRtr) was investigated to define tree species composition for each plot.

Understorey floristic diversity was evaluated through the most widespread and popular indices (Magurran, 2004)(Tab. 4.1): species richness (SR), Shannon index (H') (Shannon, 1948), Simpson index (D) (Simpson, 1949), Margalef index (MI) (Clifford and Stephenson, 1975), Fisher's alpha (Fish) (Fisher et al., 1943), Menhinick's index (Dmn) (Whittaker, 1972), Berger-Parker index (Bpi) (Berger and Parker, 1970), Gini coefficent (Gin) (Gini, 1921) and Pielou index (Pielou, 1969). Pearson's correlation coefficient was used to test relationships among the chosen variables. Weighted network visualization and analysis (Epskamp et al., 2018) were used to understand the relationship between the two groups of indices within the three information levels. Indices were graphed in a two-dimensional space, using a modified version of the force-embedded algorithm (900 iterations) defined by Fruchterman and Reingold (1991), where the distance between the indices increases with the decreasing the strength of the relationship.

Code	**Description**	**References**
	Overstory diversity indices	
AGB	Above ground biomass (Kg ha^{-1})	
DWD	Volume of deadwood (m^3 ha^{-1})	
m.D	Arithmetic mean of DBH	
Sd.D	Standard deviation of DBH	
CvD	Coefficent of variation of DBH	
H'D	Shannon index of DBH	Shannon (1948)
D.D	Simpson index of DBH	Simpson (1949)
N.ha	Stand density (n stem ha^{-1})	
Gin.D	Gini coefficent	Gini (1921)
B.ha	Basal area (BA) (m^2 ha^{-1})	
D.B	Simpson index of BA	Simpson (1949)
H'.B	Shannon index of BA	Shannon (1948)
J.B	Pielou index of BA	Pielou (1969)
Mi.B	Margalef index on BA	Clifford and Stephenson (1975)
Bpi.B	Berger-Parker index on BA	Berger and Parker (1970)
SRtr	Species richness of tree layer	
Can	Canopy cover (percentage classes)	
	Understory diversity indices	
H'	Shannon index	Shannon (1948)
D	Simpson index of BA	Simpson (1949)
SR	Species richness of tree layer	
MI	Margalef index	Clifford and Stephenson (1975)
Fish	Fisher's alpha	Fisher et al. (1943)
Dmn	Menhinick's index	Whittaker (1972)
Bpi	Berger-Parker Dominance index	Berger and Parker (1970)
Gini	Gini coefficent	Gini (1921)
J	Pielou index	Pielou (1969)

Table 4.1: Applied diversity meausures

4.3 Results

The mean number of tree species per plot was 2.4 with a miniumum of 1 and a maximum of 11, while the mean number of understory species per plot was 18.73 with a minimum of 1 and a maximum of 98. According to the pattern of rarefaction curves, the three information level classes had different values of species richness. The Mediterranean biogeographical region, followed by the Alpine biogeographical region had the greatest number of species. The Mediterranean region showed a continuously rising pattern which was more obvious than others, while the boreal region had the smallest rising pattern (Figure 4.2(a)). The High Forest - Plenterwald class of management type had the greatest values of species richness, followed by coppice stands, while the other high forest classes had the smallest values. However, all the classes had a continuously rising pattern (Figure 4.2(b)). Thermophilous deciduous forests, Alpine forests, Broadleaved evergreen forests and Montaneous beech forests are the forests types with the greatest values of species richness (Figure 4.2(c)). Swamp forest and Boreal forest types had the smalles species richness values. Although they had small species richness values, Native plantations and Exotic plantations and woodlands forest type had greater continuously rising patterns (Figure 4.2(c)).

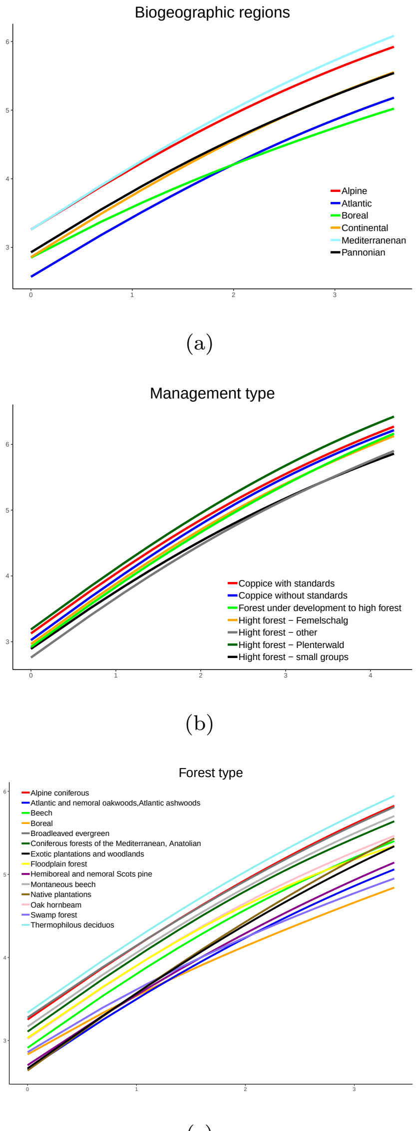
No statistically significant correlations were found between the overstory diversity indices and the understory diversity indices when considering the overall data (Figure 4.3(a), 4.3(b)).

The results of the network visualization and analysis showed that interactions changed among and within the information levels and that the relationships between the two groups of indices were the weakest ($p < 0.05$) in all cases. Considering all the connections (Tab. 4.2, Appendix 4.A, 4.B, 4.C), basal area (m^2 ha^{-1}) (B.ha, 14.60%), canopy cover (Can, 12.40%) and species richness of tree layer (SRtr, 12.95%) were the indices with the largest number of interactions with the understory indices followed by the standard deviation of diameter (Sd.D, 9.64%) coefficient of variation of DBH (CvD, 9.09%) and Above ground biomass (AGB, 9.37%) (Tab. 4.2). Gini coefficient (Gin.D) and Margalef index (Mi.B) showed interactions with the understory indices although to a lesser extent (6.61% and 6.89% respectively).

In the Alpine and Continental biogeographical regions no interactions were found between overstory and understory indices, while for the other biogeographical regions classes B.ha, AGB, and CvD were the indices with the greater number of interactions (27.69%, 16.92%, 16.92% respectively) (Appendix 4.A; Tab. 4.2). Regarding management type classes, no interaction between the two indices groups was found in the High forest - Femelschlag class. Can (14.58%), SRtr (15.63%), Sd.D (12.50%), Gin.D (12.50%) and B.ha (10.42%) were the indices that showed the greater number of interactions in the other management classes (Appendix 4.B; Tab. 4.2). In the forest type classes Can (13.59%), SRtr (14.56%) and B.ha (12.14%) had the greater number of interactions followed by Sd.D (10.68%) and AGB (11.17%) (Appendix 4.C; Tab. 4.2). The broadleaf evergreen forest type had the only class where the two groups of indices were less distant in the bi-dimensional space, albeit with weak interactions (Appendix 4.C). A summary of positive and negative connections across the three information levels is presented in Appendix 4.D.

		Percentage of link with the understory diversity indices			
Code	**Description**	**Overall**	**Biogeographical region**	**Management type**	**Forest type**
Can	Canopy cover (percentage classes)	12,26	4,62	14,58	13,59
SRtr	Species richness of tree layer	13,90	9,23	15,63	14,56
Sd.D	Standard deviation of DBH	9,54	1,54	12,50	10,68
CvD	Coefficent of variation of DBH	8,99	16,92	3,13	9,22
Gin.D	Gini coefficent	6,54	7,69	12,50	3,40
B.ha	Basal area (m^2 ha^{-1})	14,44	27,69	10,42	12,14
AGB	Above ground biomass (t ha^{-1})	9,26	16,92	0,00	11,17
N.ha	Stand density (n stem ha^{-1})	3,81	4,62	5,21	2,91
Mi.B	Margalef index on BA	6,81	4,62	8,33	6,80
m.D	Arithmetic mean of DBH	4,90	4,62	5,21	4,85
H'D	Shannon index of DBH	4,09	0,00	1,04	6,80
DWD	Volume of deadwood (m^3 ha^{-1})	1,63	1,54	0,00	2,43
J.B	Pielou index of BA	0,27	0,00	0,00	0,49
Bpi.B	Berger-Parker index on BA	1,36	0,00	3,13	0,97
H'.B	Shannon index of BA	2,18	0,00	8,33	0,00
D.D	Simpson index of DBH	0,00	0,00	0,00	0,00
D.B	Simpson index of BA	0,00	0,00	0,00	0,00

Table 4.2: Percentage of link between overstory indices with understory indices in the network analysis. The percentage of link are presented according with the three information level and considering them all together.

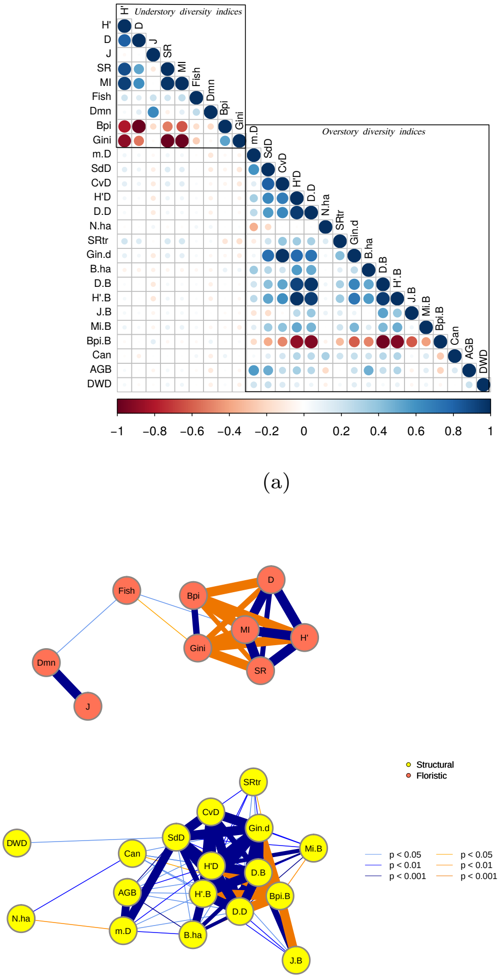


(a)

(b)

(c)

Figure 4.2: Pattern of the rarefaction curve (log-log transformation) of the vascular plant species from the Li-BioDiv database for the three information level: biogeographical region (EEA, 2016) (a), forest management (WGFB, 2011) (b), forest type (EEA, 2006) (c).



(a)

(b)

Figure 4.3: Results of Pearson's correlation (a) and Network visualization and analysis (b) considering all the data. In (a) Blue means positive relation, and red negative relation.

4.4 Discussion

The differences among biogeographical regions in terms of species richness (Fig. 4.2) are most probably due to different pressures and relative impacts, such as habitat alteration and loss of connectivity due to agriculture, forestry, hunting, and infrastructures (EEA, 2002). At the same time, a less marked difference is due to a continual change in which regions became less rich in species and more uniform (EEA, 2002). The greatest number of vascular plants occurred in the Mediterranean biogeographical region, followed by the Alpine biogeographical region as has also been reported for other studies (Nagy et al., 2003; Fenu et al., 2017).

Forest management can represent one of the primary drivers of biodiversity, shaping forest structure and, consequently, understory diversity (Calster et al., 2008; Kutnar et al., 2016). The effects of management vary depending on the type of silvicultural and local condition. Our results, showed the dominance of species richness in coppice systems while High Forest - Plenterwald class of management type had the greatest values of species richness. As has already been shown, coppices are characterized by large species richness due to the dominance of early successional species (Decocq et al., 2004). However, the number of species changes along the temporal gradient (i.e. as age of coppice increases) and light-demanding oligotrophic species reduce their abundance with an increase in forest specialist species (Decocq et al., 2004; Scolastri et al., 2017).

The greater species richness of the thermophilous deciduous forest is mainly due to mild climatic conditions that determine the predominance of mixed deciduous and semi-deciduous forest of thermophilous tree species because they provide a variety of different niches for other vascular species (EEA, 2006). The great species richness of Alpine coniferous forest, despite the cold and harsh climate, can be explained by the management type i.e. selection cutting which creates small gaps (EEA, 2006), determining large turnover in species composition and structural condition variation but also to the presence of endemic and relict post-glacier species (Casazza et al., 2008). Broadleaved evergreen forests have an important species richness, where anthropogenic influences such as fire could determine species turnover (EEA, 2006). On the opposite side, the Boreal forest, where temperature and length of the growing season are the main climatic variables which determine species composition and turnover (EEA, 2006), has smaller values of species richness. We must consider that measures based only on species richness can lead to misunderstanding, and that the relations between tree species richness and overall vascular species richness must take into account the scale dependence of this relations (e.g. Giorgini et al. (2015) and Campetella et al. (2016)).

In our study, differences among and within the information levels (biogeographical region, forest management, European Forest Type) influence the correlations between overstory and understory diversity indices (Appendix **??**). Although the connections between overstory and understory diversity indices are weak, we found that canopy cover and species richness of the tree layer are the indices for which the understory layer is most sensitive. This factor mainly depends on silvicultural methods that create conditions for the survival and growth of "desirable" plants according to management objectives (Jennings et al., 1999; Barbier et al., 2008). Canopy cover, and consequently the light regime, is one of the main determinants of the microhabitat within the forest, influencing vegetation composition (Jennings et al., 1999). Stands with species richness of the tree layer and generally mixed stands, by increasing resource diversity, host a more heterogeneous understory composition than pure stands (Hill, 1992; Barbier et al., 2008). Regarding the biogeographical region and forest type levels, tree abundance quantified as basal area (m^2 ha^{-1}) and AGB (Kg ha^{-1}), had an important connection with the overstory layer (percentage of link respectively: 27.69%, 16.92% for biogeographical region and 12.14%, 11.17% for forest type; Figure 4.2). Numerous studies have pointed out the effect of tree abundance on understory vegetation diversity (Barbier et al., 2008). However, this effect could be interpreted as an influence on light regime which may reduce system complexity by favoring ruderal species (Balandier et al., 2006). Along the management type level, structural indices (standard deviation of diameter, the coefficient of variation of diameter and Gini coefficient) in most cases has a positive connection with the understory layer. It is known that forest management determines the spatial structure of stands (Pretzsch, 2010) and it has been demonstrated to have a substantial and positive influence on understory plant diversity (Lenière and Houle, 2006; Burrascano et al., 2011). However, structural diversity is not the only factor that influences understory composition together with soil and pH which may influence understory vegetation too Lenière and Houle (2006). Hence, understory diversity sometimes could be more closely related to silviculture operations (Barbier et al., 2008), sometimes to environmental conditions (Lenière and Houle, 2006) and sometimes to overstory tree species. Forests, as dynamic systems, depend on the chain of past events that decisively influence their future behavior (Pretzsch, 2010).

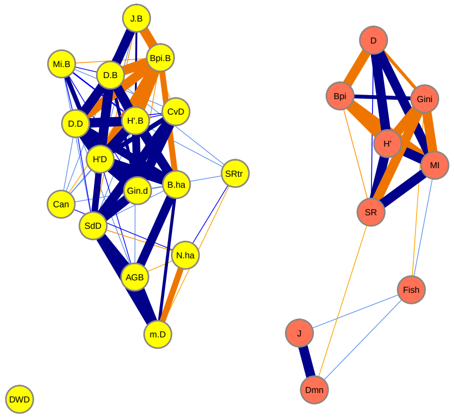
Based on our study, the use of overstory diversity indices as surrogates for total plant community diversity is not appropriate to support decision processes at the European scale. As affirmed by Noss (1990), a good indicator should be sensitivity to changes, wide applicability, and easy to collect. We demonstrated that at the European scale none of the studied indices has, alone, the required characteristics because they explain only a part of forest diversity in specific environmental conditions. European forests vary with respect to environmental conditions, forest management methods, use, infrastructures, economic challenge and institutional framework (FAO, 2017). The importance of long-term ecological monitoring has been highlighted in several studies that were relevant only for given situations (Lindenmayer and Likens, 2010). Stronger national efforts should be encouraged to promote collection of diversity data at different levels, and a set of indices should be identified according to smaller scale and environmental and ecological circumstances.

4.5 Conclusions

Reliable forest information at different scales is needed to support policy-makers and stakeholders to enhance the role of forest in the conservation of biodiversity and in reducing the impact of climate change. Our work points out that: i) overstory diversity cannot be used as a proxy for the entire forest plant community, traditionally data collected by NFIs about the overstory structure and diversity explain only a part of the entire forest diversity; ii) results range widely across biogeographical regions, management types, and European Forest Types and may be due to the local environmental condition and iii) there is the need to consolidate national forest monitoring, and to define target indicators oriented towards specific objectives. Comparable and consistent information are key elements to assess changes in forest characteristics. Thus, an update of the systematic process of gathering and collecting forest data should be strongly encouraged in the light of a diverse perspective of forest assessment that includes information from decision-making to site management.

4.A Biogeographical region

Alpine



Atlantic

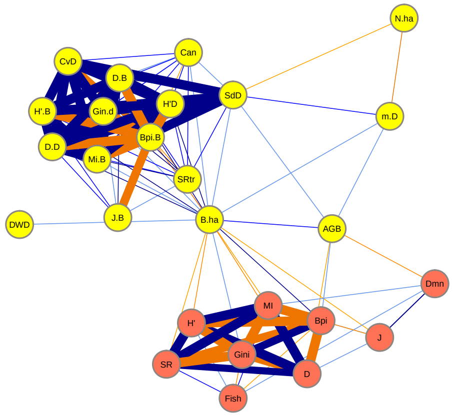


Figure 4.4: Network visualization and analysis results of biogeographical region level

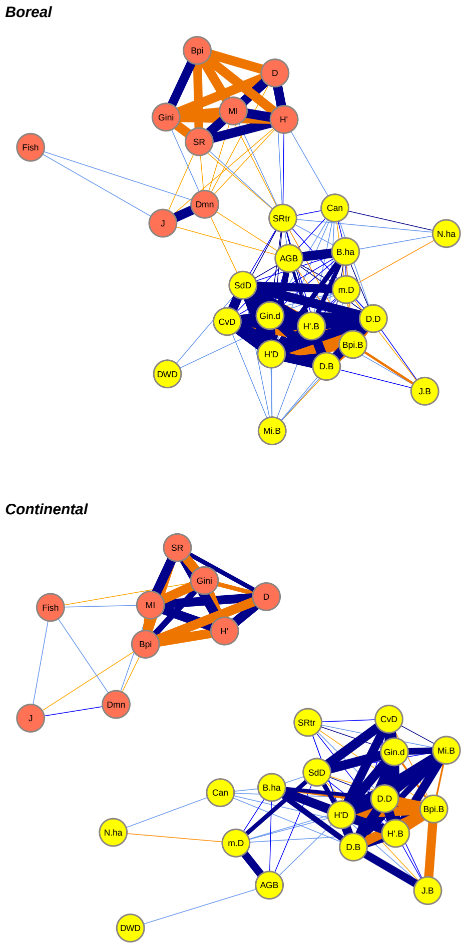


Figure 4.5: Network visualization and analysis results of biogeographical region level

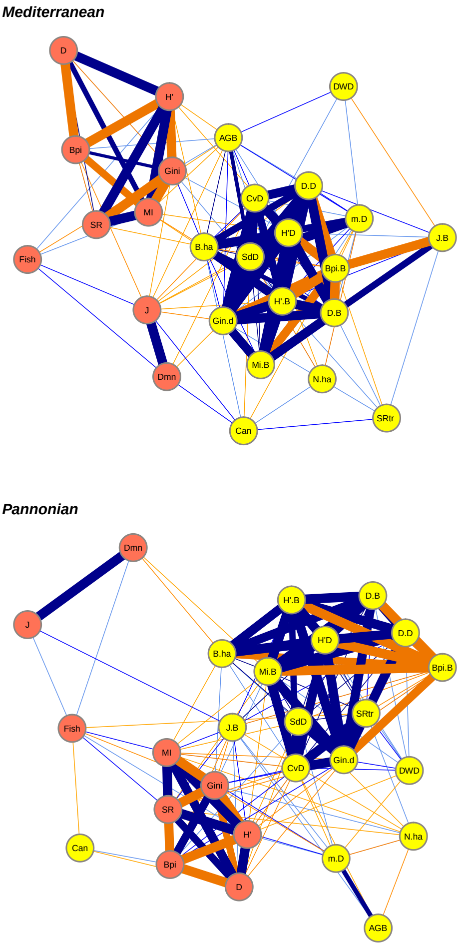
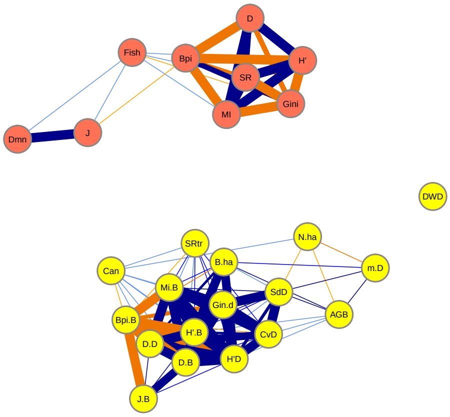


Figure 4.6: Network visualization and analysis results of biogeographical region level

4.B Management type

High forest − Femelschlag



High forest − Small groups

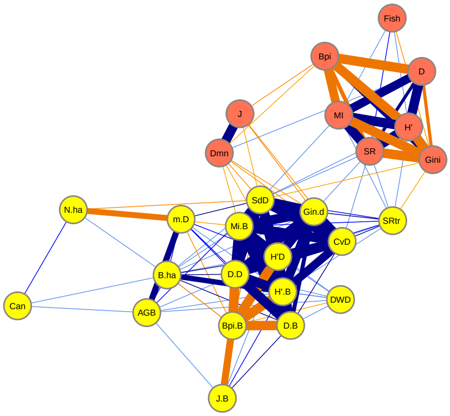


Figure 4.7: Network visualization and analysis results of management type level

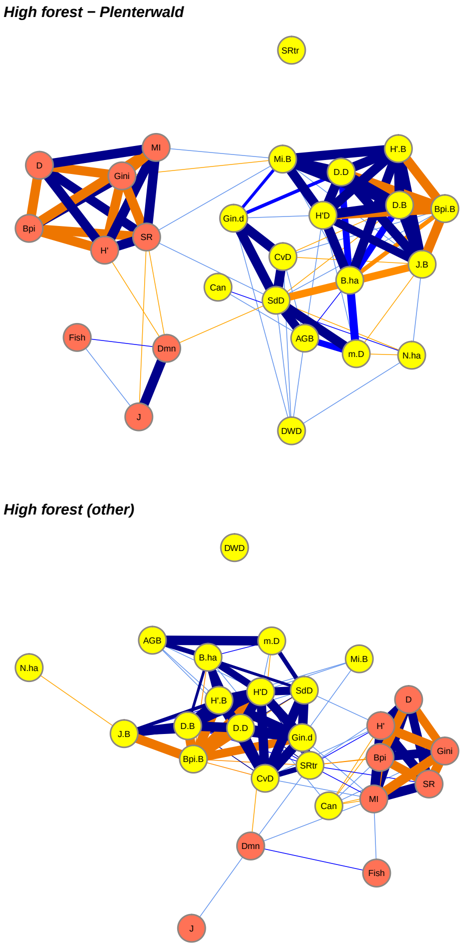
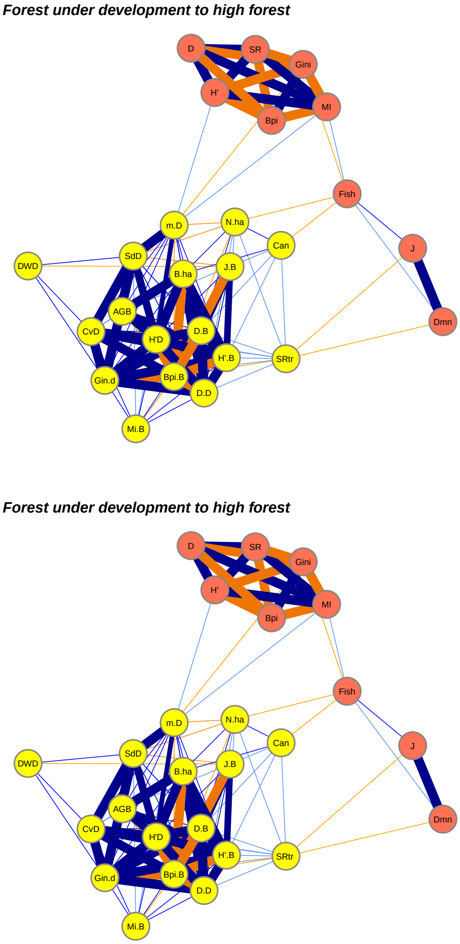


Figure 4.8: Network visualization and analysis results of management type level



Forest under development to high forest

Forest under development to high forest

Figure 4.9: Network visualization and analysis results of management type level

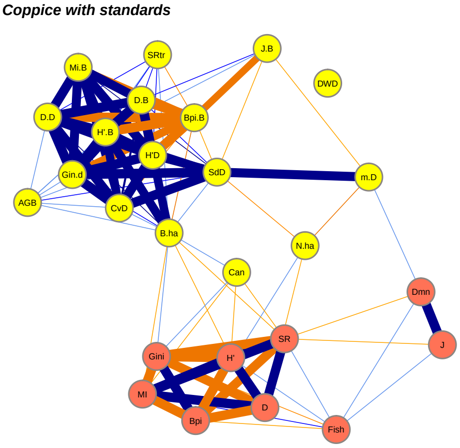
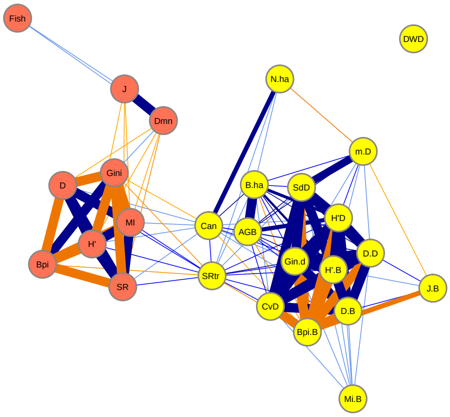


Figure 4.10: Network visualization and analysis results of management type level

4.C European Forest Type

Boreal forest



Hemiboreal and nemoral Scots pine forest

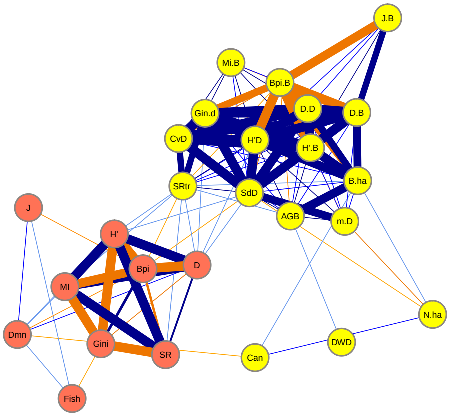
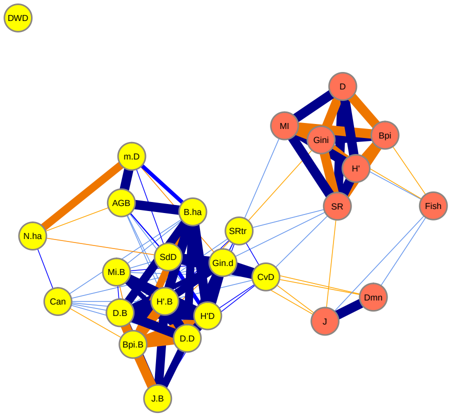


Figure 4.11: Network visualization and analysis results of European Forest Type level

Alpine coniferous forest



Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest

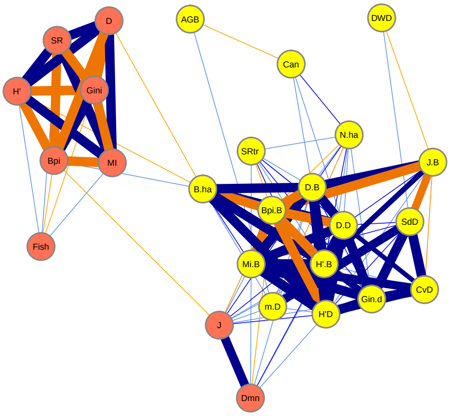
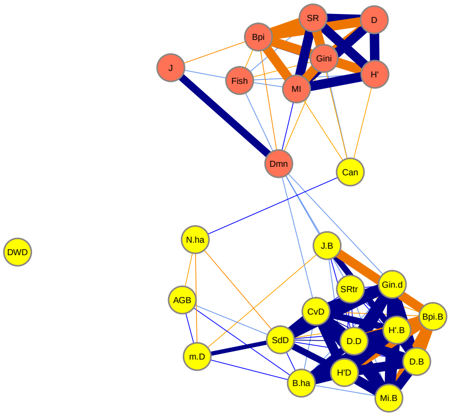


Figure 4.12: Network visualization and analysis results of European Forest Type level

Oak−hornbeam forest



Beech forest

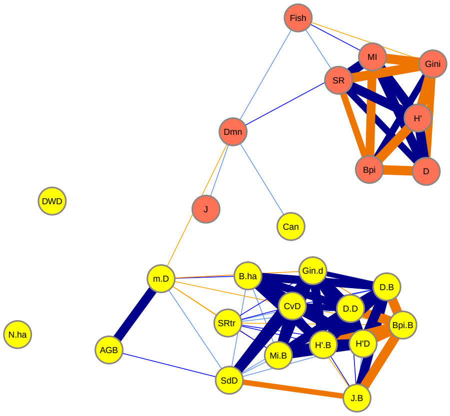
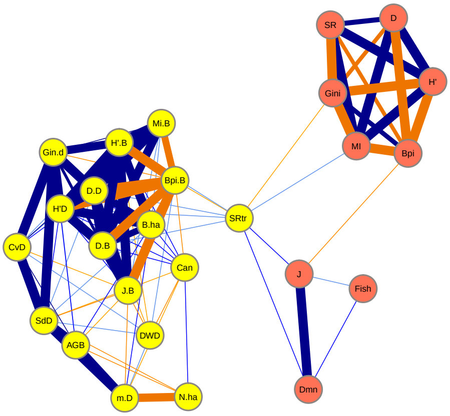
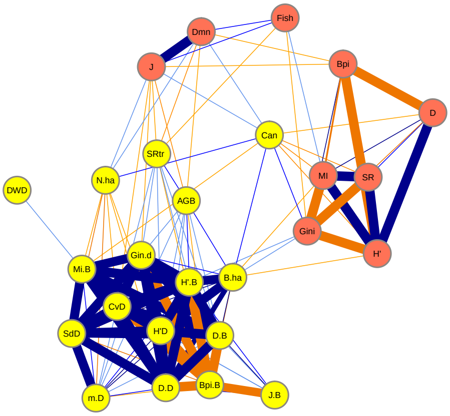


Figure 4.13: Network visualization and analysis results of European Forest Type level



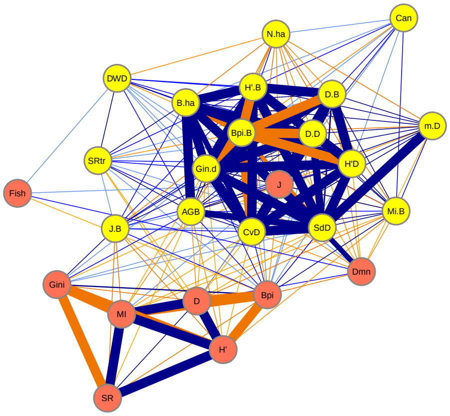
Montaneous beech forest



Thermophilous deciduos forest

Figure 4.14: Network visualization and analysis results of European Forest Type level

Broadleave evergreen forest



Coniferous forests of the Medit., Anat. and Macar. regions

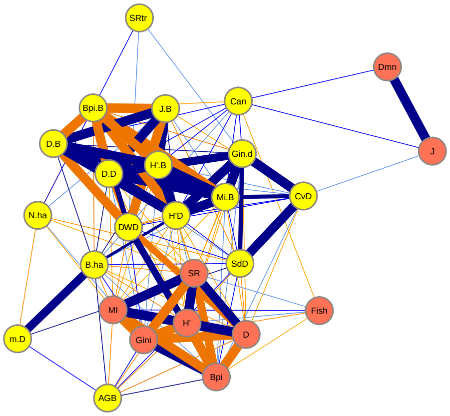
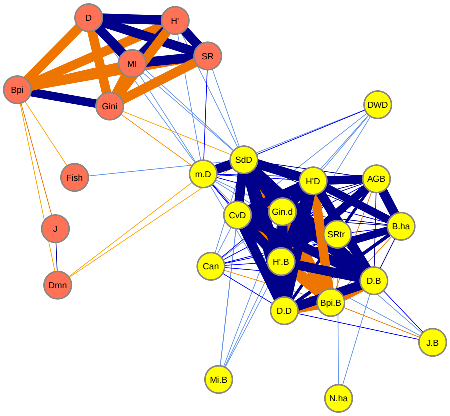


Figure 4.15: Network visualization and analysis results of European Forest Type level

Swamp forest



Floodplain forest

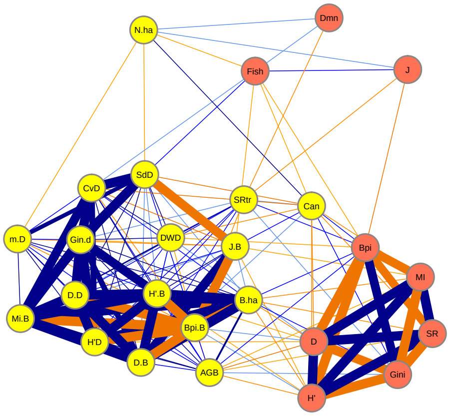
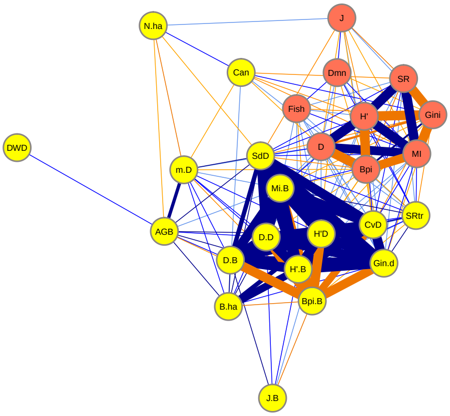


Figure 4.16: Network visualization and analysis results of European Forest Type level

Native plantations



Exotic plantations and woodlands

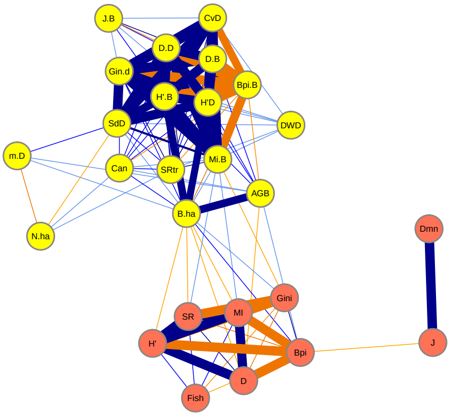


Figure 4.17: Network visualization and analysis results of European Forest Type level

4.D Summary of connection

		Percentage of link with the understory diversity indices					
		Biogeographical region		Management type		Forest type	
Code	**Description**	**Positive**	**Negative**	**Positive**	**Negative**	**Positive**	**Negative**
Can	Canopy cover (percentage classes)	0,00	12,00	18,87	9,30	16,84	10,81
SRtr	Species richness of tree layer	5,00	16,00	11,32	20,93	14,74	14,41
Sd.D	Standard deviation of DBH	2,50	0,00	11,32	13,95	9,47	11,71
CvD	Coefficent of variation of DBH	20,00	12,00	1,89	4,65	10,53	8,11
Gin.D	Gini coefficent	10,00	4,00	16,98	6,98	2,11	4,50
B.ha	Basal area (m^2 ha^{-1})	27,50	28,00	11,32	9,30	14,74	9,91
AGB	Above ground biomass (t ha^{-1})	20,00	12,00	0,00	0,00	15,79	7,21
N.ha	Stand density (n stem ha^{-1})	5,00	4,00	5,66	4,65	1,05	4,50
Mi.B	Margalef index on BA	7,50	0,00	7,55	9,30	6,32	7,21
m.D	Arithmetic mean of DBH	0,00	12,00	3,77	6,98	2,11	7,21
H’D	Shannon index of DBH	0,00	0,00	1,89	0,00	5,26	8,11
DWD	Volume of deadwood (m^3 ha^{-1})	2,50	0,00	0,00	0,00	1,05	3,60
J.B	Pielou index of BA	0,00	0,00	0,00	0,00	0,00	0,90
Bpi.B	Berger-Parker index on BA	0,00	0,00	1,89	4,65	0,00	1,80
H’.B	Shannon index of BA	0,00	0,00	7,55	9,30	0,00	0,00
D.D	Simpson index of DBH	0,00	0,00	0,00	0,00	0,00	0,00
D.B	Simpson index of BA	0,00	0,00	0,00	0,00	0,00	0,00

Table 4.3: Percentage of postive and negative link between overstory indices with understory indices in the network analysis.

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Chapter 5

Paper IV - Stand structure and plant diversity changes along a temporal gradient in thermophilous deciduous coppice forests under alternative management options

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Abstract

Forest management is one of the primary drivers of biodiversity, causing changes in species richness, evenness and composition of the different stand layers. Coppicing is a traditional management form that substantially modifies the vertical structure, microclimate and light conditions of the forest. Coppice abandonment and coppice conversion gradually leads to transformation to high forests and to consequent changes in stand structure, species composition and functional processes. In southern Europe, thermophilous deciduous forests are widespread and subject to coppicing since

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long time, but the dynamic changes in structure and plant diversity following their abandonment or conversion are still poorly known. We tested the effects of alternative managements in this forest type by comparing structure and plant diversity along a four-stages temporal gradient of stand development under similar site conditions: young coppice ($\leq$ 10 years); adult coppice (10-36 years); coppice in conversion to high forest (36-60 years) thinned less than 15 years ago; coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned. After examination of stand structure variables of the overstorey ($>$1.30 m), alpha- and beta-diversity measures were used to compare plant diversity and composition of the understorey ($<$1.30 m) in the four stages and to test the relationships with descriptors of the dominant layer. Indicator species analysis and Ellenberg values were used to investigate the floristic and ecological characteristics of the stands. Greater species richness was found in young coppices while higher structural diversity and the presence of specialist and ancient forest species characterized older coppices in conversion to high forest. Significant negative relationships were found between species richness and basal area, species richness and stand age and between Shannon index and age, while no significant relation was found between Shannon index and basal area. Based on Indicator species analysis, both ruderal and nemoral (e.g. *Melica uniflora*) species were found to be significantly associated with young coppice, while no nemoral species was found to be significantly associated with coppice in conversion to high forest where such species were present with low density. Overall, frequent forestry operations changed the complexity and the ecological value over time, limiting the spread of forest specialist species and secondary woody species. Implications for forest management of thermophilous deciduous forests are discussed in order to promote conservation-oriented management of this forest type.

Keywords: *coppice, high forest, species richness, struc-*

tural diversity, forest management, plant community.

5.1 Introduction

The processes which control temporal and dynamic plant diversity changes on short time scales are mainly driven by disturbance regimes (Denslow, 1980; Roberts and Gilliam, 1995) and plant adaptive responses (Grime, 1979). Forest management is an anthropogenic disturbance acting on environmental conditions, in particular on light regime (Decocq et al., 2004). In turn, this can influence temperature, humidity, soil moisture and other properties (Larcher, 2003), hence determining resource availability and activating dynamic processes (Decocq et al., 2004). Responses to management-related disturbances lead to stand dynamic changes, which vary depending on species-specific plant strategies. Hence management can represent one of the primary drivers of biodiversity, affecting forest structure and, consequently, plant species richness, evenness and composition in the different forest layers (Calster et al., 2008; Kutnar et al., 2016). However, the direction and magnitude of effects vary depending on the type of management and these are not always completely understood.

Coppicing is a traditional method of management mainly for firewood production, still widely adopted in Europe (UN/ECE-FAO, 2000), which substantially alters the vertical structure and light conditions of the forest at short-term intervals. In Italy, coppice-with-standards is practiced in over 50% of the thermophilous deciduous forests dominated by *Quercus* sp. Compound coppice with cohorts of uneven-aged standards forming a discontinuous canopy cover is less widely adopted (6%). Coppice in conversion to high forest is a small proportion (2%), while high forests are more widely distributed (22%) (INFC, 2005).

The effects of coppicing on the diversity and functionality of woodlands is currently a relevant issue, since the goal of forest management is increasingly shifting towards a multifunctional view where biodiversity conservation is a priority (Lindenmayer et al., 2000; Kovac et al., 2017). Early studies on the impact of coppicing on herb-layer diversity found rather dynamic effects that depended on intensity, rotation length, and time span after disturbance (Barkham, 1992; Martin and Martin, 1993; Mason and MacDonald, 2002). In France, Decocq et al. (2004) found that selective cutting with rotation period of less than 50 years and coppice-with-standards in woodlands with *Quercus robur* and *Carpinus betulus* had negative effects on both structural and functional plant diversity because the forest was permanently disturbed by a management type not matching the natural disturbance regime. Decline of those forest species that are not able of quick post-disturbance recovery was also detected (Decocq et al., 2004). Finally, cessation of coppicing in downy oak stands in France was found to increase compositional similarity to the vegetation of undisturbed woodlands (Debussche et al., 2001). Other studies in beech coppices in Italy revealed a decreasing species richness in older stand ages resulting from a decline in non-forest species only partially balanced by an increase in understorey forest specialists (Campetella et al., 2016; Scolastri et al., 2017). Furthermore, total species richness was negatively correlated with the height of standards and the number of dominant trees, though beech understorey specialists were significantly more numerous in abandoned plots (Bartha et al., 2008). In addition, coppice cutting was shown to cause a reduction in the amount of ecological niches and trophic space available to herbaceous and bird diversity in the vertical space (Melini, 2006). In the Mediterranean Region, coppice management underwent a decline during the 20th century, when conversion to high forest was increasingly advocated due to socio-economic changes. However, it gained renewed importance in the last decades, and is today widely adopted again (Harmer and Howe, 2003; Sjölund and Jump, 2015). Therefore, many broadleaf forest landscapes show rapid changes in stand structure and composition due to the co-existence of actively managed coppice stands of different age, old coppices converted to high forests, or even abandoned coppices.

In Europe, thermophilous deciduous forests have been subject to coppicing for centuries and this is still often the case in most southern countries, especially Italy and in the Balkans. Several community types, mostly dominated by oak species, are included in this broad forest category, some of which are listed in the Natura 2000 network due to their ecological value and often fragmented distribution (EEA, 2012). Recent evidence from central Italian communities dominated by *Quercus cerris* and *Q. petraea* showed a remarkable richness in secondary woody species, which increases the compositional and structural diversity of the stands and provide multiple ecosystem functions of social and economic value (Carrari et al., 2016).

At present, thermophilous deciduous forests are partly subject to coppicing, partly converted to high forests through selective thinning and partly abandoned to natural development. However, the effects of these management options on secondary woody species and on understorey diversity and composition are still poorly known, despite their potential consequences on the functional processes, resilience and ecosystems services of the forest community. Hence, we investigated the dynamic changes triggered by coppicing and conversion to high forest on stand structure, plant diversity and composition of thermophilous deciduous forests in central Italy. More specifically we selected a *Quercus cerris* dominated community recognized in Natura 2000 system, and a forest area enclosed in the network of Sites of Communitarian Interest as model system. Our approach was to compare young coppice, adult coppice, coppice in conversion to a high forest thinned within the last 15 years, and coppice in conversion to a high forest not thinned in the last 15 years. Evidence from this study will help to fill a knowledge gap about: 1) dynamic changes along temporal gradient of natural development, and 2) effects of alternative management options on the structure, floristic diversity and composition of one of the most diverse but less investigated forest types in Europe.

5.2 Material and methods

5.2.1 Study area

The study area is located in the province of Siena, Tuscany (central Italy), in the municipalities of Monticiano and Chiusdino (43.10-43.17 N, 11.13-11.21 E) (Figure 5.1; Appendix 5.A), which was selected due to the broad extension of thermophilous deciduous forests.

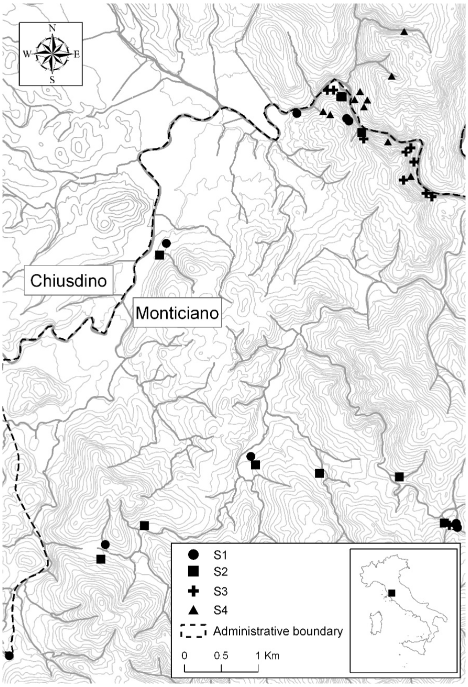


Figure 5.1: Study area location and spatial distribution of field plots. S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned

The area is hilly (200 and 600 m a.s.l), with gentle slopes ($< 60\%$) and some flat areas along the Merse River. The main bedrock is formed by quartzites and anagenites, siliceous rocks of the Verrucano formation. The climate is characterized by Mediterranean-type rainfall with a summer minimum, belonging to the humid (B1) type sensu (Thornthwaite and Mather, 1957)Thornthwaite. Data recorded by San Lorenzo a Merse – Tocchi (228-338 m a.s.l.) meteorological station show an annual average temperature of 14.3^{o}C; the highest temperatures are in July, the lowest in January. The annual average rainfall is 992 mm with November as the wettest and July the driest month. The study area falls within the Site of Community Importance (SCI) Alta Val di Merse (IT5190006) of the Natura 2000 network which hosts important forest habitats: *Castanea sativa* woods (9260), Illyrian oak-hornbeam forests (*Erythronio-Carpinion*) (91L0), *Quercus ilex* and *Quercus rotundifolia* forests (9340), *Salix alba* and *Populus alba* galleries (92A0), Alluvial forests with *Alnus glutinosa* and *Fraxinus excelsior* (*Alno-Padion, Alnion incanae, Salicion albae*) (91E0). This forest area is generally managed as coppice with standards and rarely as compound coppice. In addition, coppice in conversion to high forest or abandoned and left to natural development can be found. In Tuscany, the forestry regulation has established that the minimum rotation age for *Quercus* coppice is 18 years. When the coppice age is older than 36 years, coppicing is subject to authorization from the competent authority. When the coppice age is older than 50 years, coppicing is forbidden and the coppice must be converted to high forest, or left to natural development. In our study area, the minimum number of 60 standards per hectare requested by the regional forest law is sometimes increased to 100-150 by the local forest authorities for soil protection and biodiversity conservation purposes, particularly within protected areas.

5.2.2 Sampling design and data collection

First, the spatial distribution of thermophilous deciduous forests was extracted from a forest type map in vector data format at a scale of 1:10,000 D.R.E.AM. (2012). Polygons encompassing the target forest type were then compared with historical aerial images obtained in 1954, 1978, 1988, 2002, 2005, 2007 and 2010 to extrapolate the year of the last coppicing. Finally, field observations were carried out to classify thermophilous deciduous forests polygons into four forest "strata" based on current forest age and management (coppice with standards vs coppice in conversion to high forest): S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned. Forest strata were defined taking into account a minimal variation of climate, physical and soil conditions.

In each forest stratum, 10 plots were selected with a random approach, for a total of 40 plots (Figure 5.1). In each plot field work was carried out from February to July 2014 within a circular area 314 m^2-wide. The x,y coordinates of the plot center were recorded by a GNSS receiver, with 2-5 m positional accuracy. Stand age was assessed by extracting a core from three suckers belonging to the most frequent diameter class; the cores were extracted at the base of the trunk, and the rings were counted in the laboratory. For strata 3 and 4, the year of the last thinning was assessed by cores extracted from three young suckers present on the most recently cut stumps. The micromorphology (altitude, slope and aspect) was calculated from a Digital Terrain Model (10x10 m) in a grid data format using GIS software. Stand structure and floristic vegetation surveys were carried out in each plot. For forest structure, the diameter at breast height (DBH) of all trees and shrubs (overstorey with a DBH $>$ 2.5 cm and height $>$ 1.3 m) was measured with a caliper, all stems were classified into standards and suckers by visual inspection depending on stem size and their origin (by seed or agamic, respectively); the total tree height was measured by Vertex IV Hypsometer for a sample of trees (a minimun of 15 trees for each plot, excluding young coppices). Floristic surveys were performed, in each plot by recording all vascular plant species in the understorey (< 1.30 m height) and scored for percentage of ground cover and abundance using the Domin transformation of the Braun Blanquet scale (Braun-Blanquet, 1932). Plant identification following Flora d'Italia (Pignatti, 1982) was mostly performed in the field. The main characteristics of inventoried plots are listed in Appendix 5.A.

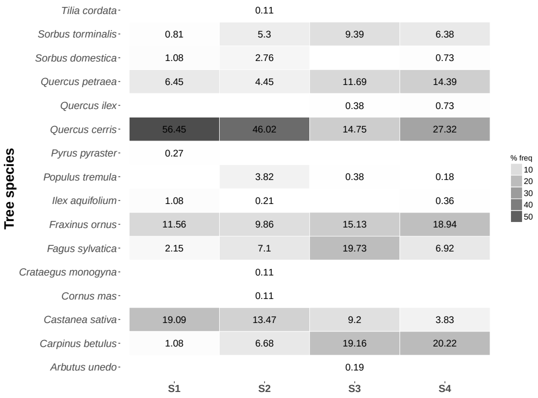
5.2.3 Data analysis

Computation of indexes and statistical analyses were conducted by using R-statistical framework (R-Core-Team, 2017). Tree and understorey components were firstly assessed for structural and diversity parameters separately. Dendrometric, species diversity, and dimensional diversity variables were computed to characterize the structure of the stands. In particular, the following variables commonly used in National Forest Inventories (Corona et al., 2011) were determined: number of stumps per hectare (Nstumps), number of standards per hectare (Nstandards), number of tree individuals per hectare (Nindivid), number of suckers per hectare (Nsuckers), basal area (m^2) per hectare (BA), growing stock volume (m^3) per hectare (V; by using the volume equation of the Italian National Forest Inventory, INFC (2005)), quadratic mean diameter (Dba, diameter of the tree with average basal area), height of the tree with the quadratic mean diameter (Hdba, m) dominant height (m) (Hdom; the mean height of the one hundred largest trees per hectare), and stand age. The total tree height for non-sampled trees was estimated by tree height–diameter relationships (i.e. hypsometric function), which were determined for each tree species using field data. Mean, standard deviation, coefficient of variation, and Gini coefficient of DBH and heights (Gini, 1921) were calculated to estimate the dimensional diversity of the trees in the stands. Shannon diversity index (H'), maximum value of H' (H'max) (Kent and Coker, 1992) and Pielou index or equitability (J) (Pielou, 1969, 1975) were calculated in order to define tree species composition and diversity of each plot. Understorey floristic diversity was quantified at different spatial scales. Gamma diversity (γ) was calculated as the pool species number for each stratum; beta-diversity (β) was estimated as the mean Lennon distance (Lennon et al., 2001) of each plot against the other plots within each stratum. Finally, alfa-diversity (α) was evaluated as the total understorey species richness (SR), Shannon diversity (H') (Kent and Coker, 1992), and Evenness (J) indexes (Pielou, 1969, 1975) for all plots and then summarized for each strata. Next, we determined the effect of the management comparing structure, dimensional and species diversity among strata by means the analysis of variance (ANOVA) and Tukey non parametric test (Siegel and Tukey, 1960). The influence of temporal stages on floristic data in terms of diversity and composition was assessed with univariate and multivariate methods. SR mean values of the four strata were compared with the Kruskal-Wallis test while H', J and Lennon distance by analysis of variance (ANOVA). Non-Metric Multidimensional Scaling (NMDS) (Oksanen, 2015) was used to visualize the compositional differences between the four strata based on the cover-weighted Bray-Curtis distance and differences in the position of plots in the multivariate space were tested using a permutational analysis of variance (PERMANOVA) with 999 permutations (Oksanen, 2015). The Indicator Species Analysis (Dufrene and Legendre, 1997) was carried out for detecting species significantly associated with each stratum considering frequency and/or abundance. In order to highlight the ecological differences in terms of light (L), temperature (T), continentality (C), soil humidity (U), soil reaction (R) and nutrients (N) at different temporal stages, the corresponding Ellenberg indicator values (Pignatti, 2005) were computed for each plot and compared with the Kruskal-Wallis test between strata. Finally, regression analysis and F statistics were used to examine the relationship between alpha diversity indexes (e.g. SR and H') and dendrometric variables (Age, Nstumps, Nstandards, Nindivid, Nsuckers, BA, V, Dba, Hdba, Hdom).

5.3 Results

5.3.1 Stand structure and overstorey species diversity

Q. cerris, *Q. petraea*, *C. sativa*, *C. betulus* *F. sylvatica* and *F. ornus* were the most abundant species in the tree layer (Fig. 5.2). Secondary woody species such as *S. torminalis*, *S. domestica* and *I. aquifolium* were less abundant (Fig. 5.2), with a frequency <10% in each stratum.



Tree species	S1	S2	S3	S4
Tilia cordata		0.11		
Sorbus torminalis	0.81	5.3	9.39	6.38
Sorbus domestica	1.08	2.76		0.73
Quercus petraea	6.45	4.45	11.69	14.39
Quercus ilex			0.38	0.73
Quercus cerris	56.45	46.02	14.75	27.32
Pyrus pyraster	0.27			
Populus tremula		3.82	0.38	0.18
Ilex aquifolium	1.08	0.21		0.36
Fraxinus ornus	11.56	9.86	15.13	18.94
Fagus sylvatica	2.15	7.1	19.73	6.92
Crataegus monogyna		0.11		
Cornus mas		0.11		
Castanea sativa	19.09	13.47	9.2	3.83
Carpinus betulus	1.08	6.68	19.16	20.22
Arbutus unedo			0.19	

Figure 5.2: Tree species percentage abundance in the tree layer over the four strata. S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) that was thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) that was thinned more than 15 years ago or never thinned

Significant differences between the four forest strata were found in terms of structure, dendrometric variables and tree species diversity, as reported in Tab. 5.1. The number of individual per hectare and the number of stumps per hectare in S2, was greater ($P<0.05$) than in S1, while the number of suckers per hectare in S1 was greater than in S3 and S4 ($P<0.05$). As expected, volume of all layers was different and significantly increased from S1 to S4 ($P<0.001$) and basal area of S1 was significantly lower ($P<0.001$). The height of the tree with quadratic mean diameter and dominant height in S4 were greater than in S1 ($P<0.01$). Standard deviation of DBH, arithmetic mean of height, standard deviation of height, and coefficient of variation of height were significantly different between strata. S2 had a smaller standard deviation of diameter than S4. Regarding the mean height, S3 had the highest value and was significantly different from S1 ($P<0.05$). Standard deviation of height was significantly higher in S4 than S2 ($P<0.001$) and coefficient of variation of height in S2 was lower than in S1 and S4 ($P<0.05$). Considering the species diversity of the overstorey, S3 had a Shannon index significantly higher than S1 ($P<0.05$).

5.3.2 Understory diversity and composition

Regarding gamma diversity, the total number of species showed a declining trend from S1 to S4, (Fig. 5.3). The mean plot-level species richness (SR) was also declining in the four strata, though significantly higher only in S1 ($P<0.01$; Fig. 5.3). Shannon index (H') ranged from 0.88 to 3.19, and equitability (J) between 0.40 and 0.87. Mean H' was different between the strata ($P<0.01$) and S1 showing the highest value, while equitability (J) was not different (Fig. 5.3).

The β-diversity analysis showed that differences in species composition in S1 and S2 were greater than in S3 and S4 ($P<0.001$; Fig. 5.4). NMDS ordination produced a scattergram of plots stretching from S1 to S4, and PERMANOVA showed that compositional differences between the strata were significant ($P<0.01$; Fig. 5.5).

Indicator species analysis results showed both ruderal and nemoral species were significantly associated with young coppice, while, unexpectedly, no nemoral species were found to be significantly associated with coppice in conversion to high forest (Tab.

Variable	**S1**	**S2**	**S3**	**S4**	**ANOVA**
	Mean ±SD	*Mean ± SD*	*Mean ± SD*	*Mean ± SD*	*P-value*
Age (years)	4.5 ± 2.5	23.9 ± 5.8	49.8 ± 6.8	65.5 ± 7.2	-
Nstumps ha^{-1}	433 ± 495^{a}	1306 ± 754^{b}	872 ± 561^{ab}	958 ± 350^{ab}	0.013*
Nstandards ha^{-1}	166 ± 70	178 ± 72	150 ± 83	197 ± 68	n.s.
Nindivid ha^{-1}	1181 ± 1143^{a}	2688 ± 1490^{b}	1528 ± 1306^{ab}	1611 ± 592^{ab}	0.041*
Nsuckers ha^{-1}	3.0 ± 2.4^{a}	1.9 ± 0.8^{ab}	1.4 ± 0.6^{b}	1.5 ± 0.3^{b}	0.046*
BA (m^{2} ha^{-1})	7.8 ± 3.3^{a}	21.2 ± 8.8^{b}	24.1 ± 7.4^{b}	26.9 ± 6.4^{b}	0.000***
V (m^{3} ha^{-1})	54.5 ± 21.4^{a}	156.4 ± 72.1^{b}	195.8 ± 49.3^{c}	252.9 ± 70.0^{d}	0.000***
Dba (cm)	12.2 ± 6.2	12.0 ± 5.5	16.0 ± 3.4	15.2 ± 2.5	n.s.
Hdba (m)	11.3 ± 3.4^{a}	12.5 ± 3.1^{ab}	14.2 ± 2.4^{ab}	15.6 ± 1.5^{b}	0.007**
Hdom (m)	17.6 ± 3.2^{ab}	17.6 ± 3.4^{b}	19.9 ± 2.8^{abc}	22.5 ± 1.9^{c}	0.001**
Gini DBH	0.7 ± 0.1	0.7 ± 0.1	0.6 ± 0.1	0.7 ± 0.1	n.s.
DBH mean (cm)	9.4 ± 5.8	9.0 ± 3.6	13.0 ± 3.3	11.4 ± 3.0	n.s.
DBH SD (cm)	7.9 ± 3.3^{ab}	6.7 ± 2.0^{a}	9.3 ± 1.8^{ab}	10.0 ± 1.2^{b}	0.017*
DBH CV	0.9 ± 0.2	0.8 ± 0.1	0.7 ± 0.2	0.9 ± 0.2	n.s.
Gini H	0.7 ± 0.1	0.7 ± 0.1	0.6 ± 0.1	0.7 ± 0.1	n.s.
H mean (m)	8.1 ± 3.6^{a}	10.0 ± 2.2^{ab}	11.5 ± 1.9^{b}	10.8 ± 2.0^{ab}	0.033*
H SD (m)	4.4 ± 1.6^{ab}	3.1 ± 1.2^{a}	4.7 ± 1.3^{ab}	5.9 ± 0.9^{b}	0.000***
H CV	0.6 ± 0.3^{a}	0.3 ± 0.1^{b}	0.4 ± 0.1^{ab}	0.6 ± 0.1^{a}	0.002*
H’	0.8 ± 0.5^{a}	0.9 ± 0.3^{ab}	1.4 ± 0.3^{b}	1.2 ± 0.5^{ab}	0.012*
H’ max	1.2 ± 0.6	1.4 ± 0.4	1.7 ± 0.2	1.6 ± 0.6	n.s.
J	58.9 ± 34.7	65.8 ± 14.7	80.5 ± 7.5	67.4 ± 27.4	n.s.

Table 5.1: Summary of variables of each strata and ANOVA and Tukey test results for structural and species diversity index. S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (5-75 years) thinned more than 15 years ago or never thinned.
Nstumps: number of stumps per hectare; Nstandards: number of standards per hectare; Nindivid: number of tree individuals per hectare; Nsuckers: number of suckers per hectare; BA: basal area; V: volume; Dba: quadratic mean diameter; Hdba: height of quadratic mean diameter; Hdom: dominant height; Gini DBH: Gini coefficient of diameter at breast height; DBH mean: arithmetic average of DBH; DBH SD: standard deviation of DBH; DBH CV: coefficient of variation of DBH; Gini H: Gini coefficient of height; H mean: arithmetic average of height; H SD: standard deviation of height; H CV: coefficient of variation of height; H’: Shannon index; H’ max: maximum value of H’; J: Pielou index or equitability.
*** $P<0.001$; ** $P<0.01$; * $P<0.05$; n.s., not significant.

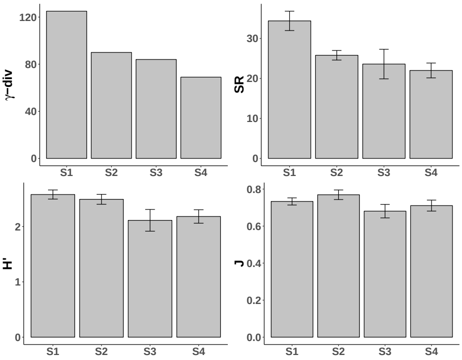


Figure 5.3: Gamma diversity (γ-div), species richness (SR, $P<0.01$), Shannon index (H', $P<0.01$) and equitability or Pielou index (J, P = not significant) of the four strata. S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned

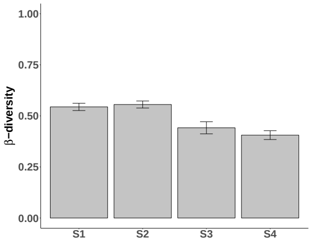


Figure 5.4: β-diversity based on Lennon distance mesure (ANOVA, $p<0.001$). S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned

5.2). However nemoral species abundance increase in coppice in conversion to high forest (e.g. *Anemone nemorosa, Festuca heterophylla, Ruscus aculeatus*, Fig.5.6) and some of them are present only in the older strata (e.g. *Daphne laureola, Veronica hederifolia, Poa nemoralis*, Fig.5.6). Ellemberg indicator values revealed that species in S1 had higher light requirements compared to the other

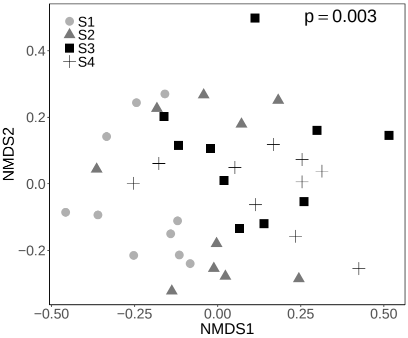


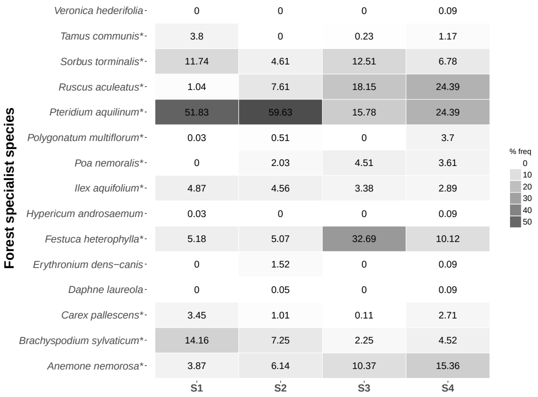
Figure 5.5: Non-Metric Multidimensional Scaling ordination diagram based on cover-weighted Bray Curtis showing compositional dissimilarities distance, between S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned. PERMANOVA with 999 permutations was used to assess the significance of results ($p<0.01$).

strata ($p<0.01$; Fig. 5.7) while less thermophilous species occurred in S2 differently from the others strata ($p<0.05$; Fig. 5.7). All strata showed low continentality values with significant differences only between S1 and S3 ($p<0.01$; Fig. 5.7); soil reaction in S1 was significantly higher than in the other strata ($p<0.001$; Fig. 5.7).

Linear regression showed a significant negative relationship between SR and BA ($P<0.05$), between SR and stand age ($P<0.01$) and between H' and age, while no significant relation was found between H' and BA (Fig. 5.8).

Stratum	**Species**	**P-value**	**Functional type**	**Ecology**
S1	*Centaurium erythraea*	0.039^{*}	Herb	Forest edge
S1	*Campanula rapunculus*	0.046^{*}	Herb	Forest edge
S1	*Viola reichenbachiana*	0.001^{***}	Herb	Nemoral
S1	*Rubus hirtus*	0.001^{***}	Shrub	Nemoral
S1	*Brachypodium sylvaticum*	0.034^{*}	Graminoid	Nemoral
S1	*Sorbus torminalis*	0.030^{*}	Tree	Nemoral
S1	*Tamus communis*	0.008^{**}	Herb	Nemoral
S1	*Sorbus domestica*	0.034^{*}	Tree	Nemoral
S1	*Veronica officinalis*	0.019^{*}	Herb	Nemoral
S1	*Rubus ulmifolius*	0.001^{***}	Shrub	Pioneer
S1	*Asphodelus albus*	0.025^{*}	Herb	Pioneer
S1	*Carex flacca*	0.037^{*}	Graminoid	Pioneer
S1	*Cirsium arvense*	0.045^{*}	Herb	Pioneer
S1	*Rosa canina*	0.046^{*}	Shrub	Pioneer
S1	*Conyza canadensis*	0.040^{*}	Herb	Pioneer
S1	*Cytisus scoparius*	0.040^{*}	Shrub	Pioneer
S3	*Quecus ilex*	0.01^{**}	Tree	
S3	*Fagus sylvatica*	0.03^{*}	Tree	
S3	*Quercus cerris*	0.049^{*}	Tree	
S3	*Melica uniflora*	0.016^{*}	Herb	Nemoral
S4	*Anthoxanthum odoratum*	0.01^{**}	Herb	Pioneer

Table 5.2: Indicator species analysis results and its association to the forest strata. S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned.
*** $P<0.001$; ** $P<0.01$; * $P<0.05$.



Forest specialist species	S1	S2	S3	S4
Veronica hederifolia	0	0	0	0.09
*Tamus communis**	3.8	0	0.23	1.17
*Sorbus torminalis**	11.74	4.61	12.51	6.78
*Ruscus aculeatus**	1.04	7.61	18.15	24.39
*Pteridium aquilinum**	51.83	59.63	15.78	24.39
*Polygonatum multiflorum**	0.03	0.51	0	3.7
*Poa nemoralis**	0	2.03	4.51	3.61
*Ilex aquifolium**	4.87	4.56	3.38	2.89
Hypericum androsaemum	0.03	0	0	0.09
*Festuca heterophylla**	5.18	5.07	32.69	10.12
Erythronium dens−canis	0	1.52	0	0.09
Daphne laureola	0	0.05	0	0.09
*Carex pallescens**	3.45	1.01	0.11	2.71
*Brachyspodium sylvaticum**	14.16	7.25	2.25	4.52
*Anemone nemorosa**	3.87	6.14	10.37	15.36

Figure 5.6: Percentage abundance of nemoral and ancient forest species in the herbaceous layer (≤ 50 cm) over the four strata. S1: young coppice (≤ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned. (*) indicates ancient forest species (Hermy et al., 1999).

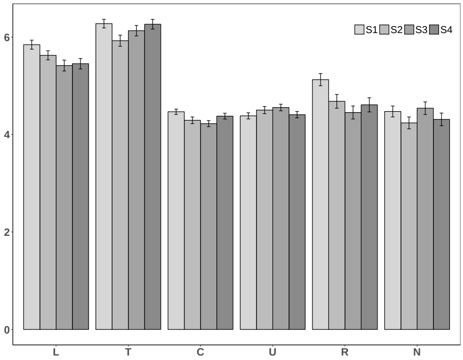


Figure 5.7: Mean and standard error of Ellemberg value for ligh (L), temperature (T), continentality (C), soil humidity (U), reaction of soil (R) and nitrogen (N). Differences are significant for L ($p<0.01$), T ($p<0.05$), C ($p<0.01$) and R ($p<0.001$). S1: young coppice (≤ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned.

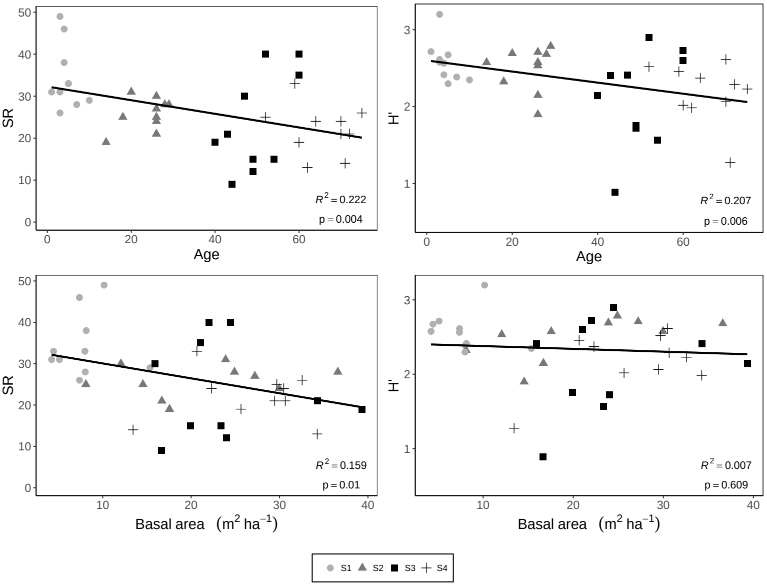


Figure 5.8: Variation of the Species richness (SR) in relation to age (P<0.01) and basal area (BA) (P<0.05) and variation of H' in relation to age (P<0.01) and basal area (not significant) measured in S1: young coppice (≤ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) thinned more than 15 years ago or never thinned.

5.4 Discussion

Our results showed temporal changes in both structural and floristic composition in thermophilous deciduous forests due to forest management associated to the coppice system and the conversion of coppice to high forest. After coppicing, stand density decreased both in terms of number of individuals and number of stumps. However, the lower number of stumps in young coppices can be explained by a greater difficulty in detecting cut stumps on the ground due to the presence of a dense herb and/or shrub layer. Regarding structural diversity both DBH, BA and height diversity increased with stand age (Tab. 5.1). *Q. cerris* was the most frequent species in the tree layer, but other tree species enters the community with increasing stand age such as *C. betulus*, *F. ornus*, *Q. petraea*, *F. sylvatica*, and *S. torminalis*. The presence of *F. sylvatica* in our lowland submediterranean area is of special ecological and conservation relevance, as this species occurs with extrazonal, relict populations of Holo-Pleistocene origin that persist in very local microclimatic refugia (Sabbatini et al., 2011). *Ilex aquifolium*, an ancient forest species found in the shrub layer, decreased its abundance over time with increasing stand age. The increase of diversity in tree size and vertical stand structure can determined different light conditions that influence as a "cascade effect" the understorey plant dynamics (Scolastri et al., 2017; Kirby et al., 2017). On the other hand, young coppices were found to have greater floristic diversity in terms of species richness as reported in other studies (Decocq et al., 2004; Kirby, 1990; Hédl et al., 2017). Coppicing causes a sudden and strong increase of light availability to the ground consequent to the reduction of forest cover, which favours the establishment of light-demanding species in the lower layers (Decocq et al., 2004; Bartha et al., 2008): ruderal annuals such as *Bromus* sp. and effective colonizers such as *Rubus* spp. After coppicing, non-forest species developed quickly and their number represented 80% of the species richness. The increase of species richness associated to coppicing has been found by other authors (Hédl et al., 2017). However, our results show that species richness decreased when coppice age increased, as reported by Bartha et al. (2008) for beech coppice stands. In addition, our results show that along a temporal gradient forest specialist species become the most common species, albeit with low density (e.g. *Anemone nemorosa, Festuca heterophylla*, Fig. 5.6), in accordance with other Authors (Decocq et al., 2004; Bartha et al., 2008). As described above, while a significant relationship between species richness, age and BA was already found in other studies (Selvi and Valleri, 2012), the absence of a significant relationship between Shannon index and BA was an interesting result in our investigation. Species richness and Shannon index are two alpha diversity indexes that showed a different trend. Species richness decreased with increasing stand age and BA, while Shannon index showed no significant decrease. In fact, species which are able to expand and prevail in the young coppice decreased with increasing stand cover, leading to an increase of evenness which counterbalances the decrease in species richness in the older strata (Fig. 5.8). Ellenberg values confirmed that the most influencing ecological factors were light and soil pH: species of the young coppice indicate higher levels of light and lower level of soil acidity than coppice in conversion to high forest. Soil reaction also plays an important role on the understory vegetation and higher soil acidity of older stands can explain the lower species richness in coppice in conversion to high forest.

Based on our results, mature stands are characterized by greater structural diversity and lower values of species richness. However, their ecological value was confirmed by the presence of specialist and ancient forest species found in S4 such as *Anemone nemorosa, Brachypodium sylvaticum, Carex pallescens*. Ancient forest species have a slower pace of territorial expansion and colonize new habitats slowly (Hermy et al., 1999). For this reason, the frequency of disturbances due to coppicing can have long-term negative effects on the biodiversity and functionality of stands, as confirmed by other studies (Decocq et al., 2004). In particular, the frequency of utilizations can lead to the decline of slow colonizers. In our study this critical aspect is pointed out by indicator species analysis that did not highlight significant association between ancient forest species and coppice in conversion to high forest due to the very low overall abundance of such species. Complexity of such coppices in conversion to high forest is highlighted by the presence of seedlings of numerous tree species in the understorey that were not found in the tree layer (*A. opalus, A. campestre, A. pseudoplatanus, A. monspessulanum, F. angustifolia, U. glabra*), which may have an important role in community dynamics in the longer term.

Coppicing can have both positive and negative effects in terms of forest dynamics complexity and biodiversity. Coppice cessation at the same time over large areas could lead to landscape-scale uniformity in the woodland structure (Kirby et al., 2017) and this is also true for uniform thinning over large areas. Therefore, the maintenance of the coppice system depends on the conservation purpose. Based on our results, the major risk is for forest specialist species and secondary woody species. Coppicing repeated frequently over time can lead to a decrease in the abundance of such species, which is not the case for older stands which have not been thinned for over 50-70 years. Large-scale forest management planning assisted by decision support systems (e.g. Bottalico et al. 2016) is an essential requirement for a good conservation level, taking into account, at the landscape level, the size of forest areas to cut, their spatial connection and the time frame of forestry operations to respect both landscape level biodiversity goals and economic needs of local communities in rural areas, which still use coppice systems as a traditional form of forest management for firewood production (Zeneli and Kola, 2017) also in protected areas such as the Natura 2000 sites (Mairota et al., 2016).

5.5 Conclusions

The aim of this study was to assess stand structure, plant dynamics and changes in four temporal stages and to understand if the coppice management system ensures a satisfactory conservation status for thermophilous deciduous forests. The comparison of this forest type in four different stages has led to an assessment of the effects of coppicing in the short and medium term and also provides a way to support the optimal management-scale in order to ensure a good habitat conservation status. Coppice system has a greater floristic diversity. However, coppice in conversion to high forest has a higher structural diversity and the presence of specialist and ancient forest species, and secondary woody species. The frequency of silvicultural interventions seems to be a factor that influences the complexity of the forest habitat. In particular, the spread of forest specialist species, many of which are ancient forest species, appears to be limited by frequent felling. From a conservation point of view, we suggest maintaining the presence of all types of management in the study area. In particular we propose to continue to manage small areas as coppice and at the same time, encourage the conversion to high forest to favour the presence of forest specialist species. Therefore high turnover in species composition and structural condition variation within the study area is suggested to increase local beta diversity. At the landscape scale our study highlights the importance of leaving also some areas to natural evolution, so as to create islands with old-growth characteristics which can serve as reservoirs of nemoral and forest specialist species. Many areas of thermophilous deciduous forests are included in the Natura 2000 network and the Habitat Directive does not provide practical management indications for the different habitats and/or species. This study provides information on thermophilous deciduous forests and their resilience in order to promote conservation-oriented management of this habitat type.

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5.A Plot description

Table 5.3: Characteristics of the plot investigated for the four strata: S1: young coppice ($\leq$ 10 years); S2: adult coppice (10-36 years); S3: coppice in conversion to a high forest (36-60 years) that was thinned less than 15 years ago; S4: coppice in conversion to a high forest (50-75 years) that was thinned more than 15 years ago or never thinned

Stratum	N	E	Age	Years from last thinning	Altitude(m a.s.l.)	Aspect	Slope(%)
S1	43.11257707	11.21144259	5	-	421	E-NE	17
S1	43.12173991	11.17777341	5	-	325	Flat	3
S1	43.11141024	11.15305027	1	-	397	Flat	9
S1	43.09806873	11.136557	3	-	408	O-SO	23
S1	43.14812918	11.16456077	7	-	285	NE	8
S1	43.16365332	11.18694183	10	-	285	N	39
S1	43.1625631	11.19561227	4	-	301	N	60
S1	43.16286895	11.19524247	4	-	298	NE	55
S1	43.11289514	11.21174767	3	-	412	E	15
S1	43.11236337	11.21193558	3	-	412	E	19
S2	43.11943718	11.18911228	20	-	340	NE	14
S2	43.12072821	11.17849471	29	-	350	NO	19
S2	43.11359651	11.15967554	26	-	381	S	0
S2	43.10961721	11.15227054	26	-	410	NE	25
S2	43.14669359	11.16338722	18	-	289	O	16
S2	43.16556208	11.19433776	26	-	290	NE	21
S2	43.16550045	11.19460165	26	-	290	NE	13
S2	43.16109366	11.19768029	28	-	213	NE	35
S2	43.11877888	11.20247994	14	-	368	S-SO	30
S2	43.11297656	11.20978268	26	-	420	NE	24
S3	43.15343563	11.20819614	40	Never thinned	267	NE	60
S3	43.15300815	11.20913114	52	12	286	N-NE	55
S3	43.15733452	11.2062895	49	14	298	E	55
S3	43.15514164	11.20447605	49	7	370	E	38
S3	43.16640194	11.19317768	60	5	273	N	30
S3	43.16639719	11.19213531	60	5	280	N	19
S3	43.16030995	11.19802052	44	14	333	N-NE	46
S3	43.15858447	11.2049944	54	12	340	N	50
S3	43.15906071	11.2058453	47	12	322	N	39
S3	43.11267237	11.2106056	43	Never thinned	415	O	47
S4	43.15557432	11.20570379	59	20	345	E-NE	55
S4	43.15988864	11.20203266	52	15	282	N-NE	35
S4	43.16339724	11.19249599	71	21	351	O-NO	37
S4	43.16382539	11.19132285	64	30	306	O	23
S4	43.17336941	11.20522306	70	28	405	N	55
S4	43.16792608	11.20293662	62	Never thinned	423	N-NE	23
S4	43.16613584	11.19759586	75	Never thinned	309	O	20
S4	43.16517077	11.19663051	70	Never thinned	280	O	50
S4	43.16507855	11.19876293	72	Never thinned	331	O	26
S4	43.16424076	11.19816801	60	Never thinned	308	O	41

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