Monitoring and Indicators of Forest Biodiversity in Europe – From Ideas to Operationality

Marco Marchetti (ed.)

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Preface – Ideas on Emerging User Needs to Assess Forest Biodiversity

U. Pinborg

EEA

Forests are coming into focus. They cover more than 1/3 of our land area and are in constant and visible change. They also contain a very large proportion of our wild and cultivated or introduced species and some of the habitats with the longest continuity. Forests are central both to protection of biodiversity as such and to securing important goods and services.

The importance of forest functions and the goods that they provide has changed radically. Market prices for timber have fallen deeply, while taxation systems make ownership transfer difficult, enforcing forest owners to change strategies of management, resource use and of time planning. The understanding of the role of forests for society and for the environment is increasing rapidly with concepts, some of which are new to both politicians and to the public. The forests as CO_2 sinks gained the highest global political significance in relation to climate change and implementation of the Kyoto Protocol. Consumers are interested in eco-certified wooden products. Drinking water-providers strive to set aside forests or afforested areas for water-protection. Hunting for recreation has grown into an important economy that has to co-exist with growing wishes for access to close-by everyday forests, sport-event forests and eco-tourists wanting access to untouched forests.

Next to the Kyoto protocol target, which is already of some years standing, stands the generic biodiversity target of halting the loss of biodiversity by 2010. This concerns the main ecosystems and includes in forests all levels from ecosystem and species to the genetic level through reduction of the pressure on forests as well as through mitigating actions.

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Introduction

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The biological diversity of European forests is the result of the evolution of the communities under ecological forces such as climate, fire, competition etc. as well as of a long history of human influence.

Pressures and stresses on forest biodiversity are sensitively increasing, so natural disturbances and human activities (forestry, afforestation of agricultural lands, clearance of forest areas for other land use, industrialization etc.) shape the biological diversity of European forests. To preserve them it is, for example, necessary to find the optimal mixture of conservation designated areas and production forests both managed with consideration to biodiversity. The role of biodiversity in the functioning of the forest ecosystems, in relation to management, must be analysed considering processes that cross multiple time and space scales. The forest must be perceived as a complex and adaptive biological system.

The process of management to pursue these aims must be based upon scientifically sound knowledge of the biological diversity and ecosystem functions. To be successful, this strategy should be developed and locally adapted in close interaction with policy makers, managers, environmentalists, forest owners and other stakeholders. On a pan-European scale there is a need to further develop biodiversity assessment tools such as indicators of biodiversity and preferred methodologies to collect data.

This volume will focus on species, stand and landscape level, with the aim to know how to successfully implement biodiversity indicators, e.g. with respect to adaptation to forest types, methods for inventory and mapping, etc.

A proper identification of the processes underlying the depletion of forest biodiversity requires an integrated approach in understanding the complex interdependencies between pressures related to the socio-economic development (agriculture, industrialization, demography) and their impacts on forest ecosystems.

Papers are collected on the indicators and monitoring of threats on European forest ecosystems in Europe, such as: forest fires, introduced and invasive species, threats (in and around) forest ecosystems, stand-level indicators and relationships with forest management, assessing and reporting on the state and changes of Sustainable Forest Management and development of indicator at landscape level. Several ongoing international initiatives covering the development and monitoring of forest biodiversity-related indicators are

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presented and discussed: the MCPFE process, the EEA core set of Biodiversity indicators, the CBD Core set of Biodiversity indicators.

Based on the Forest Principles, in 1993 European countries started the MCPFE process adopting the Helsinki, Lisbon and Vienna declarations with several important resolutions, such as the ‘General Guidelines for Sustainable Management of Forests in Europe’, where a definition for SFM was provided. The concept of MCPFE has created an indispensable network for the sustainable forest management implementation. Therefore, some initiatives to define general Principles, the SFM Criteria and a coherent set of performance Indicators for forest management were undertaken.

Sets of SFM indicators, in accordance with MCPFE, are available for many countries showing shortages of standardized, reliable and up-to-date information and variables on many forests aspects.

Other SFM issues are being analysed in Europe respect to the ‘Close-to-Nature forest management’, and, in the same, a new concept linked to the sustainable development originated by UNCED, in the Convention on Biological Diversity (CBD) is arising: the Ecosystem Approach (EA). The EA is a strategy for the integrated management of land, water and living resources that promotes conservation and sustainable use. The concept of EA includes 5 Guidelines and 12 Principles (Decision V/6 di COP, Nairobi, 2000). They show that it is inopportune to consider ecological aspects in a sectoral way owing to the complex and dynamic nature of ecosystems and the absence of complete knowledge of their functioning. At present, limited experience exists with regard to EA field level application, in forest ecosystems too. Recently, a cooperation between the MCPFE and PEBLDS started to elaborate clarifications between the EA and SFM concepts.

Management, conservation and sustainable use of renewable natural resources are thus the stated goal of both concepts. They are guided by a set of principles that, although similar, differ slightly in scope. A coherent linkage between SFM and EA can be achieved only if SFM changes its basic reference paradigm and some innovative proposal are presented in this book. There is also a need for Forest Biodiversity Conservation to strengthen cross-sectoral integration, which can be undertaken at least in part throughout application of SFM tools into other sectors; EA should, in turn, consider application of SFM tools and approaches, such as criteria and indicators, to move towards an outcome-oriented approach. Since EA is a more recent concept, detailed practical guidelines are still missing. However, no examples of a comparative analysis of the outcome of applying the two concepts to a given forest have been identified.

In this volume, the importance of linking different spatial and time scales and developing an integrated set of indicators across scale have been emphasised. In this context, an important indicator is species richness, that is a statistically significant indicator of other components of biodiversity at least at large spatial scale. Some evidence was given that species richness in one species guild could be used in predictions of species richness in other guilds. Dead wood and other structural aspects of forest stands are potentially important biodiversity indicators; they are straightforward indicators for operationality, but only partially relevant and more research is needed on natural reference points and the development of these parameters during stand development.

It is important to consider that, for an European Forests wide biodiversity assessment and monitoring system, the use of remote sensing data combined with terrestrial sampling seems to be a feasible low cost approach, despite of technological limits, costs and standardized procedures. In this way landscape structure could be monitored frequently as far as a rapid indicator for possible changes in biodiversity is needed and as input to the large-scale aspects of biodiversity changes. In times of tight budgets existing monitoring schemes (e.g. NFIs, ICP Forests) should be extensively utilised and crucial will be the harmonization and coordination efforts (e.g. ForestBIOTA, ENFIN initiatives).

It is also important to consider public wishes and political goals: they have changed in the past and will change again; therefore monitoring systems should be sufficiently robust to cope with current and future changes in policy emphasis. On the whole, perspectives and operational works tend to be fragmented.

Since the need for an overall policy relevant synthesis and guidance for decision-makers and forest managers are increasing, we hope that this book and the event that originated it in the IUFRO Conference in Florence in 2003, will contribute for a better understanding on the importance of interdisciplinary approach in forest biodiversity assessment.

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Session 1: Emerging User Needs and Pressures on Forest Biodiversity

Biodiversity Trends and Threats in Europe – Can We Apply a Generic Biodiversity Indicator to Forests?

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Abstract

At the 9th meeting of the SBSTTA of the CBD a recommendation was made to begin testing of five biodiversity indicators in order to measure progress towards the 2010 target. This paper considers one of these indicators for Pan-Europe. The basic principle of the index is to calculate the average trend in abundance of a set of ecosystem-representative species. The index is considered to be generic, i.e. applicable to all ecosystem types including forests. It can produce both headline messages for high-level policy making and communication, and detailed information for in-depth analysis. This paper describes the conceptual framework of the index and explores the data availability for the index in Pan-Europe.

Keywords: biodiversity, indicators, monitoring, forest, species.

1. Introduction

The global context

The loss of biological diversity worldwide is of serious concern to international policy makers and national authorities alike. This concern led to the adoption by the World Summit on Sustainable Development (Johannesburg 2002) of a global target to reduce the rate of biodiversity loss by 2010. The setting of such an ambitious and quantitative policy target has highlighted the need for monitoring biodiversity in order to evaluate global progress towards it. Indicators are needed, therefore, that are suitable for aggregation across nations and ecosystem types.

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At the 9th meeting of the SBSTTA of the Convention on Biological Diversity (Montreal 2003), important progress was made regarding biodiversity indicators and their potential for monitoring progress towards the 2010 target. UNEP (2003a) states that biodiversity loss can consist of:

1. A decline in extent, condition or sustainable productivity of ecosystems;
2. A decline in abundance, distribution or sustainable use of species populations and extinctions; and
3. Genetic erosion.

SBSTTA recommended that indicators for the following should be tested immediately (UNEP 2003b; 2004):

1. trends in extent of selected biomes, ecosystems and habitats;
2. trends in abundance and distribution of selected species;
3. change in status of threatened species;
4. trends in genetic diversity of domesticated animals, cultivated plants, and fish species of major socio-economic importance; and
5. coverage of protected areas.

Also, UNEP (2003a) proposed a number of requirements the indicators should meet to be useful for their target audience. In brief these criteria state that the individual indicator should: (1) adress the key properties of biodiversity loss; (2) be meaningful for the target audience; (3) be affordable and accurately measurable and make use of existing information; (4) be sensitive to change; (5) be scientifically sound; (6) have broad acceptance; (7) represent the various biodiversity levels and (8) be flexible for use in a limited number of composite indicators.

Europe

The 2010 target has also become the overarching biodiversity goal at the regional scale of Europe. Similar or even more ambitious biodiversity goals are set by the EU Sustainable Development Strategy (2001a) and various other European Union policies (EC 1998, 2001b, 2001c) The EC Birds and Habitats Directives (EC 1979, 1992) are the main instruments in the EU for achieving these goals of biodiversity conservation. On the Pan-European level the Ministerial 'Environment for Europe' process adopted a resolution on halting the loss of biodiversity by 2010 (UN/ECE 2003). In Europe, as in the CBD, work is being done to design indicators to monitor the state and trends in Europe's biodiversity, and to measure the impacts of various policies. An overview of existing indicators in Europe has been produced by ECNC (2002). There are also many initiatives to develop biodiversity indicators at the national scale (UNEP 2003c; Fischer 2002).

Forest ecosystems contain a large proportion of Europe's biological diversity. The Ministerial Conference on the Protection of Forests in Europe (MCPFE), in its Vienna declaration, built on the international commitments of the CBD and the WSSD, when committing to conserve and restore the biological diversity of forests across Europe (MCPFE 2003a). The MCPFE has included nine forest biodiversity-related indicators within its set of indicators of Sustainable Forest Management (MCPFE 2002, 2003b). The species indicators focus primarily on tree species and on species of conservation concern. The BEAR project identified a larger set of some tens of potential indicators for assessment of biodiversity in forest ecosystems (Larsson 2001). Although both the MCPFE and the BEAR indicators can contribute to monitoring progress towards biodiversity targets for forest ecosystems at the European scale, still they do not fully meet the CBD requirements for biodiversity indicators.

The indicators are often not very sensitive to change over time, they focus at species of conservation concern rather than being representative of a broader range of species, sets of indicators are long, and aggregation between indicators and across scales is not really addressed. In both the MCPFE and the BEAR sets conceptual and operational links to (sets of) indicators for other ecosystems are lacking.

The challenge

The challenge for the biodiversity research community now is to use the momentum of the 2010 target to make considerable progress in the development and implementation of biodiversity indicators. These indicators should first and foremost match the requirements of their users: policy makers on global, regional and national level. Specifically for (European) forests, we identify as the challenge:

> *to develop a forest biodiversity indicator that is suitable for comparison and and/or aggregation with measures of biodiversity for other ecosystem types in order to track generalised progress towards biodiversity targets.*

Towards a generic biodiversity index for Europe

In this paper we propose a conceptual framework for a policy relevant biodiversity index for the Pan-European scale. The basic principle of this indicator is to calculate the average trend in abundance of a set of ecosystem-representative species. The indicator is considered applicable to all ecosystems, including forests. It will give a comprehensive yet concise overview of impacts of human activities on status and trends in European biodiversity. The index corresponds to the second of the five SBSTTA indicators mentioned above and will meet the CBB requirements for biodiversity indicators.

As a starting point for developing the present index, the authors have used previous experiences with similar species-based indicators like the Living Planet Index (Loh 2002) (Figure 1) and the Natural Capital Index (UNEP 1997a; 1997b). Similar species trend indices, but only for birds, have been developed as the UK Headline indicator of wild bird populations (Gregory 2003a) and the Pan-European Common Bird Index for farmland and forest birds (Gregory 2003b). Houlahan et al. (2000) developed a global population size trend index on 157 species of amphibians.

2. The conceptual framework

Categorisation

The study area is delineated as pan-Europe, till the Ural mountains, including the Anatolian part of Turkey, and also including the offshore islands and archipelagos. To categorise the study area 11 biogeographical regions (Roekaerts 2002) are combined with the 10 top-level habitat types from the EUNIS habitat classification (Davies and Moss 2002; see also http://eunis.eea.eu.int/index.jsp) (Table 1). Forest is one of those major habitat types (Category G: woodland and forests and other wooded land). The EUNIS habitat types are mutually exclusive; there is no overlap. By combining the biogeographical regions and the major habitat types we aim to cover the main variation in Europe's biodiversity.

Species selection

Various authors have reviewed the selection of variables for indicators (Simberloff 1998; Hansson 2000; Carignan and Villard 2002). Noss (1990) distinguishes compositional, structural and functional variables. To achieve an indicator with the purpose and requirements mentioned above, we believe that species (compositional variables) have high potential, as species-based indicators are applicable to all ecosystems. Moreover, species are 1) a biodiversity key factor (sensu Larsson 2001) which is *directly* related to biodiversity; 2) the building bricks of ecosystems; 3) sensitive to change; 4) relevant and appealing to policy makers and the general public; 5) unambiguously defined; and 6) unambiguously measurable. Also they have (7) unambiguous relations with environmental conditions, i.e. they are suitable for modelling, which is important for scenario-analyses.

Various concepts have been used to select species for indicators, including keystone species, umbrella species, focal species and indicator species (resp. Paine 1995; Shrader-Frechette and McCoy 1993; Lambeck 1997; Landres et al. 1988). However, none of these concepts so far has shown a sufficiently strong link between the selected (set of) species and the components of biodiversity it was supposed to represent (Hansson 2000) In this conceptual framework we start with recognising that it is neither possible nor necessary to include all species in an index. Rather, we should try to achieve a smart sample of species, which is representative of the ecosystem as a whole.

The idea for selecting species is based on the shopping bag approach used in economics: to calculate the Retail Price Index a random sample is drawn from the products people buy in the shops. However, as in the real world it is not possible to achieve a completely random sample (monitoring is choosing), a set of criteria is used to guide the selection of species. We propose the following criteria to select a set of species for each of the categories of habitat × biogeographical region:

1. (core criterium) The set of species should be representative for the ecosystem (i.e. the categories) as a whole. This means that the set should represent the variation in (inter alia):
   - taxa
   - sub-habitats
   - abiotic conditions
   - trophic levels and guilds
   - spatial characteristics (minimum viable population area, dispersal range, migratory behaviour)
   - sensitivities to the major human pressures
   - common and rare, threatened and non-threatened species
   - endemism.
2. The individual species should be indigenous to the ecosystem, ie. alien invasive species should be avoided, to prevent ambiguous indicator results.
3. Given the criteria above, the set should also contain a *representative* share of policy-relevant species.

The selection of the species will have to be done by expert judgement, which must in turn be based on a thorough understanding of the natural history of the candidate organisms, their habitat affiliations and interactions with other organisms, and their roles in the broader ecosystem (Andreasen et al. 2001). Although the species selection will always be arbitrary to some extent, the criteria provide important guidance and make the species selection more objective. It is important to recognise that there is not one single correct set of species. To make the indicator robust, as many species as possible should be included. We believe at least some tens of species should be selected per category of habitat × biogeographical region.

As a tool to evaluate how well the selected set as a whole matches the criteria, we recommend the use of *ecological profiles*. An ecological profile of a species will contain various ecological, policy and management characteristics of that species, eg. its species group, habitats it uses, favourable abiotic conditions, trophic level and guilds, spatial characteristics, sensitivity to human pressures, threat status, endemism, policy status. By using the ecological profiles of all species of a set, it will be possible to obtain an overview of the representativeness of the set as a whole, by evaluating it against the criteria. For some of the criteria it might be possible to develop quantitative standards which should be met (e.g. % Red List species). For other criteria it might be more difficult to develop standards, but possibly minimum and maximum values can be set.

Population size

For each of the selected species, trends in population size should be estimated between two or more points in time. Population size is considered a sensitive measure for the decline of biodiversity as a result of short-term changes in human activity (Ten Brink et al. 1991; UNEP 1997b; Balmford et al. 2003). Data should be specified per habitat × biogeographical region, and per country as well. As an example: the system will have data for Atlantic forests in the United Kingdom, Atlantic forests in The Netherlands, Atlantic forest in Belgium etc. etc. Trends should be expressed as an index of change relative to a baseline situation. This will later allow aggregation of the results (see Calculation and aggregation).

The baseline can be a certain point in time or it can represent a certain state in the ecosystem. Mainly because of practical reasons of data availability (see Data availability) we propose the use of the year 1970 as a first attempt for a baseline for Europe, i.e. the index will be set at the value of 100 in the year 1970 for all habitats, biogeographical regions and countries. This means that the indicator will ignore loss of biodiversity which had already taken place due to human impacts before 1970. Also, *differences* between habitats, regions and countries which were already there in 1970 are ignored, as the baseline is a starting point in time, rather than a certain state.

The (trends in) population sizes will usually be based on absolute population counts, or on counts on sampling sites (monitoring scheme). However, quantitative data on trends in population size will not always be available. Therefore we propose to also consider *proxies* for species decline, such as trends in distribution area. This type of data may need additional interpretation in order to reflect the trends in population size. Another solution when quantitative data are not available, is the use of expert judgement to estimate trends (Failing and Gregory 2003).

Representativeness of data is crucial. For forest this means for example that data should come from production forests, semi-natural forests and natural forests. Forest type classifications, like the FTBA-system (Larsson 2001) are necessary to take fully into account the variation in Europe's forests (i.e. variation *within* the reporting units of the index) and thus achieve a truly representative picture.

Calculation and aggregation

For each of the combinations of habitat × biogeographical region × country the indicator is calculated as the average of the trends (indices) of the selected species. The results can then be aggregated towards the habitats or the biogeographical regions (and potentially also the individual countries). For example, by aggregating the results for forests in all

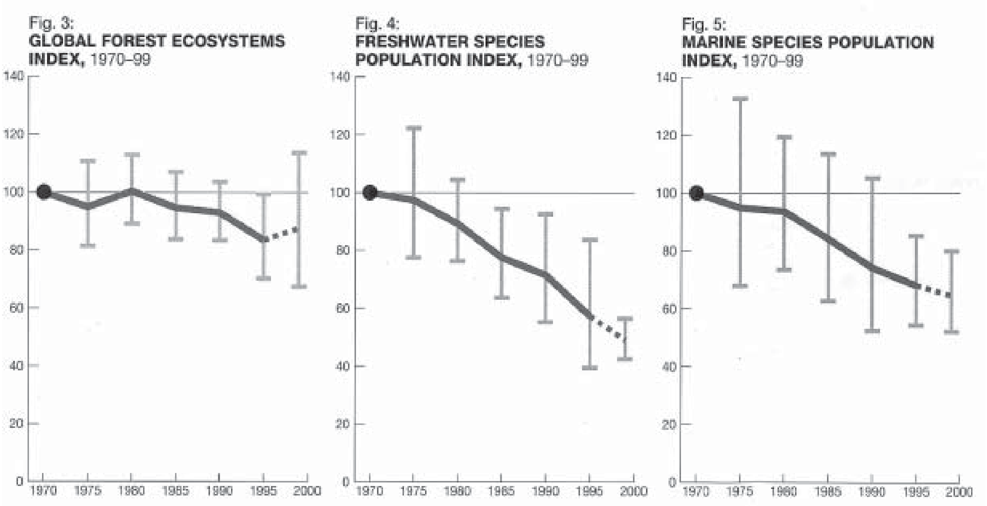

Figure 1. The Living Planet Index, showing trends since 1970 for three ecosystem types on the global scale: forest, freshwater and marine. The index is calculated as the average trend in sizes of populations of a set of some tens to hundreds of species for each of the ecosystem types (Loh 2002).

biogeographical regions and countries, an overall picture for European forest can be built. Results can then be presented in various graphical ways, as trend lines per habitat type, for example in a similar way as is done for the Living Planet Index (Loh 2002) (Figure 1).

3. Data availability

Data of various origins, collected with various methods, can be used for this indicator. As already outlined above, the requirement is that they are representative of their unit of habitat × biogeographical region × country and that they can be expressed as an index value relative to the baseline.

To explore the availability of data for this conceptual framework, an inventory of European datasources on species trends was carried out, with a focus on the terrestrial environment. From this study it appears that a considerable volume of data exists for the following species groups: birds (breeding and wintering), butterflies, mammals (large carnivores and large herbivores) and to a limited extent also for vascular plants (Appendix 1). Most of these data are held by or accessible via a small group of species-oriented NGOs (Appendix 1). These NGOs mostly are networks of experts and organisations from (nearly) all Pan-European countries, with a central coordination office. Monitoring and surveying initiatives are mostly country-based and the coordination office usually plays a role in exchange and mobilisation of experience and data. In addition to the official NGO publications, a lot of grey literature is available, which can potentially be mobilised as well. Besides, expert knowledge is present and useful for the interpretation of quantitative data, where there are problems as to representativeness.

The data are mostly available per country. The quality of the data is likely to vary. In some cases population sizes and/or trends are presented in broad classes (European Bird Database, Red Data book of European butterflies). To get an overview of the quality of the total data set, we propose to use a system for data quality. Each of the figures should be assigned a data quality indication. We suggest the following five-point scale:

Table 1. The number of selected butterfly species per category of habitat x geographical region for which data are available in one or more countries in Europe (Van Swaay in prep). The study was limited to those categories for which numbers are given in the table. Titles of the EUNIS habitat types have been shortened for this table; official codes in brackets.

	Alpine	Anatolian	Arctic	Atlantic	Black Sea	Boreal	Continental	Macaronesian	Mediterranean	Pannonian	Steppic
marine habitats (A)											
coastal habitats (B)				10					8		
freshwater habitats (C)											
mires, bogs and fens (D)				5							
grassland and tall forb habitats (E)	12			10			13		10		
heathland, scrub and tundra (F)				7					9		
woodland and forest (G)	8			11		10	13		9		
unvegetated habitats (H)	7										
cultivated agricultural habitats (I)	8			8		7	9		10	8	
artificial habitats (J)											

1. Reliable quantitative data
2. Limited quantitative data, some corrections and interpretations applied
3. Limited quantitative data, no corrections and interpretations applied
4. Extensive expert judgement
5. Limited expert judgement

Similar scales have been used by BirdLife International/European Bird Census Council (2000) and Van Swaay and Warren (1999).

The datasets for birds and butterflies and also for several of the large carnivores appear to start at/around the year 1970 (Appendix 1). The year 1970 therefore seems a feasible baseline year for the indicator.

Based on these data sources we estimate that per unit of habitat × biogeographical region × country trend figures will be available for an average of at least 10 species of each of the 4 species groups. This would result in a minimum of 40 species per unit. As an example we present the results of a detailed study on butterflies, showing that for each category of habitat × biogeographical region a set of 5–13 representative species can be selected, for which data are available in one or more countries (Van Swaay in prep) (Table 1).

Basically the NGO data cover all habitat types, including forests. At the moment we have no reason to expect that in these data sets the data volume or data quality for forests will be less than for other habitats. For example, data are available for 8–13 butterfly species of forests in each of the biogeographical regions, which is a similar number as for other habitats (Figure 1). As for other habitat types, data for forests will have their own specific problems, for example with regard to representation of different management regimes (natural, semi-natural and production forests).

Besides the species trend data which are collected by the NGOs, specific forest monitoring programmes are also increasingly collecting biodiversity trend data. The monitoring programme of ICP forests has included the monitoring of ground vegetation in the Level II plots since 1996 (EC-UN/ECE 2002). Data from the second round will become available in the coming years and will provide the opportunity for a first analysis of change. Also, an increasing number of countries include biodiversity parameters in their National Forest Inventories (Fischer 2002). These data from the forest programmes can potentially provide a substantial contribution to the total data pool available for biodiversity trend analysis.

4. Discussion

Classification

For this indicator any robust habitat classification can be used. For Europe we propose, at least provisionally, to use the EUNIS habitat classification (Moss and Davies 2002). The ten top-level habitats of the EUNIS system (major habitat types) provide meaningful and feasible reporting units for a policy relevant biodiversity indicator. The underlying habitat classes can be used to ensure that species selection is representative of the major habitat type as a whole, i.e. covers all sub-habitat types. Certainly the EUNIS classification has also disadvantages, but it is unlikely that any other classification will be able to overcome all problems, without at the same time creating new inconsistencies. Defining classes will always to some extent imply drawing subjective boundaries.

A total number of ten reporting units for Europe seems feasible in relation to data availability. More units might in some cases be desirable, e.g. to distinguish production forests from (semi-) natural forests or to distinguish production grassland from (semi-) natural grassland. However, at the moment this would require more information than is currently available.

Forest-specific variables

Potentially the indicator can also use forest-specific variables like dead wood and stand structure. To incorporate these variables they would need to be converted into measures of species abundance. More research will be needed to understand the quantitative relationship between trends in dead wood/stand structure and species abundance. If this relationship is strong and significant, the inclusion of structure variables in the indicator might be a very cost-effective approach.

Aggregation

Aggregation of the figures per species towards results per habitat type and/or biogeographical region is essential to achieve a concise, composite indicator which is easy to understand by policy makers and the general public. Various aggregation methods are possible and they should be chosen depending on the purpose of the indicator. More research will be needed to further investigate the aggregation methods.

Implementation with existing data

From the study on data availability it follows that data on species population trends do exist in Pan-Europe. With these data a first version of a European biodiversity index can be produced. Still, a number of problems deserves careful attention.

First, existing data are often available on the level of countries, but not per habitat or biogeographical region. This can provisionally be dealt with by limiting the selection of species to those which have their main distribution in just one category of habitat × biogeographical region. More research will be needed to explore to what extent this hampers the design of a robust set of species which is representative of the habitat and biogeographical region.

Second, it is unlikely that existing data will give a perfectly representative picture of Europe's biodiversity. We expect that much less data will be available for species groups other than the four discussed here, possibly with the exception of commercial marine fish. Moreover, also within the species groups we have investigated, data will not always be available for all species in all countries. Therefore, we recommend that implementation starts with using all data which are relatively easily available and evaluates how well these data match the criteria for species selection, using ecological profiles. Then it will then appear where exactly gaps and biases as to representativeness exist. These can possibly be dealt with by further targeted mobilisation of existing data. Also, biases towards groups of species might be dealt with in a statistical way, e.g. by (down-)weighting data.

Third, the quality of the available data will vary. When implementing the indicator, it will be essential to use a data quality system, in order to obtain an indication of the quality of the final results. Furthermore, many biases and gaps in time and space will occur. To some extent it will be possible to statistically repair these; see for example the computer programme TRIM by Pannekoek and Van Strien (2001). Where this is not possible, new monitoring will have to be set up, following pragmatic but statistically sound rules for design.

Pan-European biodiversity monitoring

For the structural implementation of the indicator, structural monitoring will need to be in place, to allow a frequent (once per ca. four or five years) update of the indicator. In Pan-Europe there is good potential for this, given the existing group of species-oriented NGOs and in forests the presence of the extensive network of ICP Forest plots and National Forest Inventories. For forests this would require more research on how to include biodiversity variables in the monitoring and further harmonisation of the NFIs between countries (Fischer 2002).

A set of indicators for 2010

In line with the CBD SBBSTA recommendations for evaluating the 2010 target we think this indicator should be part of a *set* of indicators to monitor status and trends in biodiversity. To give a complete picture of biodiversity state, an indicator on ecosystem extent and red list species should be used in addition to a species oriented indicator such as presented here. For the purpose of evaluating the 2010 target ('reducing the *rate* of loss of biodiversity'), as a baseline could be taken the period 1990–2000 (UNEP 2003d).

5. Conclusion

The index meets the challenge for indicator development as mentioned in the introduction of this paper: it is applicable to *all* ecosystems, including forests, which allows to compare between and aggregate across ecosystems. Moreover, the index is believed to have the following advantages: 1) it is part of the set of five indicators as recommended by SBSTTA for evaluating progress towards the 2010 target; 2) it can produce both headline messages for high-level policy-making and communication, and detailed information for in-depth analysis; 3) it will be able to produce consistent information from the (sub-)national to the regional and global scale; 4) by using existing data, it can be implemented immediately and then go through a gradual process of further development.

It will now be essential to perform a test on the conceptual framework, by actually producing the indicator, using existing data. Currently such a test is being carried out, using data on birds, butterflies, mammals and plants.

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Appendix 1. Data availability on species trends in Pan-Europe.

Species group	Organisation	Data source	Description
Birds	BirdLife International	European Bird Database (BirdLife International/European Bird Census Council 2000)	trends per species (n=515), per country, for 1970-1990. Currently an update till 2000 is being produced.
	European Bird Census Council (EBCC)	Pan-European Common Bird Monitoring Scheme (Gregory et al. 2003b)	trends for some tens of species of agricultural and forests habitats, for 18 countries, since ca. 1980
	Wetlands International	International Waterbird Census (IWC) (Gilissen et al. 2002)	trends per species (230 species counted) per region, longest time series go back some decades
Mammals	Large Carnivore Initiative Europe (LCIE)	Species Action Plans (Boitani 2000; Breitenmoser et al. 2000; Delibes et al. 2000; Landa et al. 2000; Swenson et al. 2000;) and related publications (see references in the SAPs)	status and trends for 5 large carnivore species, various starting years
	Large Herbivore Foundation (LHF)	no European overview available yet	for at least 7 species of large herbivores data exist for most of the relevant European countries. So far, these data have not been brought together.
Butterflies	Dutch Butterfly Conservation	Red Data Book of European butterflies (Van Swaay and Warren 1999) and underlying database	trends per species (n=576), per country, for 1970-present
Plants	Planta Europa	no European overview available yet	for at least ca. 10 European countries substantial datasets are available. The nature of these data varies a lot.

The MCPFE's Work on Biodiversity

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Abstract

The "Ministerial Conference on the Protection of Forests in Europe" (MCPFE) is an intergovernmental initiative for co-operation of 44 European countries and the European Community as well as 41 observer countries and organisations.

Since 1990 four Ministerial Conferences have taken place adopting 17 resolutions. Two resolutions deal explicitly with forest biodiversity, several others include issues relevant to forest biodiversity. In addition, the Pan-European Criteria and Indicators for Sustainable Forest Management (SFM) have been elaborated as a tool to monitor and assess sustainable forest management – including forest biodiversity – in Europe. Data based on the pan-European Indicators for SFM are regularly collected by the MCPFE in co-operation with international organisations.

The follow-up of the most recent Ministerial Conference, held in Vienna in April 2003, is outlined in the MCPFE Work Programme which includes several actions related to forest biodiversity.

Keywords: forest biodiversity, indicators, MCPFE.

1. Introduction

The Ministerial Conference on the Protection of Forests in Europe (MCPFE) as a policy platform for Europe has succeeded in establishing a consistent forum for dialogue on forests and their management in Europe. Since 1990 it has taken major steps towards sustainable development. With its main characteristics, i.e. the long-term commitment at the highest political level across Europe as well as the collaborative and flexible approach, the MCPFE has become the most important forest policy making entity in Europe.

The MCPFE involves 44 European countries, the European Community and, in addition, 41 observer countries and international organisations (Table 1). The MCPFE addresses forest

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Table 1. MCPFE participants.

European countries (44)
Albania, Andorra, Austria, Belarus, Belgium, Bosnia and Herzegovina, Bulgaria, Croatia, Cyprus, Czech Republic, Denmark, Estonia, Finland, France, Georgia, Germany, Greece, Holy See, Hungary, Iceland, Ireland, Italy, Latvia, Liechtenstein, Lithuania, Luxembourg, Malta, Moldova, Monaco, Netherlands, Norway, Poland, Portugal, Romania, Russian Federation, Serbia and Montenegro, Slovak Republic, Slovenia, Spain, Sweden, Switzerland, Turkey, Ukraine, United Kingdom
European Community
Observer countries (13)
Australia, Brazil, Cameroon, Canada, Chile, China, Ghana, India, Japan, Republic of Korea, Malaysia, New Zealand, USA
Observer organisations (28)
CEI Bois, CEPF, CEPI, COPA, Council of Europe, EFI, ELO, ENFE, EOMF, FAO, FECOF, Greenpeace International, IFBWW, IIASA, ILO, IPGRI, ITTO, IUCN, IUFRO, Montreal Process, UEF, UNDP, UNECE, UNEP, UNFF, UNU, USSE, WWF International

and environment-related policy issues of common relevance for all its participants. Solutions are developed in a joint and consensual effort.

2. Structure and working modalities of the MCPFE

The MCPFE is able to flexibly respond to and address urging issues and concerns of forest and environmental policies which are placed on the political agenda. Policies and actions are then developed in a joint effort, based on the consensus principle. The voluntary and non-institutional nature of the MCPFE leads to political commitments at Ministerial Conferences of the European countries and the European Community. The decisions taken at Ministerial Conferences then form the basis for the related follow-up work.

In the working process between Conferences, the decisions passed by the ministers are further specified and put into action. The MCPFE process consists of a series of meetings, namely Expert Level Meetings, the decision making body between Conferences and *ad hoc* working groups for the discussion of specific subjects, always involving all participants of the MCPFE.

The MCPFE is organised by the joint chairmanship of two countries with the support of two others. Together, these four countries form the General Co-ordinating Committee (GCC) which has the main tasks of funding the work of the MCPFE and providing guidance. These countries currently are Austria, Poland, Norway and Spain.

The secretariat of the process, the Liaison Unit, is responsible for the support of the GCC and the operations of the MCPFE in general. These range from the preparation of draft papers, reports and technical background documents to the organisation of all international MCPFE meetings and their follow-up and the representation of the MCPFE in other international meetings. The Liaison Unit has an important role as an information node for the MCPFE in general.

Table 2. Strasbourg Resolutions.

S1:	European Network of Permanent Sample Plots for Monitoring of Forest Ecosystems
S2:	Conservation of Forest Genetic Resources
S3:	Decentralized European Data Bank on Forest Fires
S4:	Adapting the Management of Mountain Forests to New Environmental Conditions
S5:	Expansion of the EUROSILVA Network of Research on Tree Physiology
S6:	European Network for Research into Forest Ecosystems

3. Ministerial Conferences in Strasbourg, Helsinki and Lisbon

3.1 Strasbourg 1990

The First Ministerial Conference on the Protection of Forests in Europe was held in Strasbourg in 1990. Against the background of the problems of forest degradation, the Conference was attended by 30 European countries and the European Community as well as by several intergovernmental organisations. The MCPFE served as a common European platform for collaboration before the institutional changes took place in Central and Eastern European Countries. Cross-border protection of forests in Europe and tackling the problem of forest dieback was at the core of the political decisions in Strasbourg. Six resolutions were adopted by the ministers responsible for forests (Table 2).

Resolutions S1 "European Network of Permanent Sample Plots for Monitoring of Forest Ecosystems", S2 "Conservation of Forest Genetic Resources" and S6 "European Network for Research into Forest Ecosystems" address the pan-European co-operation on monitoring and research related to forest ecosystems including biodiversity aspects.

3.2 Helsinki 1993

The Second Ministerial Conference in Helsinki was driven by the decisions of the United Nations Conference on Environment and Development (UNCED) 1992 in Rio de Janeiro with regard to the concept of sustainability. This marks also a shift towards a more policy oriented direction of the MCPFE. In addition to 37 European countries and the European Community, several organisations from the private sector, the international forestry community and environmental NGOs as well as the scientific community participated in this conference.

The General Declaration and the four Helsinki Resolutions reflect Europe's approaches to global environmental issues, namely promotion of sustainable forest management, conservation of biological diversity, strategies regarding the consequences of a possible climate change on forests, and an increasing co-operation with countries in transition to market economies (Table 3).

Resolution H1 "General Guidelines for the Sustainable Management of Forests in Europe" captures the common understanding of European countries on the most important aspects of sustainable forest management (SFM). Consequently, the European forest ministers agreed on a definition of SFM. Resolution H2 "General Guidelines for the Conservation of the Biodiversity of European Forests" recognises the conservation and enhancement of the biological diversity of

Table 3. Helsinki Resolutions.

H1:	General Guidelines for the Sustainable Management of Forests in Europe
H2:	General Guidelines for the Conservation of the Biodiversity of European Forests
H3:	Forestry Co-operation with Countries with Economies in Transition
H4:	Strategies for a Process of Long-Term Adaptation of Forests in Europe to Climate Change

Table 4. Lisbon Resolutions.

L1:	People, Forests and Forestry – Enhancement of Socio-Economic Aspects of Sustainable Forest Management
L2:	Pan-European Criteria, Indicators and Operational Level Guidelines for Sustainable Forest Management

forests as an essential element for SFM. It sets out general guidelines as well as a number of actions addressing forest biodiversity at all levels, i.e. the genetic, species and ecosystem level. In the implementation of Resolutions H1 and H2 the MCPFE has developed Pan-European Criteria and Indicators (C&I) as a common policy instrument for monitoring, assessing and reporting on progress towards sustainable forest management in Europe.

3.3 Lisbon 1998

The Third Ministerial Conference, held in Lisbon in June 1998, focused on the relationship and interaction between the forest sector and society and socio-economic aspects of sustainable forest management. 36 countries and the European Community signed two resolutions (Table 4).

Resolution L2 focuses on Pan-European Criteria and Indicators for SFM. At this conference the MCPFE also reported on the status of SFM in Europe, including biodiversity in European forests, by using the pan-European C&I. Furthermore a joint Biodiversity Work-Programme was endorsed by the Ministers responsible for Forests in Europe at the 3rd Ministerial Conference in Lisbon and at the 4th Ministerial Conference "Environment for Europe" in Århus/Denmark in June 1998. About 200 activities at different levels contributed to the implementation of this work programme focusing on the enhancement of biodiversity in SFM, the conservation of all types of forests through protected forest areas, the role of forest ecosystems in landscape diversity and the clarification of impacts on forest biodiversity from other sectors.

4. The Vienna Conference – Living Forest Summit 2003

The Fourth Ministerial Conference on the Protection of Forests in Europe – the Living Forest Summit – took place from 28 to 30 April 2003 in Vienna, Austria. The conference provided an opportunity to give a strong political signal on the role of forests in Europe, their

Table 5. Vienna Declaration and Vienna Resolutions.

VD:	European Forests – Common Benefits, Shared Responsibilities
V1:	Strengthen Synergies for Sustainable Forest Management in Europe Through Cross-sectoral Co-operation and National Forest Programmes
V2:	Enhancing Economic Viability of Sustainable Forest Management in Europe
V3:	Preserving and Enhancing the Social and Cultural Dimensions of Sustainable Forest Management in Europe
V4:	Conserving and Enhancing Forest Biological Diversity in Europe
V5:	Climate Change and Sustainable Forest Management in Europe

protection and sustainable management. Forty European states and the European Community signed the Vienna Declaration and five Vienna Resolutions (Table 5). 24 observer organisations and institutions as well as 4 observer countries also participated in the Conference and contributed actively to the multi-stakeholder dialogue (MSD).

The Vienna Living Forest Summit Declaration emphasises the multiple benefits which has to be taken into account for a future-oriented forest policy. In this respect co-ordination and partnerships with other sectors leading to shared responsibilities are highlighted. The main commitments made through the Vienna Declaration aim at benefiting rural livelihoods and urban societies, building strong partnerships, tackling global challenges and putting commitments of the MCPFE into action through measures and activities defined in the Declaration. In addition, in the Vienna Living Forest Summit Declaration the forest ministers endorsed the use of the Improved Pan-European Indicators for Sustainable Forest Management.

Vienna Resolution 1 aims at involving all interested sectors and groups in a dialogue by highlighting various means and approaches for the future. Through this resolution also the "MCPFE Approach to National Forest Programmes in Europe" as an instrument for optimising this objective has been adopted.

The promotion of the use of wood as an environmentally sound and renewable resource as well as the use of non-wood goods and services are highlighted through Vienna Resolution 2. Furthermore, the promotion of innovation and entrepreneurship, the enhancement of workforce know-how as well as workforce safety are main commitments of this resolution.

Vienna Resolution 3 gives increased attention to the cultural dimension of forest policy making. The promotion of the assessment of historical and cultural sites, securing property rights and land tenure arrangements and the promotion and the communication of the social and cultural dimensions are central commitments of this resolution.

Vienna Resolution 4 "Conserving and Enhancing Forest Biological Diversity in Europe" identifies pan-European issues building on international commitments of CBD, UNFF and WSSD. Policy planning and implementation in line with the conservation of forest biological diversity, combating illegal harvesting and related trade, further developing protected forest area networks, restoring biological diversity in degraded forests, preventing negative impacts of invasive alien species and monitoring the development of forest biological diversity as well as the linkage between the ecosystem approach and SFM are key commitments of this resolution. In addition, the "MCPFE Assessment Guidelines for Protected and Protective Forest and Other Wooded Land in Europe" as well as a "Framework for Co-operation between the MCPFE and Environment for Europe/PEBLDS" on key issues of forest biodiversity were adopted through this resolution.

Finally, Vienna Resolution 5 recognises the need to further promote the concept of sustainable forest management in the context of the continued debate on climate change and forests to ensure the multiple benefits of forests in the long run. Promoting the use of wood as an environmentally sound and renewable resource and as the alternative to non-renewable products is highlighted in this respect. Forest biodiversity is addressed in a commitment referring to afforestation and reforestation in the frame of the UNCFFF and the Kyoto Protocol.

The report "Implementation of MCPFE Commitments – National and Pan-European Activities 1998–2003", which was presented at the Living Forest provides information on the implementation of MCPFE commitments at both national and pan-European level. The report analyses the implementation of commitments made at the 3rd Ministerial Conference in Lisbon (1998) as well as further progress in implementing commitments made at the Ministerial Conferences in Helsinki (1993) and in Strasbourg (1990).

4.1 Improved Pan-European Indicators for SFM – Biodiversity Assessment in Forests

The first set of Pan-European Criteria and Indicators for Sustainable Forest Management had been developed in the early 1990s. In the meantime information needs, knowledge and data collection systems have developed further. At the 3rd Ministerial Conference, in Lisbon in 1998, it was not only decided to implement but also to continuously review and further improve the Pan-European Indicators for SFM. Subsequently, the MCPFE conducted political and technical evaluations of the usefulness, strengths, weaknesses and feasibility of the existing Pan-European Indicators for SFM. An MCPFE Advisory Group of scientific and technical experts from relevant key organisations in Europe was formed to elaborate recommendations for an improved set of Pan-European Indicators for SFM, based on the existing criteria and indicators. In order to reflect the diversity of national situations and the wealth of existing knowledge and experiences the MCPFE organised consultations with a wide range of experts in four workshops during 2001 and 2002. Consequently, a list of improved indicators was discussed by the MCPFE and adopted at expert level. The use of these Improved Pan-European Indicators for SFM was finally endorsed by the ministers at the 4th Ministerial Conference in Vienna in 2003.

Biodiversity aspects are addressed by Criterion 4: "Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems". The following nine quantitative indicators, one quarter of the 35 pan-European quantitative indicators, are used to report on forest biodiversity aspects under this criterion (Table 6).

The report "State of Europe's Forests 2003", which has been jointly prepared by the MCPFE Liaison Unit Vienna and UNECE/FAO for the Vienna Conference, gives an overview of the latest facts and figures on forests and the status of sustainable forest management in Europe. It is structured according to the six Pan-European Criteria for SFM. Data on forest biodiversity is an essential part of the report.

4.2 The MCPFE Assessment Guidelines for Protected and Protective Forest and Other Wooded Land

One topical work of the MCPFE on forest biodiversity focused on the elaboration of a common approach to the assessment of protected and protective forest and other wooded land in Europe (Table 7). As recent reporting activities could not capture the diversity of protection regimes the MCPFE Assessment Guidelines were developed. They aim to give a comprehensive picture of protected and protective forest and other wooded land in Europe, while also keeping links to international classification systems for protected areas, especially to IUCN.

Table 6. Pan-European indicators for SFM under Criterion 4.

Indicator 4.1	Area of forest and other wooded land, classified by number of tree species occurring and by forest type
Indicator 4.2	Area of regeneration within even-aged stands and uneven-aged stands, classified by regeneration type
Indicator 4.3	Area of forest and other wooded land, classified by "undisturbed by man", by "semi-natural" or by "plantations", each by forest type
Indicator 4.4	Area of forest and other wooded land dominated by introduced tree species
Indicator 4.5	Volume of standing deadwood and of lying deadwood on forest and other wooded land classified by forest type
Indicator 4.6	Area managed for conservation and utilisation of forest tree genetic resources (in situ and ex situ gene conservation) and area managed for seed production
Indicator 4.7	Landscape-level spatial pattern of forest cover
Indicator 4.8	Number of threatened forest species, classified according to IUCN Red List categories in relation to total number of forest species
Indicator 4.9	Area of forest and other wooded land protected to conserve biodiversity, landscapes and specific natural elements, according to MCPFE Assessment Guidelines

Table 7. MCPFE Assessment Guidelines.

MCPFE Classes		EEA	IUCN
1: Main Management Objective "Biodiversity"	1.1: "No Active Intervention"	A	I
	1.2: "Minimum Intervention"	A	II
	1.3: "Conservation Through Active Management"	A	IV
2: Main Management Objective "Protection of Landscapes and Specific Natural Elements"		B	III, V, VI
3: Main Management Objective "Protective Functions"		(B)	n.a.

The MCPFE Assessment Guidelines paved the way for collecting comparable data on indicator 4.9 "Protected Forests". In the "State of Europe's Forests 2003" report comprehensive information on protected forests in 34 European countries was, for the first time, presented according to the detailed guidelines.

4.3 The Framework for Co-operation between MCPFE and EfE/PEBLDS

At the regional European level, the MCPFE has been acknowledging the importance of creating and enhancing mutual supportiveness with other relevant instruments and processes related to forests and biodiversity. Following the positive and constructive experience gained in the past with the Biodiversity Work Programme, the ministers responsible for forests and for the environment confirmed the importance to continue this co-operation also in the future.

For the purpose of future co-operation a Framework for Co-operation between MCPFE and Environment for Europe/PEBLDS was elaborated and adopted at ministerial level by both processes in spring 2003. The contribution to the implementation of global provisions, in particular the expanded work programme on forest biological diversity of the CBD, constitutes an important aspect of this framework. Priority themes for co-operation between MCPFE and EfE/PEBLDS for the period 2003–2005 are the following:

- The theme "*Ecosystem approach*" will contribute to the clarification of the relationship between the Ecosystem Approach and Sustainable Forest Management, building on the work achieved so far by MCPFE on SFM.
- The theme "*Protected forest areas*" will contribute to the global work on protected forest areas and simultaneously contribute to the general work on protected areas for CBD-COP7 (2004) by making a link between the concepts of protected forest areas and protected areas in general. In the pan-European context, the work will build on existing work on protected areas of the MCPFE and current work on ecological networks.
- The theme "*Forest law enforcement with regard to biodiversity conservation"* is a global cross-cutting issue, which is also of pan-European relevance and refers to the impacts of illegal harvesting and related trade and institutional capacity building.
- The elaboration of "*Recommendations for site selection for afforestation"* in the context of the decisions of the UNFCCC and its Kyoto Protocol, taking account of biodiversity interests, was identified as a fourth area of co-operation. This work will build on recent work by IUCN and UNEP, adapted to the European situation.

5. Follow-up of the Vienna Conference

In addition to national level implementation an MCPFE Work Programme, containing 30 pan-European actions, was adopted at the Expert Level Meeting in October 2003. It is structured according to the three pillars of sustainable forest management and aims to contribute to the sustainable development of society at large. All MCPFE Resolutions and in particular the Vienna Declaration and the five Vienna Resolutions are put in relation to the concept of SFM, indicating the main pan-European issues of each Resolution.

Pan-European actions for implementation of Vienna Resolution 4 address *inter alia* the development of a pan-European understanding on the linkage between the ecosystem approach and SFM, an analysis of European networks of protected forest areas concerning the comprehensiveness, representativeness and adequacy of protected forests with regard to the conservation goal, forest genetic diversity as well as the elaboration of a pan-European understanding on forest classification.

In addition to the ongoing MCPFE activities on SFM related monitoring, assessment and reporting through criteria and indicators the MCPFE Work Programme contains also explicit actions facilitating the C&I work, *inter alia* in 2006 a MCPFE workshop on a pan-European understanding of forest classification is scheduled in co-ordination with UNECE/FAO, IUCN and EEA.

The MCPFE Work Programme also aim to contribute to the further implementation of global commitments, such as those agreed at WSSD, UNFF, CBD, UNFCCC and UNCCD, and to maintain linkages with other regional processes and initiatives as noted in the Vienna Declaration and the Vienna Resolutions. The Work Programme also follows a co-operative approach with all MCPFE partners, i.e. relevant organisations, initiatives and stakeholders which is a precondition for a fruitful follow-up of the Vienna Conference.

List of Abbreviations

CBD	Convention on Biological Diversity
C&I	Criteria and indicators
COP	Conference of the Parties (of the CBD)
EfE	Environment for Europe
FAO	Food and Agriculture Organization of the United Nations
H	Helsinki Resolution
IUCN	The World Conservation Union
L	Lisbon Resolution
MCPFE	Ministerial Conference on the Protection of Forests in Europe
PEBLDS	Pan-European Biological and Landscape Diversity Strategy
S	Strasbourg Resolution
SFM	Sustainable forest management
UNECE	United Nations Economic Commission for Europe
UNEP	United Nations Environment Programme
UNFCCC	United Nations Framework Convention on Climate Change
UNFF	United Nations Forum on Forests
WSSD	World Summit on Sustainable Development

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Forest Biodiversity Indicators – A Contribution to an EEA Core Set of Biodiversity Indicators

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Abstract

Various driving forces have an influence on forest biodiversity. To assess their impact and the current state of forest biodiversity different initiatives are developing indicators that contribute to this goal. The main European initiatives emerge from the Ministerial Conference on Protection of Forests in Europe and the concerted action Indicators for Monitoring and Evaluation of Forest Biodiversity in Europe. Also the European Environment Agency is currently setting up a set of indicators where forest aspects are included within the biodiversity section. This paper presents a selection of indicators highlighting the multiple aspects of biodiversity in relation to pressures and societal demands towards forests. The indicators relate to topics urban forestry, genetic aspects of forest conservation and forest certification. They can be seen as a contribution to a core set of indicators promoted by the European Environment Agency and find application in policy and decision making processes.

Keywords: forest biodiversity, indicators, impacts.

1. Introduction

Forests constitute an important natural resource. They cover about one third of the European land area, totalling 3.2 million km^2. They form a major component of the landscape and provide a wide range of goods and services for society. These include renewable fibre and timber resources and non-wood goods and services. Not only due to their share in the total land cover but importantly also because of their diversity in structure and species composition, forests are key reserves of Europe's biodiversity and provide important ecological functions. They serve as carbon sinks, improve water quality and protect soils and are also of great value for tourism, recreation and education. An important characteristic of European forests is that within each country there exists own management culture and specific goals, varying ownership structures and particular societal demands and pressures towards forests (e.g. climate change, biodiversity

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loss, illegal logging). This is one of the reasons that European forests are subject to many political initiatives and processes at different levels.

The 6th Environment Action Programme of the European Union calls for policy making based on sound knowledge of environmental issues, their geographical distribution and driving forces (COM (2001) 31 final, 2001/0029 (COD)). Means for implementation will require the collection of relevant data while ensuring their continuous availability over time. Only this will allow adequate analysis and interpretation to support policy making. The four groups of priority issues are: (1) tackling climate change, (2) nature and biodiversity, (3) environment and health and (4) ensuring the sustainable management of natural resources and wastes.

The European Environment Agency (EEA) plays an important role in this context. Its strategy for reporting on environmental trends is based on an indicator approach. Recent EEA reports including forestry issues are 'Environmental Signals 2002 – Benchmarking the Millennium' (EEA 2002) and 'Europe's Environment: the Third Assessment' (EEA 2003). The latter was prepared for the 5th Pan European Ministerial Conference 'Environment for Europe' held in Kiev, Ukraine (21–23 May 2003).

The EEA is presently developing a core set of policy-relevant indicators for six environmental issues (air pollution, climate change, water, waste and material flows, biodiversity and terrestrial environment) and five sectors (transport, energy, agriculture, tourism and fisheries). This core set provides a key part of the framework within which the EEA with its partners will gradually contribute to the shared European Environmental Information System. The work related to biodiversity aspects is coordinated by EEA's European Topic Centre on Nature Protection and Biodiversity (ETC/NPB) of which the European Forest Institute (EFI) is one core partner. EFI contributes to the work of the ETC/NPB by identifying forest-related indicators at the landscape level.

The process of the Ministerial Conference on the Protection of Forest in Europe (MCPFE) plays a crucial role in developing a set of criteria and indicators for sustainable forest management. The 4th MCPFE in Vienna 28–30 April 2003 has adopted an improved set of criteria and indicators. A set of 35 indicators is grouped under 6 criteria one of which relates to forest biodiversity (C4: Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems). Signatory countries now face the challenging task of developing effective approaches for monitoring and implementation of the improved set of criteria and indicators of sustainable forest management (MCPFE 2003a).

In order to monitor and report on indicators a consistent flow of forest/forestry information and data are crucial. A pre-requisite for monitoring changes is the need for internationally agreed definitions. This is the case for forest resources assessments implemented by the United Nations/Food and Agriculture Organization and United Nations Economic Commission for Europe/Food and Agriculture Organization (UNECE/FAO 2000). Also forestry statistics provided by Statistical Office of the European Communities are of importance, as well as those from other international data collection initiatives related to forests and forestry (MCPFE 2003b).

This paper presents a selection of topics highlighting the multiple aspects of biodiversity in relation to pressures and societal demands towards forest. It will investigate the applicability of such indicators and their potential to serve as input to policy and decision making process.

2. Forest biodiversity indicators – MCPFE and the EEA Core Set

The EEA core set of indicators includes 63 indicators under the topic of biodiversity. Six of nine improved pan-European indicators of sustainable forest management of the MCPFE that

Table 1. MCPFE indicators under Criterion 4 adopted by the EEA core set of biodiversity indicators.

MCPFE Criterion 4: 'Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems' indicators	EEA core set (taken from MCPFE indicators)
Tree species composition (4.1)	Tree species composition in forests (BDIV02d)
Regeneration (4.2)	Not represented in the EEA core set
Naturalness (4.3)	Naturalness of forests (BDIV01d)
Introduced tree species (4.4)	Introduced tree species (BDIV07c)
Genetic resources (4.6)	Forest genetic resources (BDIV04a)
Deadwood (4.5)	Deadwood (BDIV14); not yet developed; no reference to MCPFE indicator
Landscape pattern (4.7)	Landscape-level spatial pattern of forest cover (BDIV06a)
Threatened forest species (4.8)	Threatened forest species (BDIV03e)
Protected forests(4.9)	Not represented in the EEA core set

Note: EEA core set at: http://ims.eionet.eu.int/Topics/BDIV/indicators; visited 08.01.04.

are related to forest biodiversity have been adopted within the EEA core set (Table 1). The MCPFE indicators Area of regeneration within even-aged stands and uneven-aged stands, classified by regeneration type, Deadwood Volume of standing deadwood and of lying deadwood on forest and other wooded land classified by forest type, Protected forests Area of forest and other wooded land protected to conserve biodiversity, landscapes and specific natural elements are to date not included in the EEA core set.

Apart from the indicators mentioned above numerous other aspects of forests and biodiversity will need to be given attention, in order to find synergies on the one hand and conflict areas on the other.

Such effects can be illustrated by looking at the influence of management practices on biodiversity and carbon sequestration. The afforestation of native grasslands for example will increase the potential of carbon sequestration, while it may have a negative influence on biological diversity. Prescribed burning will have a negative effect on carbon sequestration but may allow an increase in biological diversity.

3. Assessing impacts on forest biodiversity

Within the EEA core set of indicators framework, EFI has identified and elaborated on behalf of the ETC/NPB potential indicators for development in the short term and allow continuous monitoring. Assessments have been made on the influence of different driving forces on forest biodiversity for the issues (1) climate change, (2) urban forestry, (3) tourism, (4) carbon sinks, (5) forest certification, and (6) genetic aspects of forest conservation. Within these indicators, some sub-indicators were identified. Selected indicators covering each

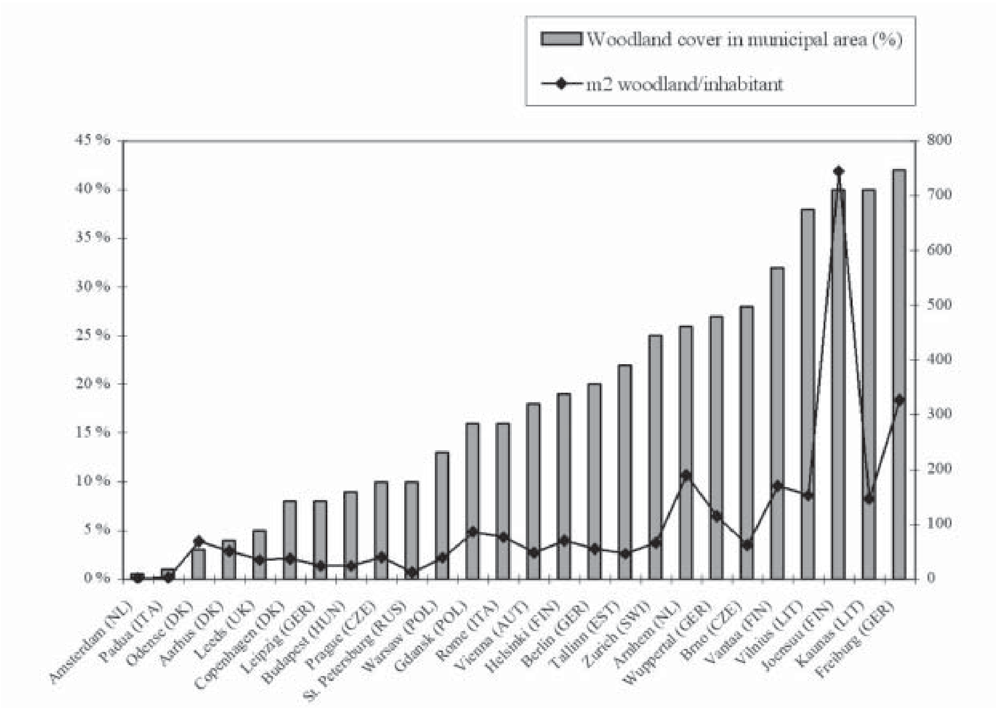

Figure 1. Woodland cover as part of the municipal area and in m^2 per inhabitant for selected European cities (Konijnendijk 2001). Note: data mainly provided by municipalities. Included are land areas defined as woodlands in the case of each specific city.

theme were described following the standard Indicator Fact Sheet model established by the EEA. The indicators were identified using a compromise between data availability and the soundness of the indicator.

The following chapters highlight the topics urban forestry, genetic aspects of forest conservation and forest certification with regard to the multiple aspects of biodiversity and the relation to pressures and societal demands. Further the applicability is investigated to serve as input to policy and decision making.

3.1 Example: urban forestry

Urban forests and green areas are increasingly recognised as being important for the overall quality of human life, particularly given the ongoing urbanisation process across the globe. The indicator selected is the urban forest index.

Urban Forest Index: Urban woodland cover in European cities

Within urban green structures, woodlands play a crucial role as popular recreational environments for urban dwellers. Woodland cover and availability of woodland per inhabitant differ considerably across Europe, with higher cover and availability in Nordic and Eastern European countries.

The urban forest index can be used to assess the availability of one of the main components of urban forests, i.e. urban woodlands, for a city/urban area. Within an urbanised society, trees, gardens, parks and woodlands contribute to the quality of living, working and leisure environments through a wide range of functions. These functions are based strongly upon social, cultural-historical and environmental values. As a result urban green spaces and urban forests have considerable impact on urban societies in terms of for example enhancing human health and wellbeing. Also the quality of the urban environment is becoming recognised as an integral component to the economic and social development of urban areas. Urban green spaces are in addition instrumental from a perspective of conservation and biodiversity (EEA 1995, 1999; World Resources Institute 1996; EC 1998).

The urban forest index is in particular relevant with regard to current policy focus on woodlands situated in and near urban areas as nearby recreation environments and providers of multiple benefits. Several European countries have afforestation policies focusing on urban centres. The indicator can be used to identify urban areas with low forest cover and allow monitoring of changes in forest cover/availability over time.

Urban forests often hold a special role due to their specific structure, species composition and history. Urban forests are also in many cases quite valuable with regard to biodiversity. Studies have shown that species richness within cities' boundaries can be surprisingly high, sometimes even surpassing that of surrounding areas (Deelstra 1990). The share of exotic species (trees, other vegetation and fauna) is usually higher in urban forests than in forests elsewhere (Profous and Rowntree 1993). Even if not very rich in natural biodiversity, they can be very important from a biodiversity perspective as they offer urban people a chance to experience the richness of nature in close proximity. By offering easily accessible recreation areas, urban forests can contribute to reducing pressures on more remote and vulnerable forests and nature areas. Urban forests also have the potential to contribute to reducing negative impacts of cities on biodiversity by means of absorbing pollution and noise and protection of drinking water resources.

3.2 Example: genetic aspects of forest conservation

Genetic diversity is the basis of the ability of organisms to adapt to changes in their environment through natural selection. Populations with little genetic variation are more vulnerable to the arrival of new invasive species, pests or diseases, pollution, changes in climate and habitat destruction due to human activities or other catastrophic events. The inability to adapt to changing conditions greatly increases the risk of extinction. Adequate genetic diversity, however, is important for future productivity of forests, the maintenance of this biodiversity depends strongly on forest management and related policies. For these reasons sustainable forest management emphasises forest genetic resources conservation (MCPFE 2003a).

Two indicators are presented below, namely gene conservation networks in Europe and national programmes on forest genetic resources.

Gene conservation network in Europe

The Convention on Biological Diversity and the MCPFE are the main foundations for the protection of forests in different countries. As a result an extensive programme dealing with genetic aspects of forest conservation, the European Forest Genetic Resources Programme (EUFORGEN), was established as a mechanism to implement Resolution S2 Conservation of Forest Genetic Resources of the MCPFE (1990). It became operational in 1994. This

Table 2. Participating countries in EUFORGEN networks (total number of networks joined).

Countries	Number of EUFORGEN networks joined
Bulgaria, France, Germany, Italy Malta Slovenia Turkey United Kingdom	5
Austria, Belgium, Croatia, Czech Republic, Hungary, Ireland, Poland, Portugal, Slovakia, Spain, Sweden, Switzerland	4
Finland, Lithuania, Moldova, Netherlands, Norway, Ukraine	3
Albania, Denmark, Estonia, Luxembourg, Macedonia, Romania, Russia, Serbia & Montenegro, Bosnia & Herzegovina	2
Armenia, Cyprus, Latvia, Georgia, Azerbaijan	1

Note: Figure was initially elaborated with data from EUFORGEN website (www.euforgen.org). The Chairs of the Networks and the EUFORGEN coordinator provided updates of the original table. Last update 4.08.2003 (Mediterranean Oaks and *Populus nigra*).

collaborative programme among European countries has the objective to ensure effective conservation and sustainable use of forest genetic resources in Europe. It has become one of the main programmes in Europe related to forest genetic conservation. It is moving towards common systems of information sharing by developing mechanisms for information access available to researchers, scientists and policy-makers. EUFORGEN involves 5 networks, namely the Conifers, Mediterranean Oaks, Noble Hardwoods, *Populus nigra*, and Temperate Oaks and Beech Networks (Table 2).

EUFORGEN has the potential to provide expert input and technical support to European forest policy in the field of use and conservation of forest genetic resources. For this purpose, it has to date produced several technical guidelines (www.euforgen.org). Furthermore, EUFORGEN Networks were encouraged to play a vital role in identifying priorities and in supporting the implementation process of the new European Council Regulation on Genetic Resources in Agriculture, which includes forestry ecosystems (IPGRI 2003). EUFORGEN also contributes to facilitating information flow among countries and raising public awareness on the conservation of forest genetic resources.

Genetic variation within species is important as growth and resistance to various stresses depend on such variation (FAO 2002; Paul et al. 2000). Adequate genetic diversity is important for the future productivity of forests and the maintenance of this biodiversity depends strongly on forest management practices and related policies.

National programmes on forest genetic resources as an integral component of national forest programmes

For an adequate genetic conservation there is firstly a need for the existence of a national programme on forest genetic resources with clear objectives. Less than 30% of the European countries have such programmes. More efficient conservation of forest genetic resources will require that such programmes are strongly linked to overall National Forest Programmes (NFPs) that establish policies and resources for implementing sustainable forest management at national level (Koskela et al. 2003).

The existence of a formal national programme on forest genetic resources with well stated objectives and funding is a relatively good indicator for the level of gene conservation activities in a given country. EUFORGEN has contributed to the establishment and implementation of such programmes/strategies on forest genetic resources at national level. Countries that have integrated their national programmes on forest genetic resources into

NFPs, indicate a strong willingness to actively conserve forest genetic resources. Approximately two thirds of the European countries have a NFP, but they often do not emphasise the importance and benefits of gene conservation efforts to the forest sector as a whole very clearly.

3.3 Example: forest certification

The impact of forest certification on forest biodiversity is difficult to assess in view of the limited practical experience. Certification is a quite new tool and the amount of certified forests has been growing substantially only during the last years. Two main indicators can be determined, giving an indication on the influence of certification on forest biodiversity. These are area covered by certification and characteristics of these forests and elements of biodiversity and biodiversity standard requirements used.

The modern concept of sustainable forest management includes all forest values: social, economic, environmental, cultural and spiritual. The definition of sustainable forest management used in international forest policy contexts, such as the elements expressed in the UNCED Forest Principles, the International Timber Trade Organization or the MCPFE definitions, reflects this multidimensional complexity. There is a close relation between criteria and indicators for sustainable forest management and forest certification standards. The primary differences rest in the degree to which the procedures are binding. Forest certification standards are performance standards, setting minimum requirements for the attributes to be evaluated. It reflects a voluntary activity between market actors including industry, trade, consumers, forest owners, and environmental groups all driven by a variety of interests and investors.

Area covered by certification and characteristics of these forests

In 23 European countries 72.6 million hectares of forests, representing 47% of their total forest area were certified voluntarily by third party forest certification until December 2003 (Figure 2). Two international systems, namely the Forest Stewardship Council (FSC) and the Pan European Forest Certification (PEFC), are in operation; and the amount of certified forests is increasing steadily.

Forest certification is directed primarily at the management of and the production of goods from multifunctional forests, and can be regarded as a simple tool to communicate and acknowledge that wood products derive from well-managed forests. Maintaining high biodiversity is one important goal of certification, but needs to be balanced with other components of the modern concept of sustainable forest management. As a market driven tool forest certification is an effective means to increase awareness of forestry in society. By creating a positive image for wood as a renewable material, certification can promote sustainability as an asset in comparison to other non-renewable materials. The difficulties faced within the international debate are what constitutes credible certification schemes and whether or how cooperation between individual schemes can be arranged. Mutual recognition has been proposed as one solution to overcome the problem of proliferation of national certification schemes.

The basis for any sort of certification scheme that is aiming to promote sustainable forest management is the prevailing forest infrastructure, legislation, guidelines for silviculture, dissemination of information in the society and the possible support mechanisms for private forest owners in countries where private forest ownership dominates. In countries with a long forestry tradition, stable forest organisations and ownership or strict legislation, the additional

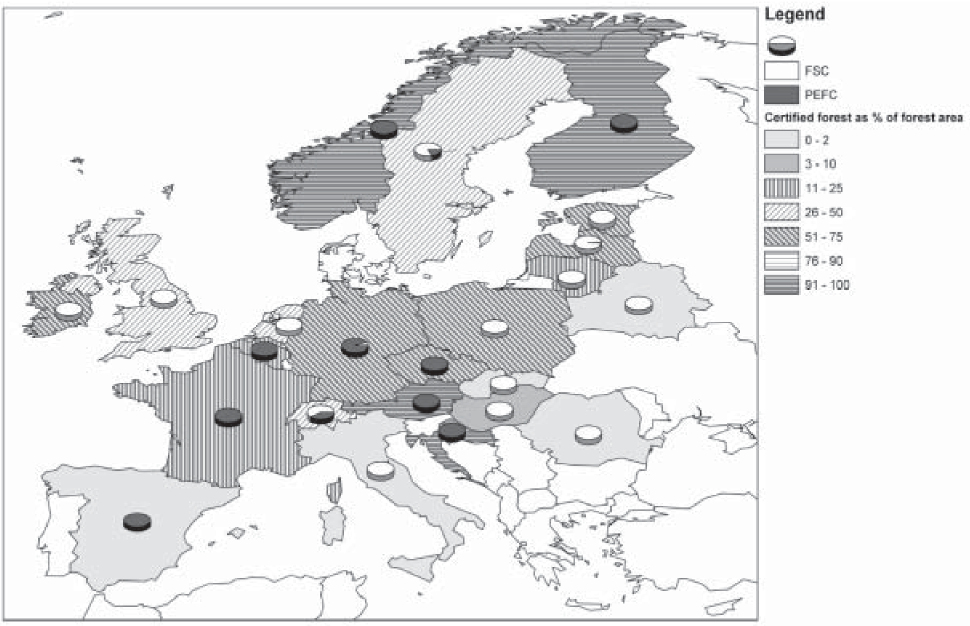

Figure 2. Shares of PEFC and FSC certified forests of the total forest area in European countries (in %). Sources for 'certified forest' are the web pages: www.pefc.org and www.fscoax.org (December 2003); Forest area is taken from: UN-ECE/FAO (2000). Countries where amount of certified forest was lower than 10 000 hectares are not included.

value of forest certification for promoting the biodiversity may be low. This is the case in many of the European countries. In some cases, forest managers may have to implement a set of significant biodiversity improvements in order to be awarded a certificate. Nevertheless, it should be kept in mind that also non-certified forests can be managed sustainably and provide high biodiversity values.

Elements of biodiversity and biodiversity standard requirements used

Biodiversity standards considered within forest certification schemes vary considerably across Europe, reflecting the diversity of vegetation zones, naturalness of the forests, traditional forest use, fragmentation, forest ownership structure, stakeholder views and local conditions. Common denominators are preservation of habitats, endangered species, nature value trees and dead wood component, exotic/native tree species and species mixture. The setting of quantitative threshold values has been made in consensus, due to deficiency of monitoring data (Table 3). The threshold values again may differ between countries and certification schemes as illustrated for the 'dead wood (nature value trees) component (Table 4).

The standard requirements of certification schemes are the most essential factors that can have an effect on biodiversity and create credibility for certification. Most standards have elements, which represent broader requirements than the legal performance for forest management. Standards may base on silvicultural/logging guidelines or recommendations. Often these standards emphasise specific aspects that are of critical importance for environmental impacts at local level.

When looking at the two main certification schemes, FSC and PEFC, no essential differences can be identified with regard to biodiversity characteristics included in their

Table 3. Biodiversity standards in most forested countries in Europe.

1. Preservation of habitats:	
a) key biotopes	a) descriptive, also threshold
b) ancient woodland	b) descriptive
c) reference sites	c) descriptive
2. Endangered species/their habitats (richness of flora/fauna)	descriptive
3. Genetic diversity	descriptive
4. Nature value trees	threshold values given
5. Dead wood component	**threshold values given**
6. Exotic tree species vs. indigenous	descriptive
7. Species mixture	descriptive, also threshold
8. Natural regeneration	descriptive
9. Special, traditional management systems	descriptive
10. Buffer zones	descriptive, also threshold
11. Prescribed burning	descriptive, also threshold
12. Fragmentation	descriptive

Table 4. Differences in threshold values between selected European countries by certification standards. Example ‘dead wood’ (nature value trees) component.

Country	Certification system	Characteristics
Finland	FFCS	5 stems/ha
Sweden	PEFC	5–10 stems/ha, 2–3 m^3/ha
	FSC	10 stems/ha
Germany	PEFC	descriptive
	FSC	10 stems/ha
United Kingdom	UKWAS	-
	FSC	windfalls 3 m^3/ha
Austria	PEFC	descriptive
Czech	PEFC	descriptive
Estonia	FSC	5 m^3/ha
Poland	FSC	5% of standing volume
Latvia	PEFC	10 trees/ha
Netherlands	FSC	4 trees/ha

Sources for Table 3 and 4: web sites www.pefc.org, and www.fscoax.org.

standards, whereas considerable differences can be observed between countries. This is quite evident when taking into account the various characteristics of forests in different parts of Europe. Genetic diversity, for example, is not a discussion item in the Nordic countries, and therefore not included in the standards. The opposite case can be found in Central European countries.

4. Conclusions

Various driving forces affect biodiversity. These include e.g. climate change, land use and management practices, policy, socio-economics and tourism making it difficult in some cases to quantify each of these factors' share in biodiversity change. The most comprehensive effort on forest biodiversity indicators at pan-European level has been done under the MCPFE and the BEAR concerted action Indicators for Monitoring and Evaluation of Forest Biodiversity in Europe (Larsson 2001). The strength of this set of biodiversity indicators is that they are mostly obtained from national forest inventories, a data set that is available in European countries and updated more or less frequently. The issue of data availability and consistency of collection based on commonly agreed definitions has to be seen as the foundation for monitoring changes over time. Only if such is guaranteed will selected indicators be able to serve their purpose.

Assessment: urban forests

The indicator urban forest index is useful, but holds a number of weaknesses. The percentage of woodland cover can be considerably higher for cities with large tracts of rural land within their jurisdiction or definitions of forest can differ between countries (EC 1997).

The indicator seems useful in the light of growing policy focus on the woodland element of urban green structures. It offers a monitoring tool for afforestation policies. The indicator would become even more communicative once improvement in standardisation between urban areas (e.g. by including only actual urban areas in the assessment) would be in place. Alternative approaches might be to work with fixed radius from the urban area centre (woodlands within 5 km, 10 km, 50 km from the centre).

Assessment: genetic aspects of forest conservation

One of the main challenges is to develop forest policies to minimise the negative effects of, e.g. climatic change on forests to ensure continuous adaptation of tree species in the future (IPGRI 2003). Therefore, national forest strategies, conservation strategies and forest plans will have to play a crucial role for future actions related to conservation issues. Although biodiversity conservation has indeed received considerable attention, most resources have been addressed towards habitat and species conservation and less have gone to activities to set up national programmes on forest genetic resources (Koskela et al. 2003). Efforts will have to be made to improve this situation and investigate the linkage of national programmes on forest genetic resources to National Forest Programmes.

Assessment: forest certification

Forest certification has limited but positive impact on forest biodiversity. In how far forest certification is effective in ensuring the conservation and sustainable use of biological resources is unclear. The effects are highly variable, depending on local circumstances. Threshold values should be based on measurable indicators introduced through research results. In spite of considerable research efforts, biodiversity characteristics are still not very well known. Therefore quantitative threshold values are set allowing consensus decisions.

Concluding remarks

An extensive number of indicators related to forest biodiversity in regional, national, European and global processes have been and are being elaborated. Further investigations looking into other topics that relate to forests and biodiversity are ongoing.

The questions to be asked in this context are (1) in how far can overlapping pressures be identified and responses formulated, (2) how can sometimes conflicting demands and expectations from society towards forests be integrated into biodiversity management and most importantly (3) which most stringent measures need to be taken in order to halt the loss of biodiversity or reduce the rate of loss.

Thomas et al. (2004) have predicted extinction risks for approximately 20% of the earth's surface using projections of species' distributions for future climate scenarios. They have come to the conclusion that under different global warming scenarios by 2050, 15–37% of species in the area under investigation will be committed to extinction. When looking at these estimates it becomes very evident that there is a need for rapid implementation of technologies to decrease greenhouse gas emissions and carbon sequestration strategies. The level of extinction as described by Thomas et al. (2004) raises the issues of prioritising activities within biodiversity indicator development and their practical implementation as well as the set up of effective catalogues of support actions to counteract such developments.

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Acronyms

BEAR	Biodiversity Evaluation Tools for European forests
EEA	European Environment Agency
EFI	European Forest Institute
ETC/NPB	European Topic Centre Nature Protection and Biodiversity
EUFORGEN	European Forest Genetic Resources Programme
FSC	Forest Stewardship Council
MCPFE	Ministerial Conference on the Protection of Forests in Europe
NFP	National Forest Programmes
PEFC	Pan European Forest Certification
TBFRA 2000	Temperate and Boreal Forest Resources Assessment
UNCED	United Nations Conference on Environment and Development

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Forest Biodiversity Indicator: Dead Wood – A Proposed Approach towards Operationalising the MCPFE Indicator

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Abstract

Dead wood is a typical feature and a key factor of biodiversity in the sense of species richness in natural forests. Late development phases are characterised by a high amount and diversity of dead wood. Because European forests have been intensively managed over long periods of time, the late development phases are scarce or completely missing. Such shortage of the dead wood component has led to a severe loss of habitats for saproxylic species. This has been recognised at the political level by the Ministerial Conference on the Protection of Forests in Europe (MCPFE). Within the Pan-European Criteria and Indicators for Sustainable Forest Management, nine improved indicators under the criterion 4: "Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems" have been adopted at the 4th Ministerial Conference in 2003 One of these indicators addresses dead wood. In this paper habitat requirements are identified for saproxylic species groups and their sensitivity towards key criteria of dead wood assessment. The different functional types of dead wood are then defined, categorised and compared to existing assessments within monitoring schemes (national and international). Finally the identified attributes are investigated on their applicability for operational use of monitoring dead wood within monitoring schemes. On this basis, a method of dead wood assessment on the scale of natural and biogeographic regions is suggested.

1. Introduction and aim

In natural forests dead wood is a typical feature and a key factor of biodiversity (Bader et al. 1995; Christensen and Emborg 1996; Sverdrup-Thygeson and Midtgaard 1998; Siitonen

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2001; Humphrey et al. 2002; Komonen 2003). It may generally be defined as above and below ground woody detritus, i.e. dead stems, branches, twigs and roots. However dead wood below ground as well as dead wood attached to living trees is mostly excluded from inventories because these components are difficult to quantify. Dead wood above a certain diameter limit (mostly > 10 cm) is often categorised as Coarse Woody Debris (CWD).

As European forests have been intensively managed over long periods of time, late development phases, characterised by a high amount and diversity of dead wood, are scarcely represented or completely missing. The subsequent shortage of dead wood compared to natural forests (e.g. Harmon et al. 1986; Leibundgut 1995; Korpel 1995; Tabaku and Meyer 1999) has led to a severe loss of habitats for saproxylic species. This has been identified as one of the main reasons for the overall decrease of biodiversity in European forests. A considerable number of saproxylic species are rare and/or threatened. Therefore to increase and sustainably manage dead wood is widely accepted as a key measure to enhance biodiversity in forests (Larsson et al. 2001).

Political processes

This has also been recognised at the political level by the Ministerial Conference on the Protection of Forests in Europe (MCPFE) by stating that “dead wood in the form of dead standing and dead lying trees is a habitat for a wide array of organisms. Due to the lack of dead wood, many of the dependent species are endangered” (MCPFE 2003).

Within the MCPFE process a set of Pan-European Criteria and Indicators for Sustainable Forest Management has been developed. Under Criterion 4 “Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems” nine improved indicators have recently been adopted by the 4th Ministerial Conference in Vienna, Austria in April 28–30, 2003. One of these indicators addresses dead wood (Criterion 4.5: “Volume of standing dead wood and of lying dead wood on forest and other wooded land classified by forest type”).

Monitoring schemes

The European Parliament and Council Regulation concerning monitoring of forests and environmental interactions in the Community, Forest Focus, aims to provide a framework for a Community scheme to contribute towards the protection of forest ecosystems by monitoring the conditions of these ecosystems. The Regulation (EC) No 2152/2003) has been adopted by the European Parliament and the Council of the European Union on the 17th of November 2003. Besides other targets, Forest Focus will also investigate options of monitoring biodiversity, climate change, carbon sequestration and soils.

It is planned to test and evaluate the feasibility of using dead wood for assessing biodiversity for example in the proposed study ForestBIOTA (ForestBIOTA 2003) using 123 intensive monitoring plots of the ICP Forest network (ForestBIOTA 2004).

Data availability

The international forest policy community has repeatedly stressed the need for more and better information on forest resources. The global Forest Resources Assessment carried out by the Food and Agriculture Organization (FAO 2001) is a response to such needs. In particular within the Temperate and Boreal Forest Resources Assessment 2000, including Europe, an attempt has been made to collect data on the volume of dead trees in the standing volume (UNECE/FAO 2000). Not all countries provided information and in most cases no explanations were provided on how these data were collected.

According to this study, the highest values of dead trees in standing volume on forest were found in Switzerland (~11 m^3/ha), Czech Republic (~6.5 m^3/ha), Austria (~6 m^3/ha) and Slovenia (~6 m^3/ha). Relatively low values were reported by Bulgaria and Slovakia (~0.3 m^3/ha), the United Kingdom (~0.4 m^3/ha), Greece (~0.7 m^3/ha) and Spain (~0.9 m^3/ha). Belgium, Cyprus, Germany, Ireland, Italy and Romania did not report data (UNECE/FAO 2000).

The above indicates that dead wood is assessed in many European countries. This is done within national forest inventories, forest reserves research, and through other research activities. However the approaches of measurement can differ between countries. It will therefore be important to investigate options of harmonising field measurements in order to generate comparable data on dead wood. The MCPFE (MCPFE 2003) underlines such needs by stating that "harmonised data on the volume of lying and standing dead wood is so far not yet available in most European countries, but due to the ecological importance of dead wood, it is expected that appropriate data will be collected soon".

Research activities such as the COST Action E4 Forest Reserves Research Network have provided recommendations on methods to be used for the establishment and design of harmonised stand inventories including the collection of dead wood data (Hochbichler et al. 2000). Also the ForestBIOTA initiative is proposing approaches towards its measurement (ForestBIOTA 2003).

Objective

The objective of this paper is to review the role of dead wood as an indicator of forest biodiversity and to examine currently applied monitoring activities. The authors propose an approach for the operational application of dead wood as an indicator of forest biodiversity with reference to the MCPFE indicator 'volume of standing and lying dead wood on forest and other wooded land classified by forest type'.

2. Methodological approach

2.1 Ecological literature review

In order to derive valid and operational indicators of saproxylic biodiversity, a stepwise literature review was carried out. The first step was to review the relevance of dead wood for biodiversity in forests and the impact of forestry. Then the relationship between ecological factors and saproxylic biodiversity was analysed. On this basis indicative variables were derived that are (a) closely related to ecological factors and (b) suitable for the assessment in the frame of forest inventories.

2.2 National forest inventories

Forest inventory activities in European countries were investigated with respect to the methods applied for the collection of data on dead wood. The review is based on the results of the European Forest Information and Communication System (EFICS) report on European National Forest Inventories (EC 1997). The study was implemented in 1996/97 and provides information on national forest inventory methods for 22 European countries. In addition an expert consultation was implemented by means of a questionnaire in order to update or confirm the results from the EFICS study. The questionnaire results were complemented by a literature study.

Furthermore the results from the COST Action E4 Forest Reserves Research Network and the currently running COST Action E27 Protected Forest Areas in Europe were investigated. This was implemented by identifying literature and sending a questionnaire to experts related to forest reserves research. This approach allowed the experts to provide additional input on the issue of dead wood data collection methods and the importance of attributes characterising dead wood as an indicator for biodiversity. The activities of ICP Forest (Fischer 2002) to initiate a biodiversity programme were also taken into consideration.

3. Results

3.1 Ecology of saproxylic species

3.1.1 Contribution to biodiversity

Saproxylic species are defined as species that are “dependent, during some part of their life cycle, upon dead or dying wood of moribund or dead trees (standing or fallen), or upon wood-inhabiting fungi, or upon the presence of other saproxylics” (Speight 1989). Thus this definition includes only obligatory saproxylic species. They represent one of the most diverse species groups in forest ecosystems. In particular dead wood is known to be a critical factor for the following taxa:

- fungi (e.g. Jahn 1990; Möller 1994; Ferris et al. 2000; Edman and Jonsson 2001; Siitonen 2001; Heilmann-Clausen and Christensen 2003);
- epixylic lichens and bryophytes (e.g. Söderström 1988b; Daniels 1991; Esseen et al. 1997; Crites and Dale 1998; Rambo and Muir 1998; Kruys et al. 1999, Kruys and Jonsson 1999; Odor and Standovar 2001; Humphrey et al. 2002);
- arthropods (e.g. Palm 1951, 1959; Brechtel 1991; Westrich 1991; Köhler 2000; Siitonen 2001; Dorow 2002; Alexander 2003; Komonen 2003);
- mammals such as bats and dormice (e.g. Maser and Trappe 1984; Jüdes 1991; Taake 1991; Erickson and West 2003); and
- birds (e.g. Scherzinger 1982; Sandström 1992; Blume 1993; Angelstam and Mikusinski 1994; Bunnel et al. 2002; Utschik 1991; Bursell 2002).

It has been suggested that approximately 20–25% of forest-occurring species depend on decaying wood (Elton 1966; Siitonen 2001, Alexander 2003), thus reflecting the high habitat diversity and ecological complexity of this ecosystem component. However reported numbers differ strongly. This may be due to the fact that knowledge about the ecology of many species groups (e.g. *Diptera*, *Hymenoptera*, *Nematodes*, *Acarina* etc.) is considerably poor.

Fungi are the most important agents of wood decomposition (Swift 1977a; Cooke and Rayner 1984; Rayner and Boddy 1988; Dix 1995; Boddy 2001) and therefore play a functional key role in forests. They in turn serve as habitat for many arthropods (Benick 1952; Paviour-Smith 1960; Jonsell 1999; Jonsell et al. 1999; Thunes et al. 2000; Komonen 2001; Jonsson et al. 2001; Müller et al. 2002; Rukke 2002).

A great number of forest-occurring beetle species are restricted to woody debris. The number varies widely from country to country (e.g. there are approximately 700 forest-occurring beetle species in Norway and England and nearly 1400 in Germany, s. Økland et al. 1996; Kaila et al. 1997, Jonsell 1999; Köhler 2000, Ehnström 2001, Siitonen 2001, Alexander 2003). Renner (1991) estimates the proportion of saproxylic beetles in Central Europe to be about 17–20% of all beetles. With regard only to forest-occurring species, the proportion is much higher; for instance 56% in Germany (Köhler 2000).

3.1.2 Impact of forestry

In general the amount of dead wood in managed forests is much smaller than in natural forests. This is accompanied by a different structure of dead wood supply. In managed forests CWD is mostly represented by stumps and logging waste. In contrast to natural forests, large snags and entire dead trees are rare (Harmon et al. 1986; Albrecht 1991; Rauh 1993; Väisänen et al. 1993; Samuelson et al. 1994; Siitonen 2001). Therefore saproxylic species suffer from quantitative as well as qualitative shortages of habitat in space and time. High proportions of endangered and even regionally extinct species, listed in several Red Data Books, reflect this. For instance in Germany nearly 60% of saproxylic beetle species are listed in Red Data Books (Geiser 1994, 1998). In Britain the percentage is 54% for beetles and 33% for flies (Alexander 2003). The effects of modern forestry on saproxylic biodiversity have been studied in detail by comparing managed and unmanaged forest stands. Significant differences have been reported in particular for cryptogams (Söderström 1988a; Bader et al. 1995; Lindblad 1998; Odor and Standovar 2001; Sippola et al. 2001) and for invertebrates (Schimitschek 1953a,b; Biström and Väisänen 1988; Økland 1994, 2000, 2002; Köhler 1996, 2000; Martikainen et al. 2000; Siitonen et al. 2001; Jonsell and Nordlander 2002; Similä et al. 2003; Sörensen 2002; Sverdrup-Thygeson 2002).

3.1.3 Ecological factors

Climatic conditions
The general relationship of decreasing biotic diversity with increasing latitude and altitude also holds for saproxylic species diversity (Geiser 1994; Berg and Tjernborg 1996; Hammond 1997; Rydin et al. 1997; Köhler 2000; Siitonen 2001; Berg et al. 2002; Brechtel and Kostenbader 2002). This reflects the influence of macroclimatic conditions – in particular temperature and moisture – on the level of biotic diversity (Derksen 1941; Palm 1951, 1959; Schimitschek 1953a; Dajoz 1966; Klausnitzer and Sander 1981; Köhler 2000).

Differences in species composition between open and closed forest sites document the role of microclimate in relation to the habitat requirements of species (Otte 1989a,b; Kaila et al. 1997; Kopf 2000; Martikainen 2001; Snäll and Jonsson 2001; Dorow 2002; Økland 2002; Schlechte 2002; Wermelinger et al. 2002).

Sun-exposure is an important factor for saproxylic species (Sverdrup-Thygeson and Ims 2002). Geiser (1994) assumes that the proportion of saproxylic beetle species with preference for high levels of sun-exposure ranges between 25–50%. However the majority seems to be adapted to shady conditions. Köhler (2000) points out that only approximately 35% of the saproxylic beetles found in Germany are associated with open and half-open woodland. The degree of sun-exposure or shade respectively determines temperature and moisture regime. Saproxylics with preference for cool and moist conditions (e.g. many bryophytes and dipterans) inhabit shaded logs, while species adapted to warm and dry conditions thrive on sun exposed wood structures (e.g. many hymenopterans, buprestids and lichens). Furthermore the spatial distribution of species in different tree strata (roots, stem, crown) can partly be explained by different microclimatic conditions. The percentage of red-listed xerothermophile species is much higher than of those that prefer shaded substrates (Berg et al. 1994; Jonsell et al. 1998; Geiser 1998).

Chemical and physical properties of wood: the effect of decay stage and tree species

It can be assumed that the suitability and attractiveness of dead wood as food source and habitat depends on a variety of different chemical and physical properties, e.g. content of nutrients, repellant and attractant substances, moisture, pH, C/N- and C/P-ratio as well as structural features such as hardness, inner surface and porosity. These factors are predominantly correlated with decay stage, tree species and type of dead wood (stump, snag, whole tree).

In the course of dead wood decay, important properties like bark cover, density, moisture and gaseous regime, contents of nutrients, cellulose and lignin, inner surface and porosity change continuously (Käärik, 1974; Ausmus 1977; Swift 1977a, b; Lambert et al. 1980; Triska and Cromack 1980; Swift and Boddy 1984; Harmon et al. 1986; Rayner and Boddy 1988; Hicks and Harmon 2002; Idol et al. 2001; Schäfer 2002).

As these processes vary in space and time – even within a single dead wood element – the heterogeneity in terms of structural, chemical and physical features increases. However, continued decomposition reverts this development. During the late decay stages dead wood becomes more and more homogeneous and the proportion of ground-dwelling species increases.

As a consequence of these processes the occurrence of species vary markedly in the course of dead wood decay. Distinct communities can be determined for certain stages. Different authors have described the successional pattern of species communities on different types of decaying wood, e.g. concerning cryptogams (Ricek 1967; Runge 1975, 1991; Daniels 1991; Lange 1992; Boddy 2001; Heilmann-Clausen 2001; Schlechte 2002; Heilmann-Clausen and Christensen 2003) and insects (Krogerus 1927; Derksen 1941; Schimitschek 1953a,b, 1954; Dajoz 1966; Pfarr 1990; Simandl 1993; Kleinevoss et al. 1996; Haase et al. 1998; Dorow 2002; Flechtner 2002; Hövemeyer and Schauermann 2003). The results show that in general there are more saproxylic species in intermediate stages of decay and fewer in the early and late stages of decay.

Strict host-specificity at the tree species and even at the genus level is not common. But often a preference can be observed, although this correlation may change within the distribution area (Palm 1959; Elton 1966; Klausnitzer and Sander 1981; Jahn 1990; Bense and Geis 1998; Köhler 2000; Siitonen 2001). Most species only show restriction either to deciduous or to coniferous tree species. A majority of species is associated with deciduous trees. In Germany 67% of saproxylic beetles thrive exclusively on deciduous and only 23% on coniferous trees, while 11% can be found in both groups. Only 8% are restricted to a single tree genus of deciduous trees and only 5% to a single tree genus of coniferous trees (Köhler 2000).

Species-specific characteristics like content of nutrients, wood extractives and volatiles or physical properties are still identifiable in dying and recently dead trees (Cooke and Rayner 1984; Rayner and Boddy 1988; Boddy 2001). Therefore species living in moribund and recently dead trees (e.g. many scolytids and buprestids) show a stronger correlation to a single tree genus than invaders that occur mainly in later successional stages (e.g. lucanids and many elaterids) (Saalas 1917, 1923; Palm 1959; Köhler 2000; Brechtel and Kostenbader 2002). In general similarity of saproxylic communities of different tree species increases throughout succession (Palm 1959; Elton 1966; Köhler 2000; Runge 1975).

Dead wood-type

Communities of saproxylic species differ depending on the dead wood-type, e.g. log, snag, stump, root or attached branches. (e.g. Derksen 1941; Schimitschek 1953a,b, 1954; Brauns 1954; Palm 1959, Pfarr 1990; Runge 1991; Lindblad 1998; Jonsell and Weslien 2003; Lohmus and Lohmus 2001; Sverdrup-Thygeson and Ims 2002).

Snags are important habitats for lichens, hymenopterans and cavity-nesting birds. They provide a vertical gradient of microenvironments (e.g. insulation, temperature, water content, aeration) with corresponding species assemblages. Logs on the forest floor host their own species communities. They are considered the most diverse dead wood-type for saproxylic fungi and epixylic bryophytes (Rydin et al. 1997; Øhlsson et al. 1997; Sippola and Renvall 1999; Lohmus and Lohmus 2001; Humphrey et al. 2002; Heilmann-Clausen and Christensen 2003).

Microclimatic conditions and moisture regime are considered to be the main factors affecting differences in species occurrence in dead wood-types (Øhlson et al. 1997; Sverdrup-Thygeson and Ims 2002).

Size and short-term continuity

Several studies confirm the influence of dimension on species number and community structure at tree level (Palm 1951, 1959; Geiser 1994; Bader et al. 1995; Økland et al. 1996; Haase et al. 1998; Kruys et al. 1999; Ehnström 2001; Lindström 2003). A number of species show a clear preference for woody debris of large diameter, whilst relatively few species are more or less confined to dead wood of small diameter. Nevertheless, fine and small woody debris is important for biodiversity, particularly in managed forests (Kruys and Jonsson 1999; Schiegg 2001).

The increase of species richness with increasing size of dead wood may be related to several factors. On the one hand coarse woody debris simply provides more volume and surface and thus can be host to more individuals and species. Moreover, substrate heterogeneity also increases with dimension. Micro-environmental conditions like moisture, gaseous regime and microclimate are affected by diameter (Schimitschek 1953a,b, 1954; Dajoz 2000; Boddy 2001; Rubinio and McCarthy 2003). Additionally specific small-scale habitats like hollows suitable for e.g. insects, bats or birds are restricted to objects above a certain threshold of diameter and length (Ranius 2002a,b)

Microhabitats

Within a single dead wood object numerous microhabitats can be distinguished, which increase biotic diversity of saproxylic species (Winter et al. 2003). The most prominent structural elements are probably tree hollows and cavities. Although many microhabitats occur only on mature or so called ‘veteran trees’ they are considered in this paper because they play a key role for many saproxylic species.

Tree hollows with wood mould host a large variety of specialized species, e.g. beetles (Palm 1951, 1959; Dajoz 1966; Jonsell et al. 1998; Butler et al. 2002) amongst which *Gnorimus variabilis, Tenebrio opacus, Potosia aeruginoa, Elater ferrugineus* and the Habitat Directive Annex II species *Osmoderma eremita* can be found. Ranius (2002b) showed that the occurrence of several beetle species is correlated to the presence of hollows as well as several tree and stand site characteristics (sun-exposure, height above ground, size of hollow). Also dipterans, mites and pseudoscorpions are frequently found in tree hollows (Ranius 2002b). Some species of bats use cavities in snags as maternity roosts or as summer and/or winter roosting sites. For many birds hollows are required for breeding (Scherzinger 1982; Utschik 1991). Woodpeckers are a keystone-species because they build cavities that can be used by a great variety of post-tenants. This highlights the fact that apart from dead wood, old and still living trees are a crucial factor of forest biodiversity.

Fruitbodies of wood-decaying fungi have their own species rich community (Benick 1952; Komonen 2003). For example, Komonen (2001) has found more than 50 insect species in fruitbodies of *Fomitopsis rosea* and *Amylocystis lapponica* in Finland. In Germany 214 beetle species depend on sporocarps (Köhler 1999). Most insect species colonizing fungi are more or less generalists (Økland 1995; Fossli and Andersen 1998; Jonsell 1999; Guevara et al. 2000a; Jonsell et al. 2001; Komonen 2001, 2003; Jonsell and Nordlander 2002) and the species composition varies with decay stage (Thunes 1994; Thunes and Willassen 1997; Jonsell 1999; Guevara et al. 2000b; Thunes et al. 2000). Monophagous species tend to occur in early successional stages; this is probably caused by specific host chemistry (Jonsell et al. 2001).

Forest and soil type

Many ecological factors already referred to are closely related to forest type, in particular tree species composition and climatic conditions. Soil type is a major factor determining forest type and tree species composition, thereby influencing saproxylic species indirectly. Apart from that it can be assumed that there are further effects on saproxylic species communities (Möller 1991; Grosse-Bruckmann 1994; Hölling 2000; Heilmann-Clausen and Christensen 2003). On the one hand moisture differs between sites according to water storing capacity of soil as well as exposition. Thus atmospheric humidity, quality of substrate and progress of decomposition will be influenced. On the other hand a considerable proportion of humus layer and soil inhabiting species also colonize dead wood in contact with ground especially in late decay-stages (e.g. Derksen 1941; Schimitschek 1953a,b; Köhler 1996). The composition of this species group changes with soil type and subsequently the saproxylic community will differ.

Accessibility, continuity and spatial distribution

Dead wood is often a transitory habitat, which is colonised by several saproxylic species only in certain development stages. The probability of colonisation depends on the spatio-temporal distribution of suitable dead wood objects as well as the dispersal abilities of the saproxylic species.

There are strong indications that dispersal rate and range can be regarded as evolutionary adaptations to primeval forest conditions, in particular concerning the spatio-temporal distribution of dead wood (Schiegg and Pasinelli 1999; Ranius and Hedin 2001; Jonsson 2002).

It has been suggested that many saproxylic species have small dispersal abilities (e.g. Geiser 1994, Ranius and Hedin 2001). The assumption of low dispersal power is confirmed by the spatial distribution of saproxylic species. Many have been identified as indicators of relict ancient forests (Paulus 1980; Garland 1983; Harding and Rose 1986; Alexander 1988, 1998; Harding and Alexander 1993; Ssymank 1994; Bredesen et al. 1997; Sverdrup-Thygeson 2002). This underlines the importance of long-term habitat continuity at the landscape level.

A correlation between the spatio-temporal distribution of dead wood resources, forest history and the occurrence or biodiversity of saproxylic species has been found in many studies (Økland et al. 1996; Kaila et al. 1997; Ohlson et al. 1997; Bakke 1999; Rukke 2000; Ek et al. 2001; Komonen 2001; Kouki et al. 2001; Butler et al. 2002; Jonsell and Nordlander 2002; Jonsson 2002; Sorensen 2002). The results of Schiegg (2000, 2002) underline the meaning of connectivity of dead wood resources in space and time at the stand level. As a consequence of discontinuous availability, local extinctions may take place (Heilmann-Clausen and Christensen 2003). Re-colonisation of newly generated dead wood habitats depends on rate and dispersal abilities of the species.

In contrast Groven et al. (2002) did not find a close correlation between continuous supply of woody debris and the occurrence of wood-inhabiting fungi at stand level. The distribution of fungal species on stand scale seems not to be restricted by dispersal power (Edman and Jonsson 2001).

Disturbance regime

Disturbances like storm, fire or pests are major agents of forest dynamics (Oliver and Larson 1990). Apart from competition and senescence they are the main causes of tree death. Different disturbances create different types of dead wood (e.g. complete or partial uprooting, stem breakage, fire damage) in different environmental conditions. As a consequence saproxylic species assemblages and the pathway of decomposition differ. For example, burnt trees offer various niches (Wikars 1992, 1997; Ehnström 2001). Some species can be considered to be fire-dependent; for example, Siitonen (2001) reported that approximately 30 insect species are fire-dependent in Fennoscandia. In Sweden the fungi *Daldinia loculata* and several beetles like *Melanophila acuminata*, *Stephanopachys substriatus* and *Acmaeops septentrionis* are restricted to burnt wood (Wikars 2001).

3.2 Dead wood assessment in national forest inventories

3.2.1 Attributes

National Forest Inventory (NFI) methods for 22 countries were analysed with regard to the assessment of dead wood (Figure 1). The investigation was based on the "Study European Forest Information and Communication System – reports on forestry inventory and survey systems" (EC 1997). Updating of the information was achieved through literature review and expert consultation.

The attributes measured in these countries are listed in Table 1. According to the results, 16 of 22 countries reported the implementation of dead wood assessments in the course of their forest inventory activities. When taking a closer look at more concise attributes as for example species (group) or decay stages only few countries include those in their NFI monitoring activities. Countries performing very extensive assessments of dead wood include, for example, Austria and Sweden. Several countries have recently added dead wood assessments (e.g. Germany) or are planning to do so for future inventories (e.g. Italy, where the 2nd NFI is due to start in 2004). The review could not clarify what dead wood attributes are assessed in Greece (tree quality for commercial purposes) and Liechtenstein in their NFI, only that dead wood is assessed.

If countries collect data on the same attributes it is not self-evident that this is performed using comparable methods and definitions (Table 2). The available information can vary considerably between countries. Measurements range from detailed inventories to partial or descriptive assessments.

European countries distinguish dead wood in their NFIs into standing and lying dead wood. Standing and lying dead wood has to fulfil certain criteria in order to be measured (minimum height, minimum diameter at breast height, etc.). The minimum *diameter* at breast height of standing dead wood can range from >0 cm (Czech Republic) to >20 cm (Germany). The

Figure 1. Countries analysed with regard to the assessment of dead wood within forest inventory (dark grey).

Table 1. Dead wood information in National/Regional Forest Inventories (EC 1997; modified from Fischer 2002).

	Country												
	AT	BE	CH	CZ	DE	ES	FI	FR	LT	NL	NO	SE	UK
Dead wood information included	X	X	X	X	X	X	X	X	X	X	X	X	X
Components assessed													
Standing	X	X	X	X	X		X	X	X	X	X	X	X
Lying	X	X	X	X	X		X		X	X		X	X
Standing and broken	X		X		X						X		
Stumps	X		X		X		X				X	X	
Woody debris	X		X								X	X	
Abandoned timber	X				X							X	X
Attributes													
Dimension/ quantity	X	X	X	X	X	X	X	X	X	X	X	X	X
Species (group)	X		X		X		X					X	
Decay stage	X				X		X				X	X	
Causal agent of damage			X			X	X	X		X		X	
Other	AT (debris coverage, origin of debris), FI (description of damage, age of damage, damage degree, location of decay), SE (type of damage), CH (type of damage)												

Updated through expert consultation and literature review (Bundeswaldinventur II 2000; Schieler & Hauk 2001; Van Loy et al. 2003; VMI 9 1996).
NFI: AT = Austria (NFI), BE = Belgium (Forest inventory of the Flemish region), CH = Switzerland (NFI), CZ = Czech Republic (NFI), DE = Germany (NFI), ES = Spain (NFI), FI = Finland (NFI and Permanent Field Plots), FR = France (NFI), LT = Lithuania (NFI), NL = Netherlands (Forest Functions Survey), NO = Norway (NFI/Monitoring of Forest Condition), SE = Sweden (NFI), UK = United Kingdom (National Inventory of Woodlands and Trees – Survey of small Woodlands and Countryside Trees)

same or similar thresholds apply for lying dead wood. *Tree species* are described in most countries either by individual species or species groups. *Abandoned timber* is recorded in Germany as forgotten harvesting debris (piled). The debris coverage (twigs, branches) of inventory plots in Austria is estimated by %-classes. Descriptions of decay stages – if considered – are very detailed (Table 2).

The causal agents of the damage, description of damage and cause of death are assessed in some of the investigated countries. Finland defines the *causal agent of damage* as either unknown, wind/snow damage, or as other causes including frost, water and nutrient availability, human activities (harvesting damage) or game damage (elk). In addition a detailed description of insect (e.g. *Tomicus* spp.) or fungi damage (Scots pine blister rust) is performed. The *type of damage* in Portugal is described by seven attributes, including: dead, healthy, sick, or seriously damaged tree. Furthermore the description of *cause of death* can vary from unknown causes to burnt trees and death due to competition. Finland and Sweden describe also the *age of damage.*

3.2.2 Sampling methods (examples)

Belgium (Flanders)

The 1st forest inventory was carried out in 1997–1999 and will be repeated in ten years time.

Inventory design: The inventory system includes about 2665 plots. A forest inventory plot consists of a general description of the stand, combined with measurements of the woody layer. The measurements are applied to living as well as dead standing trees. The woody layer is sampled by a plot design of four concentric circular sample units (A1 to A4) with different radius – R1 = 2.25 m (height <2 m), R2 = 4.5 m (d.b.h. <7 cm and height ≥2 m), R3 = 9 m (d.b.h. 7 cm ≤ d.b.h. <40 cm) and R4 = 18 m (d.b.h. ≥40 cm) – according to the dimension of trees and shrubs (Van Loy et al. 2003).

Dead wood assessment: In circles A1 and A2, all seedlings and shrubs/small trees are sampled. Species and stem number are recorded. Trees over 7 cm d.b.h. are sampled in A3 were the species type is recorded, d.b.h. and height are measured and the individual trees are positioned using polar co-ordinates. For trees over 40 cm d.b.h, the circular plot A4 is sampled, and similar measurements as in A3 are performed. These measurements apply to living as well as dead standing trees. The herb layer is recorded in half of the plots (1331 plots), using a 16 × 16 m square plot. Lying dead wood is recorded within the herbal layer and divided into four diameter classes (2–7 cm, 7–22 cm, 22–40 cm, and >40 cm). Density and stem length of the logs are estimated for the first class and the last two classes (Van Loy et al. 2003).

Finland

The 9th Finnish NFI cycle started in 1996 and is carried out within 5 to 8 years.

Inventory design: The inventory system includes about 70 000 forest and mire plots. The plots are grouped in temporary and permanent clusters (tracts) at approximately 200–300 m distance. Each temporary cluster contains 14–18 sample plots (10–14 sample plots in permanent cluster) (VMI 9 2000). Each plot has a radius of 12.46 m in which measurements on living, standing dead and lying dead trees are performed (VMI 9 1996).

Dead wood assessment: Standing dead wood is assessed if the d.b.h. is higher than 9.5 cm and their height more than 1.3 m. Lying dead wood is measured if the d.b.h. exceeds 9.5 cm (VMI 9 1996).

Table 2. Differences between dead wood attributes in national/regional forest inventories (examples). (For sources and notes, see Table 1).

Standing dead wood
AT almost or already dead; scale (0) none, (1) dead tree insignificant, (2) dead tree significant; **DE** whole tree or parts (height >1.3 m); **ES** d.b.h. over predetermined value that depends on distance between tree and centre of plot; **FI** (height >1.3m); **NL** amount of (still standing) dead trees as percentages of total number of trees present in plot (1) indefinite, (2) no standing dead trees, (3) few dead trees (maximum 5%), (4) moderate number of dead trees (5–30%), (5) many dead trees (>30%); **NO** (1) entire tree, (2) part of tree
Lying dead wood
AT dead lying trees or parts of them; **FI** (0) mostly not lying on ground, (1) mostly lying on ground but dried instead of rotting. (2) mostly lying on ground, (3) lying on ground in several pieces, length or bearing of felling hard to determine; **NO** (1) downed entire dead tree, (2) downed part of dead tree; **CH** (E) lying dead wood, (F) lying tree, still alive; **UK** number of fallen trees recorded by d.b.h. size class
Stumps
AT diameter >20cm, decay stage; **FI** age, height, diameter, how tree was felled; **DE** height < 1.3 m, diameter at stump surface; **NO** (1) stumps present, (2) stumps not present Diameter at breast height (d.b.h.) measurement in standing dead wood **AT**, **NO** (>5 cm); **BE** (>0 cm and height <2 m); **CH** (>12 cm); **CZ** (>0 cm); **DE** (>20 cm); **ES** (>7.5 cm); **FI** (>9.5 cm); **SE** (>10 cm); **UK** (d.b.h. >7 cm)
Diameter at breast height (d.b.h.) measurement in lying dead wood
AT (>20 cm); **BE** (4 classes (2–7 cm, 7–22 cm, 22–40 cm, >40 cm)); **CH** (>12 cm); **CZ** (>0 cm); **DE** (if diameter of fallen dead tree at thick end >20 cm); **FI** (>9.5 cm, length); **NL** (>7 cm); **UK** (>7 cm, 4 classes (7–20 cm, 20–50 cm and >50 cm); **NO** (downed trees without green foliage >10 cm, diameter measured at the midpoint of the part of the stem falling within the sample plot); **SE** middle diameter, length
Definition of decay stage in standing/lying dead wood
AT (*dead lying trees or parts of them, stumps, harvesting debris and/or obviously forgotten logs on sample plot*; categories: (1) timber hard; (2) timber in peripheral parts of stem soft, stem centre hard; (3) timber in peripheral parts of stem hard, stem centre soft; (4) timber decomposed, completely soft; calliper, ocular estimation); **DE** ((1) dead recently (bark still on trunk); (2) starting decomposition (bark loose to missing, wood still sound, in case of heart rot <1/3 of diameter); (3) advanced decomposition (sapwood soft, heart only partly sound, in case of heart rot > 1/3 of diameter); (4) heavily decomposed (wood soft all way through, crumbly if trodden on, contours disintegrated)); **FI** (*standing = severity of decay*, (A) most of bark and branches still remain on tree, (B) coniferous tree that lost bark, most of branches fallen off, deciduous trees not lost bark but the stem starts to rot from inside, (C) stem has become rotten and deciduous tree lost their branches, (D) dry standing trees; of deciduous trees only aspen normally reaches this stage; *lying = severity of decay in the largest part of the fallen tree*, (1) recently fallen, timber still hard, bark remains on stem, (2) timber still rather hard, often covered by bark. Knife penetrates the timber 1–2 cm, (3) rather soft lying tree, bark often lost in large areas. Knife easily penetrates the timber 3–5 cm, (4) timber soft and rotten, knife penetrates the timber completely, (5) timber very rotten and soft, material dispersed when handled); **NO** ((1) recently dead tree (0–3 years), bark intact or loose after attacks by bark beetle; (2) loose bark, wood partly soft; (3) soft outer layers of log; (4) soft, no hard pieces; (5) decomposed, usually overgrown).

Germany
The 2nd German NFI cycle started in 2001 and is near completion.

Inventory design: The inventory records the attributes tree species group, dead wood type, diameter, height/length and stage of decay. Trees are recorded by a plot design in two of four concentric circular sample units – R1 = 1.75 m (d.b.h. <7 cm and height >50 cm), R3 = 10 m (d.b.h. >7 cm).

Dead wood assessment: Dead wood is assessed in R2 with a radius of 5 m. Lying dead wood is included in the measurement if the butt end of the stem is located in the sample plot and is exceeding a minimum diameter of 20 cm. Standing dead wood is measured if the d.b.h. is >20 cm. Stumps are included if they have a minimum height of 50 cm or 60 cm diameter at stump surface. Not regarded as dead wood are harvesting debris, recently felled trees, recently died trees with branches and dead wood parts on healthy trees (Bundeswaldinventur II 2000).

Spain
The Spanish NFI is implemented in 10-year cycles. At present the 3rd NFI is ongoing. The measurements are applied to the same permanent plots in every cycle. Work units are the provinces (administrative divisions). This systematic design was applied in 2nd NFI (1986–1996) with the first re-measurement of the plots in the 3rd NFI during 1997–2006 (Gonzalez 2003).

Inventory design: Trees are sampled by a plot design of four concentric circular sample units with variable radius – R1 = 5 m (d.b.h. >7.5 cm), R2 = 10 m (d.b.h. >12.5 cm), R3 = 15 m (d.b.h. >22.5 cm) and R4 = 25 m (d.b.h. >42.5 cm).

Dead wood assessment: Standing dead wood that fulfil certain requirements are assessed using the same approach as living trees (d.b.h., height and causal agent of damage). The requirements are that the d.b.h. must exceed a predefined value depending on the distance of the tree to the centre of the plot. No stages of decay are assessed (Bravo et al. 2002; Gonzalez 2003).

Switzerland
The 2nd Swiss NFI was carried out during the period of 1993–1995.

Inventory design: Within the NFI data have been collected on about 6400 field plots. Trees are sampled by a plot design of two concentric circular sample units – R1 = 7.98 m (d.b.h. >12 cm), R2 = 12.62 m (d.b.h. >36 cm).

Dead wood assessment: For every tree – no matter if they are dead or living or lying or standing – base data are collected including besides others species identification, position, d.b.h. and the distance from the plot centre. The height is calculated via a model built from a sub-sample of trees for which the height is measured. The length of the dead wood pieces is not measured. Qualitative data on the sample trees including decay stages and dominance of moss on the surface of dead trees will be collected in upcoming 3rd inventory. Such data were so far not assessed in Switzerland (Brassel and Lischke 2001; Böhl 2003).

3.3 Dead wood assessment proposals resulting from other initiatives

COST Action E4 – Forest Reserves Research Network makes recommendations on methods for the establishment and design of stand inventories in forest reserves research based on permanent sample plots supplemented by more extensive permanent ‘core areas’ (Hochbichler et al. 2000). Hochbichler et al. presented an approach for the assessment of dead wood in strict forest reserves (Table 3).

Table 3. Dead wood attributes to be measured in strict forest reserves. Data collection in sample plots and core areas (Hochbichler et al. 2000).

Type of dead wood and diameter threshold	Attribute
Standing dead wood (d.b.h. =5 cm)	species d.b.h. height stage of decay x,y (location in sample plot)* *x,y (location in core area)*
Lying dead wood (d.b.h =10 cm)	species component (whole tree/stump/stem/branch) diameter length stage of decay *x,y (location in core area)*

* Note: if using the data for modelling forest dynamics, these data are required. Italic attributes only apply in core areas

The project proposal Forest Biodiversity Test-phase Assessments (ForestBIOTA) under Regulation (EC) No 2152/2003 (Forest Focus) for the development of forest biodiversity monitoring (Art. 6(2) monitoring test phase) aims at the contribution to operationalising dead wood assessment and monitoring. It targets the investigation of the potential for embedding such assessments within the framework of the existing Intensive Monitoring sites established under ICPForest (ForestBIOTA 2003). One of the ForestBIOTA working groups is engaged in preparing a dead wood monitoring protocol.

3.4 Expert consultation

The following results are based on expert consultation. Questionnaires were sent to forest inventory experts and experts in the field of forest reserves research and nature conservation. The experts were asked to provide information regarding a potential set of attributes for dead wood as an indicator of forest biodiversity. A total number of 14 questionnaires were distributed, 10 of which were returned (a reply rate of about 70%). The questionnaire was divided into 4 sections: (1) dead wood definitions; (2) standing dead wood; (3) lying dead wood; and (4) a general assessment of dead wood as a biodiversity indicator. The questionnaires served to confirm and update information found from the literature review, and to provide expert input on operational approaches and attributes for applying dead wood as a biodiversity indicator.

Dead wood definition

In the introduction to this paper a general definition of dead wood was presented. When going into more details of definitions and related terminology, these may vary between countries. More details on terminologies have been collected in a database, which is accessible at http://www.efi.fi/projects/coste27/Databases.html. A few examples of definitions for dead wood provided by the consulted experts are given below:

1. Material over a certain size (threshold >10 cm diameter) that is not more living and that is left in the forest.
2. Any woody material from forest trees and shrubs no longer living, but that has been living in the respective stand.
3. All above and below ground woody detritus in forests comprising stems, twigs and roots excluding dead wood parts on living trees. In general below ground dead wood is excluded from terrestrial assessments.
4. The sum of all woody necromass in a forest stand, original from the stand and originating from natural processes of mortality and decay (calamities like windthrow, natural decline of old or suppressed trees) or human interventions (left crown wood, stumps, girdled trees, etc.).

A problem of dead wood assessment in the field that has not been addressed by the experts is the distinction between moribund (e.g. trees with only one living branch) and dead trees.

Attributes:

1. Dead wood type: The experts agreed on four classes: (1) whole standing tree; (2) snags; (3) stumps; and (4) lying dead wood. In addition more detailed attributes could be assessed for defining the dead wood type or the cause of tree death: (5) log with root plate (from windthrow); (6) dead branch/crown hanging; (7) dead branch/crown fallen.

2. Tree species: The majority of the experts agreed that data on the species should be collected. The main reasons are that habitat qualities (breeding and food habitat) and the progress and type of decay are highly dependent on the tree species. The minimum level of defining the species should be to distinguish between conifers/broadleaves and/or hardwoods/softwoods.

3. Dimension: Tree height and diameter are the two main attributes to quantify the dimension of standing dead wood. Experts stressed the need to add qualitative aspects to the measurements as many dead stems will not be complete cylinders or may be partially hollow. This is necessary to allow adapted volume calculations. Experts are in agreement that diameter should be measured at breast height (d.b.h.). Opinions vary as to whether the measurement should be done with girth tapes or if the stems should simply be placed into broad diameter class categories by ocular estimation with a given lower threshold. The latter is seen as the most effective approach to apply within forest inventory assessments. The most simple and cost effective method for measurement of lying dead wood is an ocular estimation of the volume/ha. More accurate but time consuming approaches are to measure tree length and diameter. It was expressed that the diameter should be an ocular estimate (classification by diameter classes) or measured with calipers. A minimum diameter for registration should be given for both methods.

4. Cause of damage/death: Some experts stressed that the cause of damage might be difficult to define as the results are very subjective and will not allow collection of reliable data. Mortality can be a combination of several factors – e.g. windbreak after fungi-attack to a suppressed tree. Other contacted experts stressed the importance of this attribute. The cause of damage could in the simplest case be divided into natural or anthropogenic damages or combination of both. For this approach an ocular estimation would be sufficient.

5. Cavities/hollows: This attribute was seen as problematic to assess, as the probability to identify small cavities or cavities in the crown is rather small. Such investigations are to be implemented through in-depth studies or specific/high intensity assessments, but outside forest inventory activities. In NFIs a simple ocular estimation method could be applied (cavities present: Yes/No) or could be derived indirectly based on tree dimensions, decay stage and dead wood type.

6. Amount of bark left: The experts proposed numerous methods to assess the amount of bark. Some stressed that the attribute could be included in a description of decay classes (see below). All experts agreed that an ocular estimation should be used to describe the amount of bark left.

7. Decay stage: A good and easy in-the-field subdivision has been suggested by Christensen and Hahn (2003, Table 4). It combines different attributes in six decay classes taking into account bark cover, presence of twigs and branches, wood hardiness, surface and shape. Other approaches as used within NFIs were listed. All experts stressed the need for a harmonised classification scheme for deriving decay stages. The scheme should be transparent, sufficient and easy to apply in the field.

4. Comparative analysis

As shown in the ecological literature review, there are numerous factors that influence saproxylic biodiversity (Figure 2). They are both primary factors (i.e. abiotic environmental influences) and conditioning factors (i.e. dead wood characteristics like dimension or decay state). Further an investigation of current dead wood assessments within NFIs and an expert consultation has been performed.

Based on these sources we will describe a set of valid attributes and indicators of saproxylic biodiversity suitable to be applied within forest inventories. We distinguish between attributes, i.e. parameters measured within an inventory (e.g. d.b.h.), and indicators, i.e. statistics used to give valid information about saproxylic biodiversity (e.g. average amount of dead wood per forest-type in a certain biogeographic region).

Derivative attributes and indicators

The influence of ecological factors on saproxylic biodiversity is considered a main criterion for deriving recommended attributes, which in turn serve as basis for indicators of saproxylic biodiversity. Different attributes suitable for the assessment show different degrees of sensitivity to ecological factors (Table 5). Also the interdependence between attributes was evaluated in order to decrease redundancy. For instance presence of epiphytes and microhabitats depends mainly on forest- and dead wood-type as well as decay stage and the site influence is highly correlated to certain biogeographic/natural regions and forest-types.

The results of this approach yielded seven attributes to be of crucial importance for indicating saproxylic biodiversity (Table 5). Three of the attributes (biogeographic/natural region, forest type and forest history) have been listed by Larsson et al. (2001) as structural, compositional or functional key factors of biological diversity in European forests.

The biogeographic region can easily be determined. The stratification of information into forest-types is widely agreed to be a prerequisite of a valid indication system of biodiversity (Larsson et al. 2001; MCPFE 2003). The Forest Types for Biodiversity Assessment suggested by Larsson et al. (2001) have been designed particularly to meet the needs of biodiversity assessment. However, it is more difficult to cross-reference to other classification schemes and allow for Europe-wide comparisons. The EUNIS Habitat classification (Davies and Moss 2002) may be better suited as it is explicitly designed to correspond to other major habitat systems in Europe, e.g. the EU Habitat Directive and the Corine Land Cover Classification.

To reveal in-depth information on forest history, studies of historical maps and records are inevitable. The proper application of such information, however, will require further efforts to harmonise definitions, and in particular that of ‘ancient forests’.

Table 4. Example for the description of decay classes for *Fagus sylvatica* (Christensen and Hahn 2003).

Decay class	Bark	Twigs and branches	Softness	Surface	Shape
I	intact or missing only in small patches, >50%	present	present hard or knife penetrate 1–2 mm	covered by bark, outline intact	circle
II	missing or <50% only branches	only branches >3 cm present	hard or knife penetrate less than 1 cm	smooth, outline intact circle	circle
III	missing	missing	(III) begin to be soft, knife penetrate 1–5 cm	smooth or crevices present, outline intact	circle or elliptic
IV	missing	missing	soft, knife penetrate more than 5 cm	large crevices, small pieces missing, outline intact	flat elliptic
V	missing	missing	soft, knife penetrate more than 5 cm	large pieces missing, outline partly deformed	flat elliptic
VI	missing	missing	soft, partly reduced to mould, only core of wood	outline hard to define	flat elliptic – covered by soil

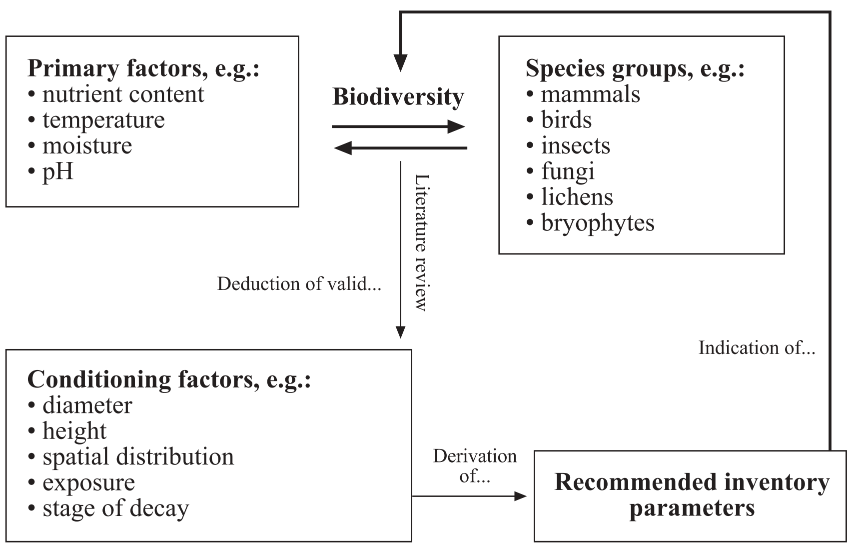

Figure 2. Approach for deriving valid indicators of saproxylic biodiversity.

Table 5. Sensitivity of inventory attributes in relation to ecological factors, which determine saproxylic biodiversity. Recommended attributes are indicated by bold type. Abbreviations: N = numerical, C = categorical attribute, white = no sensitivity,– = low, ± = medium, + = high sensitivity.

Aggregated Ecological Factor	*Scale*	**Attribute** **Dimension (diameter, length, height)**	**Tree species**	**Decay-stage**	**Dead wood-type**	Presence of Epiphytes/Fruitbodies of Fungi	Presence of cavities/hollows	Spatial distribution	**Biogeographic region/Natural region**	**Forest type**	Site	**Forest history (ancient forest)**	Present fragmentation of forests
		N	C	C	C	C	C	N/C	C	C	C	C	N
Microclimatic conditions	Object	+	±	±	+				+	±	+		
Chemical wood properties	Object		+	+						+			
Physical wood properties	Object		+	+	±					+			
Structural wood properties	Object		+	+						+			
Short-term continuity	Object	±											
Heterogeneity	Object	+		+		+	–			+			
Microhabitats	Object/Stand	+	±	+	+	+	+						
Accessibility	Stand/Landscape							+					+
Climatic conditions	Stand/Landscape								+	+	+		±
Continuous existence of forest	Stand/Landscape											+	

Table 6. Example of a table to indicate state and change in dead wood amount per biogeographic and natural region
Biogeographic region: Atlantic
Natural region: Lüneburger Heide

Forest-type	Amount and change of dead wood per decay-stage							
	I		II		III		IV	
	m^3/ha	**Δ%**[1]	m^3/ha	**Δ%**	m^3/ha	**Δ%**	m^3/ha	**Δ%**
Mixed oak forests								
Ash wood								
Lowland beech								
Atlantic dune forests								
Pine plantations								
Spruce plantations								

[1] = net-difference to last inventory

Additional attributes to the above are dimension (diameter, length), tree species, decay-stage and dead wood-type.

The attributes *tree height/length and d.b.h.* are essential for calculating dead wood amount (m^3/ha). In order to increase efficiency, classed estimates could be favoured instead of measurements. It is crucial to agree on a minimum diameter for assessment.

The identification of the *tree species* or at a minimum distinguishing between conifers and broadleaves is crucial as habitat qualities and the progress and type of decay differ strongly.

Instead of direct estimation of the amount of *bark* left the *decay stages* should suffice. Decay stage and dead wood type are considered two crucial attributes concerning saproxylic biodiversity. Suggestions for assessing the decay stage have been presented by Albrecht (1991) for a general key, Christensen and Hahn (2003) and Müller-Using and Bartsch (2003) for European beech (*Fagus sylvatica*), Maser et al. (1984) for Douglas-fir (*Pseudotsuga menziesii*) and Stöcker (1998, 1999) for Norway spruce (*Picea abies*) and Scots pine (*Pinus sylvestris*). Meyer (1999) suggested a classification scheme of dead wood-types that is being successfully applied in monitoring of unmanaged strict forest reserves in Germany. At least whole standing trees, snags, stumps and lying dead wood should be distinguished.

Ståhl et al. (2001) and Meyer et al. (2003) have summarised appropriate methods to assess the described attributes in the field. Nevertheless there is an urgent need to harmonise threshold values (in particular minimum diameter), decay-stages and the classification of dead wood-types. Keys need to be transparent, sufficient and easy to apply in the field. The attributes identified in this review match well with those of the COST Action E4 and the proposal of ForestBIOTA.

Concerning indicators, the ecological literature review revealed that amount of dead wood and biodiversity are closely correlated (e.g. Økland et al. 1996; Humphrey et al. 2000; Siitonen 2001), suggesting that the amount and share of dead wood could be used as main indicators of biodiversity. This is in line with the MCPFE indicator 4.5 'Dead wood' that requires the calculation of 'volume of standing dead wood and of lying dead wood on forest and other wooded land classified by forest type' (MCPFE 2003). However, the ecological literature review also revealed that amount and share of dead wood can only be regarded as a valid indicator if it is broken up by the following stratification variables: biogeographic/natural region, forest type, ancient/secondary forest, species or species group, decay-stage, dead wood-type and dimensions. We designed examples of how to present indicators of saproxylic biodiversity (Table 6).

Table 7. Example of a table to indicate accessibility and spatial distribution of dead wood per biogeographic and natural region.
Biogeographic region: Atlantic
Natural region: Lüneburger Heide

Dead wood share	Frequency of plots/stands **%**[1]	**D%**[2]
<1%		
1–3%		
3–5%		
5–10%		
10–20%		
>20%		

[1] = state at present inventory; [2] = net-difference to last inventory

Concerning spatial resolution, the lowest level is the natural region. Statistics can easily be scaled-up to biogeographic regions. Information at the stand level is out of the scope of large-scale inventories. If m^3/ha is used as measurement unit it might become problematic to ensure comparability between biogeographic regions and forest types. The relative measure 'share of dead wood' may be more suitable for comparisons.

Dead wood within a landscape is not necessarily evenly distributed. It can be highly aggregated thus creating very different habitat conditions for saproxylic organisms than if it were distributed evenly over larger areas. The spatial distribution can be accounted for by using frequency data (Table 7). Irregular and aggregated spatial distribution is indicated by low frequency values of plots or stands with a medium to high amount of dead wood.

The co-existence of different size classes and decay types is another important criteria for the assessment of saproxylic biodiversity. This could be an indicator derived from measurements (e.g. co-existence of more than three dead wood size classes: Yes/No).

Dead wood assessments in national forest inventories

At the national level monitoring dead wood seems most efficient if integrated into already established monitoring activities such as NFIs. It has been shown that in many European countries collection of data on dead wood, both standing and lying, is incorporated into their NFI assessments. Many of the investigated European countries assess the dead wood type, dimension by tree species or tree species groups and decay stages. The collection processes may deviate from one another in terms of amount of detail, methods and thresholds (e.g. minimum diameter >0 to ≥20 cm). This coincides with the MCPFE Advisory Group Recommendations for Improved Pan-European Indicators for Sustainable Forest Management, which state that 10 cm diameter (and 1 m length/height) is an appropriate threshold for standing and lying dead wood (MCPFE 2002).

Many of the consulted experts proposed a minimum diameter of dead wood at ≥10cm. Such a minimum diameter would allow normal logging residues and 'natural' dead wood to be better distinguished. The European guidelines of COST E4, recommended a minimum diameter of 5 cm for standing and 10 cm for lying dead wood in forest structure in strict forest reserves (Hochbichler et al. 2000). It is advisable to agree on measurement without bark.

The sampling methods for collecting dead wood data presented in this paper are shown to be comparable in their approach. There are differences in basic plot design, number and size of sampling units and the measurement thresholds for dead wood. It should be considered in this context that small permanent sample plots may not always be very efficient to gain the desired information on dead wood with acceptable accuracy. Assessment on transects or line intersect sampling combined with strip surveys may be more appropriate (Ståhl et al. 2001). To address such questions will be the task of the recently established European Network of Forest Inventories (ENFIN). It could engage also in harmonisation issues (definitions and thresholds) on dead wood and investigate approaches that allow compatibility of results for international data reporting.

5. Making the MCPFE indicator dead wood operational

In most cases assessments of dead wood refer to forest departments or even larger scales above the stand level, i.e. landscapes. The information is required for forest management planning, silvicultural treatment of stands and for monitoring changes.

The requirements of the MCPFE indicator ‘dead wood’ reflect this goal. It is defined as the volume of standing and lying dead wood, classified by forest type on forest and other wooded land (MCPFE 2003). The measurement unit proposed for state descriptions is m^3/ha and for change m^3/ha/year with a periodicity of 10-year intervals (MCPFE 2002). Separate figures should be reported on the different stratification attributes described above (biogeographic/natural region, forest type, ancient/secondary forest, species or species group, decay-stage, dead wood-type and dimension), both on forest land and other wooded land. This calls for a common course of action including the development of harmonised collection guidelines in order to guarantee the availability of comparable data (Table 8).

NFI systems, based on grids of individual plots distributed systematically are not designed to give estimates for individual forest stands. A stand level estimation would only become possible by e.g. adding plots.

A first fundamental task will be to build clear consensus on the use of terminology as it represents the foundation for collecting comparable data. The terms and definitions that have been collected during the elaboration of this paper have been made available through an on-line database at http://www.efi.fi/databases/coste27/terms_deadwood/deadwood.php.

In order to achieve consensus on definitions, methods, thresholds and collection guidelines, all the main players involved in dead wood assessment will need to be brought together to agree on elaboration of a common approach for making dead wood an operational indicator at European level. A crucial role will fall to the ENFIN Network and specialist groups on dead wood types (Table 8).

Site factors highly correlate to a biogeographic region, natural region and forest-type. The authors consider the spatial scale of natural regions as an appropriate means for determining the amount of dead wood. Natural regions can easily be determined when plot inventory data is assessed. The collected information within a natural region can then be scaled-up to biogeographic regions. If absolute measurement units are used (m^3/ha) the comparability between biogeographic regions and forest types may become difficult. One solution may be to use relative measures as for example the share of dead wood in order to guarantee comparability.

If the approach of forest type reporting is pursued, comprehensive check lists of forest types will need to be elaborated in order to facilitate cross referencing and relationships to other classification systems, e.g. used within NFIs or other forest resources assessment activities. The reporting of dead wood by forest types will call for close cooperation with the appropriate experts. Ongoing initiatives are looking at options of developing a simpler

Table 8. Proposed attributes for measurement, foreseen actions and potential contributors for operationalising the MCPFE indicator of dead wood.

Attribute	Assessment	Level	Need for action	Possible contributors	MCPFE reference
dimension	measurement or classed estimates of diameter, height, length	Object	minimum values; modes of measurement; classes for estimates	ENFIN	diameter: 10 cm height, length: 1 m
tree species; species groups	determination / classification acc. to key	Object	classification key if species groups shall be employed	expert group	
dead wood type	classification acc. to key	Object	classification key	expert group	
decay stage	classification acc. to key	Object	classification key	expert group	
forest type	classification acc. to key	Plot/ Stand	cross-referencing to other classifications needed; operational determination in the field required	expert group	report by forest type (in m^3/ha)
forest history	definition indoor	Plot/ Stand	review of literature and historical maps	project activity	
biogeographic region/natural region	definition indoor	Plot/ Stand	both easy to determine;	expert group	

classification that is more appropriate for monitoring forest biodiversity. They would consist of 14 European forest types by combining some of the BEAR FTBAs (Møller and Bradshaw 2004.).

Further the inclusion of information on forest history and structural dead wood diversity will need more attention as it may reveal important information on the importance of dead wood stands and associated saproxylic communities.

One should also not overlook the costs associated with collecting dead wood information. Additional costs will need to be acceptable in particular if a country may not have a suitable inventory for dead wood already in place. The application of efficient sampling techniques and measurements (e.g. dimensions would not be measured, but substituted by classed estimates) would allow additional costs to be kept low. Similar considerations will need to be made for the description of tree-species. Broader species groups or the distinction between softwood and hardwood as well as broadleaved and coniferous species could be appropriate as it is often quite difficult to distinguish tree species in late decay stages.

For example, a consulted expert stated that measuring the dead wood component in Finland may increase the inventory cost by up to 10% as compared to the normal stand measurements.

It should be kept in mind that dead wood is only one indicator of forest biodiversity. The indicator dead wood and its importance should be put into relation to eminent threats to biodiversity that may result e.g. from predicted climatic changes over the next decades

(Thomas et al. 2004). Also the potential use of dead wood data in other fields of research should be investigated (e.g. carbon sequestration). This implies that collection methods and data serve both various research groups and monitoring requirements under different processes and conventions.

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Biodiversity Indicators for UK Managed Forests: Development and Implementation at Different Spatial Scales

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Abstract

This paper reviews research on identifying indicators of biodiversity in UK managed forests for use by policy makers, forest managers and strategic planners. The importance of linking different spatial scales and developing an integrated set of indicators across scales is emphasized, as the majority of woodland species respond to both stand (e.g. structural characteristics such as deadwood) and landscape (e.g. spatial configuration of habitat patches) factors. In many cases, species responses are governed by a complex of biotic and abiotic factors operating at these different scales, which makes it difficult to provide simple sets of indicators for implementation by end-users. Various approaches to the problem of implementation are described, with special reference to the development of guidance on the management and measurement of deadwood, and the assessment of landscape quality using focal species. Suggestions are also made for improving the current national-scale set of indicators for monitoring biodiversity as part of sustainable forestry requirements.

Keywords: woodland area, habitat quality, landscape pattern, habitat networks, deadwood.

1. Introduction

Forest cover in the UK has increased from 3.5–11.9% of total land area over the last hundred years (Forestry Commission 2002). Expansion has been achieved largely by afforestation of open ground by exotic conifer species such as Sitka spruce (*Picea sitchensis*). Many remnant semi-natural woods were also converted to plantations, especially after 1930 (Thompson et al. 2003). In the 1990s changes in government policies led to the implementation of sustainable forest management principles including the maintenance and enhancement of biodiversity in all

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forest types (Forestry Commission 1998). At the beginning of the 21st century, biodiversity conservation remains high on the public agenda (Forestry Commission 2003a) as does the need to deliver improved social and community benefits from forestry. However, there are also a range of other drivers affecting UK forestry. The continuing economic down turn in timber prices has led to a re-appraisal of commercial forestry especially in marginal areas, and there is a prospect that large tracts of spruce forest may be converted to other land-uses such as naturalised mixed woodland or open ground (Humphrey 2004).

Agri-environment and forestry incentives are increasingly targeted towards agricultural diversification and biodiversity enhancement and restoration at the landscape scale (Forestry Commission 2003b). As these changes take place in tandem with climate change (Broadmeadow 2002) the picture emerges of dynamic forest landscapes where biodiversity values are likely to evolve considerably in subsequent decades.

Biodiversity is a multi-scaled concept, traditionally considered at a hierarchy of spatial scales, ranging from the genetic, through species, to habitat, landscape and regional/national scales (Larsson 2001; Noss 1990). However, this categorisation can be unhelpful as it ignores the effect of ecological processes operating across scales (Hansson 2001). An increasing number of studies have shown that species responses are multi-factorial (Humphrey et al. 2004; Verheyen et al. 2003), where diversity and distributions are influenced concurrently by large-scale factors such as landscape pattern and configuration, and habitat scale factors such as site quality (e.g. occurrence of particular stand structures), and land-use history (Brotons et al. 2003; Honnay et al. 1999a). Responses to these factors are species-specific leading to differential effects of changes in habitat availability, quality and configuration on species distribution and diversity (Fahrig 1997; Honnay et al. 1999a).

However, complete assessments of biodiversity are not practically achievable, hence the search for reliable indicators or short-cut measures of biodiversity (Ferris and Humphrey 1999; Jonsson and Jonsell 1999; Noss 1999; Simberloff 1998). The principle behind the indicator concept is that the characteristics of an easily measured feature such as an organism or aspect of forest structure can be used as an index of attributes (e.g. diversity) that are too difficult or expensive to measure for other species and communities (Williams and Gaston 1998; Landres et al. 1988). Larsson (2001) has proposed a broad typology of indicators for assessment of biodiversity in European forests at a range of spatial scale. Indicators are classed either as structural (e.g. presence of deadwood; landscape pattern), compositional (e.g. species) or functional (e.g. rate of species dispersal, or successional processes). Structural and compositional indicators are considered to be more tractable for end-users (Angelstam et al. 2001).

A variety of indicator systems have been developed both at the EU level (MCPFE 2002) and within individual countries (e.g. Ministry of Agriculture and Fisheries 1994; Eeronheimo et al. 1997). The nature of the indicators depends on who is using them and for what purpose. Policy makers require broad- headline indicators which illustrate trends at regional or national scales (e.g. DEFRA 1999) whereas forest managers are more interested in indicators which measure "outcomes" or the results of management activity at stand or compartment scales (1–100 ha) (Ferris and Humphrey 1999; Kremsater et al. 2003). At intermediate spatial scales such as the landscape or forest management unit (Angelstam et al. 2001), strategic planners need robust measures which can both predict and monitor the effects of landscape change on biodiversity. At all levels, it is important to link indicators to goal setting (Lindenmayer et al. 2002; Noss 1999) and to define in what way indicator values are expected to change over time either in response to policy implementation or management practice (Failing and Gregory 2003).

In the UK, individual countries are currently developing sets of biodiversity indicators for monitoring the outcomes of both management activities and policy implementation (e.g. Scottish Biodiversity Group 2003b). Indicators of forest biodiversity are included as a subset of these wider sets of indicators and also as a subset of sustainable forestry indicators for the

UK as a whole (Forestry Commission 2002). This paper: 1) reviews current developments in forest-related biodiversity indicators in the UK; 2) examines the scientific rationale behind some of the measures that have been adopted and how these might be refined and developed through further research; 3) Proposes additional indicators which might be useful for strategic planners and forest managers involved in land-use change.

The paper is not intended to be a statement of government policy on biodiversity indicators either at the UK or individual country level.

2. Woodland area indicators

The existing UK national indicators for forest biodiversity are described in Forestry Commission (2002a) and listed in Table 1. The first two measures are primarily concerned with habitat area change based on remote-sensed and mapped based datasets such as the National Inventory of Woodland and Trees (NIWT – Forestry Commission 2002b). Ancient woodlands (*sensu* Peterken 1993) are considered to have higher value for woodland biodiversity compared to more recent woodlands, but the broader term of native woodland is also included as the strict definition of ancientness can be less relevant to conservation value in some parts of the UK (Roberts et al. 1992). It is axiomatic that the ancient woodland indicator is primarily concerned with monitoring habitat loss as no "new" ancient woodland is theoretically possible. However, the trend in native woodland area can, in theory, be positive or negative. The total ancient semi-natural (i.e. native woods which are also ancient) woodland area in the UK is estimated as 288 000 or 10.4% of total woodland area.

There are numerous examples world-wide where remote-sensed data have been used to estimate changes in broad forest habitat types over large areas and hence to imply changes in biodiversity (e.g. Brotherton 1983; Cushman and Wallin 2000). However, focusing simply on broad area changes in ancient/native woodland ignores changes in habitat classes within these categories. For example, loss of highly valued old pinewood could be balanced by gain of less valuable new birchwood. The indicator suggests no change in biodiversity, but in reality the change would be negative in the medium term. A potential solution may be to use the broad and priority habitat classes set out in the UK Biodiversity Action Plan (UKBAP – UK Biodiversity Group 1995). Work is continuing to develop accurate figures on changes in the area and condition of priority woodland habitats as part of the UKBAP reporting, and these figures could be used as national biodiversity indicators. However, there remains no system to monitor actual losses of ancient woodlands, native woodland or individual UKBAP priority woodland types.

3. Woodland condition/quality indicators

Indicators of woodland condition are essentially structural in nature and include key components which relate to ecosystem health and habitat quality. Although "diversity of woodland within a stand" is included as a separate indictor in Table 1, in reality it is a sub-set of the woodland condition indicator being concerned principally with measures of habitat quality. Habitat quality is an important determinant of species occurrence and persistence (e.g. Honnay et al. 1999b) although landscape structure, patch area and connectivity are also important (see Fahrig 1997 and later discussion).

The current condition indicator (Table 1) focuses primarily on the condition of woodland on Sites of Special Scientific Interest (SSSIs). SSSIs are judged to have the highest value for

Table 1. Detail of UK national-scale forest biodiversity indicators – adapted from Forestry Commission (2002a). NIWT = National Inventory of Woodland and Trees (Forestry Commission 2002b).

Indicator	Description	Assessment of trend	Assumed relationship with biodiversity
Ancient woodland	Areas continually wooded (can be semi-natural or planted) since 1600 in England and Wales; 1750 in Scotland; 1830 in Northern Ireland	Possibility of ancient woodland inventory revision (Roberts et al. 1992; Spencer and Kirby 1992)	Assumed that ancient woods have higher biodiversity value than more recent woodland
Native woodland	Woodland where at least 50% of canopy cover is site-native species; can be semi-natural or planted	Baseline area surveys are planned in Scotland, only estimates of area are available for the other countries	Woods comprising native trees species are thought to have higher biodiversity value than woods of non-native species
Native woodland condition	Identification of “features of interest” which are related to habitat quality	Condition of woodland SSSIs in the UK is assessed every 6 years; this could be extended to UK BAP priority woodland types through NIWT	Assumed that habitat quality is a principal determinant of biodiversity value
Diversity of woodland within a stand	Subset of woodland condition currently focusing on deadwood	NIWT provides information on deadwood and stand structure	As for woodland condition
Abundance of fauna	Restricted at present to the Woodland Bird Index	Breeding Bird Survey and Woodland Bird Survey (RSPB 2002)	Assumed that the Woodland Bird Index co-varies with changes in diversity of a wide range of other species groups
Richness of flora	Species-richness of woodland flora	Further developments of Countryside Survey 2000 (Sheail and Bunce 2003) and inclusion of NIWT data	Assumed that trends in species richness reflect changes in other species groups
Natural regeneration of woodland	A functional indicator referring to way woods are regenerated or new wood established	Woodland Grant Scheme applications for non-state (Forestry Commission) woodland creation (Forestry Commission 2003b)	Assumed that woodland established and maintained by natural regeneration as opposed to planting will support higher species diversity

Table 2. Standing and lying deadwood in high forest in Great Britain, adapted from Forestry Commission (2002a). Figures are percent high forest area (i.e. excluding coppice and wood pasture/ parkland).

No of pieces per ha	England	Scotland	Wales	GB
Fallen deadwood				
0	84.5	88.6	98.2	88.0
4–8	8.6	5.2	0.7	6.1
12–16	3.4	3.0	0.2	2.9
20–24	1.4	1.4	0.1	1.2
>24	2.0	1.9	0.8	1.8
Standing deadwood				
0	94.9	96.5	99.8	96.2
4–8	2.0	1.0	0.0	1.3
12–16	1.1	0.7	0.0	0.8
20–24	0.7	0.7	0.0	0.6
>24	1.3	1.2	0.1	1.1

woodland biodiversity. Approximately 13% of the estimated 416 000 ha of semi-natural woodland in England is designated as SSSIs (Forestry Commission 2002a). Kirby et al. (2002) have produced guidance on methods for setting objectives for woodland condition within SSSIs but much of this is applicable to native and semi-natural woodland in general. Criteria relating to the "features of interest" or attributes upon which the original SSSI designation was founded are proposed to help with setting objectives and targets. One of the key features in most woodlands is the survival of veteran/ancient trees and the occurrence of dead and decaying wood habitats. Table 2 illustrates current levels of deadwood across all woodland types in the UK (not just native woodlands); a high proportion of the current high forest area contains no deadwood whatsoever.

There are plans to include other stand diversity indicators in the suite of national-scale woodland quality indicators, such as tree species and stand structure diversity etc. There is some evidence that these measures have potential as indicators (Larsson 2001; Humphrey et al. 2003a) but validation is critical as some studies have not established very convincing associations between structural attributes and species diversity (Jonsson and Jonsell 1999; Neumann and Starlinger 2001).

4. Problems with relying on indicators relating to key sites and habitat types

While the area and condition of ancient and native woodland is clearly of importance to biodiversity, it would be unwise to rely solely on these indicators as meaningful estimates of woodland biodiversity as a whole. UK BAP native woodland types and ancient woodlands are not the sole repository of forest-related biodiversity in Britain; recent and long-established planted forests of non-native tree species (which make up nearly 90% of the forest cover in the UK) provide suitable habitat for a wide range of flora and fauna (Humphrey et al. 2003a). Hence, condition and extent of these forest types are also important. Similarly in the Pacific North-West, the protection of old-growth forests, although important is now seen as just one of several conservation measures needed to conserve forest biodiversity in its totality (Er and Innes 2003).

There is also increasing evidence that indicators which focus on the condition of special (often small) sites alone are not sufficient to represent biodiversity, as many species use supposedly sub-optimal habitats (Angelstam and Andersson 2001; Johansson and Gustafsson 2001), suggesting that knowledge of the habitat requirements of these species is often incomplete (Hansson 2001; Johansson and Gustafsson 2001). Site-based approaches are also seen as unrealistic as species need to disperse amongst suitable habitat patches in the landscape in response to environmental change or the effects of natural disturbance (Simberloff 2001). Many key habitats cannot be expected to retain their habitat value indefinitely as they will be impacted by successional processes (Hansson 2001).

5. Species indicators

Given the problems with regarding habitat area and quality as adequate surrogates for biodiversity, there has been increasing interest in identifying species-based indicators of biodiversity (Linnell et al. 2000; Sætersdal et al. 2003; Simberloff 1998) whereby changes in the distribution and diversity in a measured species (or species group) reflects changes in unmeasured groups (Jonsson and Jonsell 1999; Williams and Gaston 1998). There is no implication of functional linkage amongst species groups, as the concept is essentially empirical. For example species with large area requirements – termed umbrella species (*sensu* Simberloff 1998) – have been used as indicators of diversity of other species groups occurring within the habitat or range area of the umbrella species (Linnell et al. 2000). This concept has also been developed for communities, where in large areas, smaller, species poor communities may be nested within larger species-rich communities (Hansson 1998).

In the UK, woodland birds are regarded as good indicators of the broad state of wildlife because they are wide ranging in habitat distribution and tend to be at or near the top of the food chain (Table 1). As a consequence, the woodland bird index has been adopted as a headline measure of sustainability in the UK Government's Quality of Life Counts initiative (DEFRA 1999). However, there is no universal acceptance of these (functionally unproven) species-based indicators (Williams and Gaston 1998). Good correlations have been observed amongst some species groups (e.g. Sætersdal et al. 2003), but not in others (Jonsson and Jonsell 1999). If species-based indicators of biodiversity are to be used, it is imperative that there are clear functional relationships established between the indicator species and the diversity of those species groups it is intended to represent. For example, bird diversity has been shown to be related to forest structure at the landscape scale, where different successional stages support different bird assemblages (Fuller and Browne 2003); successional stage can also be related to other species groups such as invertebrates (Humphrey et al. 1999).

6. Landscape metrics, focal species and habitat networks

There is increasing recognition that species distribution and diversity is governed by the pattern and configuration of forest landscapes as well as by habitat-scale measures such as patch size and quality (Brotons et al. 2003; Fahrig 1997; Lindenmayer et al. 2002; Noss 1999). There is a considerable body of literature concerned with the identification and enumeration of landscape metrics as biodiversity indicators (e.g. measures of connectivity; fragmentation; configuration etc.) without explicit link to species requirements or validated relationship with biological phenomenon (Haines-Young and Chopping 1996; Pukkala et al. 1997; Riitters et al. 1995).

To be useful as surrogate measures of biodiversity, landscape metrics must have some functional relationship to species requirements (Lindenmayer et al. 2002). The focal species approach (Lambeck 1997; Noss 1999) attempts to address the complexities inherent in relating species ecology to landscape parameters. The approach involves identifying a number of species each of which is used to define different attributes (e.g. key spatial and compositional elements) that must be present in a landscape if it is to retain its biota. Species should encompass a range of variability in landscape parameters such as habitat preferences, dispersal abilities and minimum patch size requirements.

A prototype GIS-based Decision Support System called BEETLE (*Biological & Environmental Evaluation Tools for Landscape Ecology*) is currently being developed to carry out landscape-scale assessments of biodiversity in Britain (Watts in press). BEETLE is based upon the concept of ecologically scaled landscape indices (Opdam et al. 2002; Vos et al. 2001) which when combined with spatial data on the distribution and extent of different habitat types, allows evaluation of different landscape scenarios for a range of focal species.

The basic outputs from the model are maps of functional networks for individual species. These networks comprise areas of suitable habitat which are functionally linked, i.e. the species in question is able to disperse successfully between the patches. The degree of dispersal is determined by the “permeability” of the matrix between suitable habitats. This is a crucial novel element of the modelling process, as landscape elements are often simply classed as “habitat” or “non-habitat” elements. Recent research suggests that in landscapes with high matrix quality (in terms of a particular species) less habitat is needed to ensure species survival than in landscapes with low matrix quality (Fahrig 2001).

Landscape quality for each focal species is assessed in terms of the number of networks supported by the landscape, the size of those networks, the size of the habitat patches within each network and the proportion of habitat in large networks. Some of these measures of landscape quality could be used as indicators of biodiversity. Overall biodiversity value could be assessed by comparing network quality for a range of focal species with different ecological requirements. The BEETLE modelling approach is currently being tested in a number of locations in the UK. The example in Figure 1 shows the effects of adding additional woodland on the number and size of networks in the Scottish Borders for red squirrel – a core woodland species (Humphrey and Quine in press). Further work is planned to assess the effects of this change in woodland on open ground species.

7. Conclusions and practical application of indicators

This short review of biodiversity indicators currently in use or being developed for use in British forestry, has highlighted that species diversity is influenced by a wide range of factors acting concurrently at multiple-spatial scales. Of the current suite of indicators currently in use (Forestry Commission 2002a), most are easy to operationalise (e.g. deadwood or woodland area) but provide an incomplete measure of biodiversity. In terms of future national-scale biodiversity indicators, a step forward might be to include area and condition targets for UK BAP priority woodland types. Measures of condition could be expressed either in terms of status or trend (or both) for example the number of positive, neutral or negative trends. Information on woodland condition is available from the National Inventory of Woodland and Trees, but care will be needed in the interpretation of specific measures of woodland quality as the relevance of these will differ between woodland types.

However, national-scale indicators are not particularly helpful in aiding land-use decision making at the landscape scale. Most British landscapes cater for both forest and non-forest

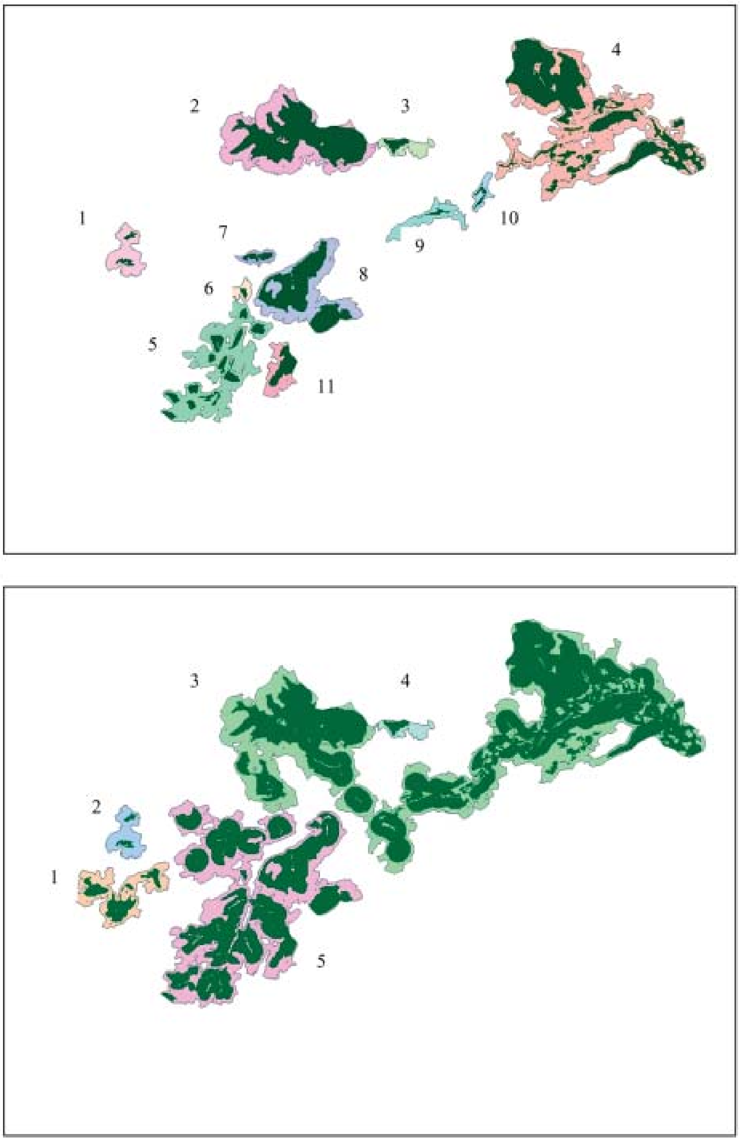

Figure 1. Habitat networks for red squirrel in part of the Scottish Borders, UK; A) before woodland expansion; B) after woodland expansion. Woodland patches are shown in dark shading, the lighter shaded areas represent the "non-habitat" parts of the network, i.e. parts of the network which species can disperse through. Individual networks are numbered. Reproduced with permission from Humphrey and Quine (in press).

species, and simple measures of change in woodland area and quality are meaningless unless set in the context of a particular landscape or forest management unit (Angelstam et al. 2001). There is increasing awareness that in some regions of the UK, a balance needs to be struck between the needs of open ground species and woodland species (Humphrey et al. 2003b). Landscapes which are fragmented from a wooded perspective often hold important populations of open ground species, and attempts to re-connect and expand woodland fragments could have a deleterious effect on the viability of populations of these open ground species (Humphrey et al. 2003b). The benefit of the focal species approach is that it allows evaluation of landscape quality (expressed in terms of network metrics) for a range of species with differing habitat requirements and has potential for aiding in scenario modelling of landscape change.

In terms of site and stand based management there is much that forest managers can do to enhance habitat quality through silviculture. However, these habitat enhancement measures need to be undertaken within the context of landscape-scale planning and site history (Angelstam et al. 2001; Kremsater et al. 2003; Verheyen et al. 2003). An example of this is the creation and maintenance of deadwood habitats in British woodland (Humphrey et al. 2002). A strategic approach is suggested, whereby woodlands are stratified into different types in terms of the potential importance of their deadwood habitats. Benchmark quantities/volumes of deadwood are proposed for each woodland type, for instance in old conifer stands the benchmark volume is set at 20–40 $m^3 ha^{-1}$ of a mix of standing and fallen deadwood. Such volume measures could be used as stand-level indicators. However, the potential value of this deadwood is much higher where sources of colonising species are in close proximity. Humphrey et al. (2004) found that the species richness of epixylic bryophytes in conifer stands was positively related to the amount of semi-natural woodland within 1 km.

In conclusion, it is apparent that a broad suite of indicators will be needed to adequately deliver biodiversity information at the range of different scales required by policy, makers, strategic planners and forest managers. The way that the indicators are packaged and presented will be equally as important as ensuring that the scientific basis is valid and robust (Angelstam et al. 2001). Measures of biodiversity need to be relevant and understandable to those that will be using and interpreting them.

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JRC Contribution to Reporting Needs of EC Nature and Forest Policies

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Abstract

Progress towards objectives set out in the European Policies (Natura2000/ Habitat Directive and Forest Focus regulation) could be monitored through the implementation of the European forest biodiversity indicators defined in MCPFE, BEAR project and EEA.

The target user group interested in biodiversity indicators may be split in (1) policy- and decision-makers at European scale, and (2) local and regional administrators and site managers. For the two groups, information needs, gaps and similarities are discussed.

The research activities conducted at the JRC are then introduced. They focus on the development and implementation of surrogate indicators (qualitatively connected) of forest biodiversity derived from multi-temporal Earth Observation and ancillary data sources. These indicators are geared to describing the European forest biodiversity in terms of (1) forest type, area and changes and (2) landscape forest spatial pattern and changes. Preliminary results are provided over both the whole European territory and selected Natura 2000 sites in the Alpine region.

Keywords: forest monitoring, spatial indicators, Natura 2000, Forest Focus, user needs, remote sensing.

EC Nature and Forest Policies and existing indicator initiatives

The work done at JRC aims to contribute to the reporting on the status and trend of European forest biodiversity to support the following instruments:

(1) Birds and Habitats Directives (Council Directive 79/409/EEC on the conservation of wild birds and Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora). These directives target the conservation of, among others habitats,

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natural forest habitats and the wild fauna and flora. The Natura 2000 network of protected sites is currently being established under the Habitat Directive. The Natura-2000 network demands for a report describing the conservation status of the sites and their hosting species and habitats on a 6-year basis.

(2) the EU Forest Focus regulation (Regulation No. 2152/03 of 17 November 2003 concerning monitoring of forests and environmental interactions in the Community) explicitly considers investigating biodiversity issues in addition to the requirements for monitoring of the impacts of atmospheric pollution and fires on European forests.

A consistent surveillance of the present state of forests among Member States is achieved by the deployment of both, harmonized spatial databases and customized indicators on pan-European level (Estreguil 2002).

There is an urgent need to implement currently available indicators for reporting on biodiversity at European level (Delbaere 2002). Available European sets of indicators relevant for forest biodiversity are:

1. "The improved Pan-European Indicators for Sustainable Forest Management" from the Ministerial Conference on the Protection of Forests in Europe (MCPFE 2003) comprising a set of six criteria and 35 quantitative indicators. Specific reference is on indicators set under criteria 1 (forest resources) and 4 (biodiversity);
2. The key factors of European forest biodiversity from the BEAR Concerted Action of the 5th EU Framework Programme (www.algonet.se/~bear). The forest ecosystem is characterized according to the major ecosystem attributes (structure, composition and function) and scales (national/regional, landscape, stand) (Larsson et al. 2001a)
3. Indicators core set for Biodiversity and Nature Protection and for Terrestrial environment of the European Environmental Agency (EEA 2003).

Users groups, information needs and gaps

The implementation of biodiversity indicators is a multi-scale and multi-disciplinary challenge involving the participation of different types of organizations with different information priorities based on the spatial extent of their responsibilities (Puumalainen et al. 2002). We propose to split the users group of forest biodiversity related indicators into two sub-groups:

1. Policy-makers at European level evaluating the progress made towards objectives stipulated initially. Policy development and reporting require information collected at local and regional levels serving as an input for computing indicators at national and ultimately European level.
2. Local and regional administrators deploying EU policies and indicators for practical use in management at the site level.

The geographical scopes, associated map scales, and indicator types of these hierarchical levels are summarised in Figure 1.

European policy makers need clear statements concerning:

- The present situation of European forest biodiversity at different hierarchical levels like species, habitat, ecosystem.
- Evolution over time based on historical data and used to measure progress towards biodiversity goals

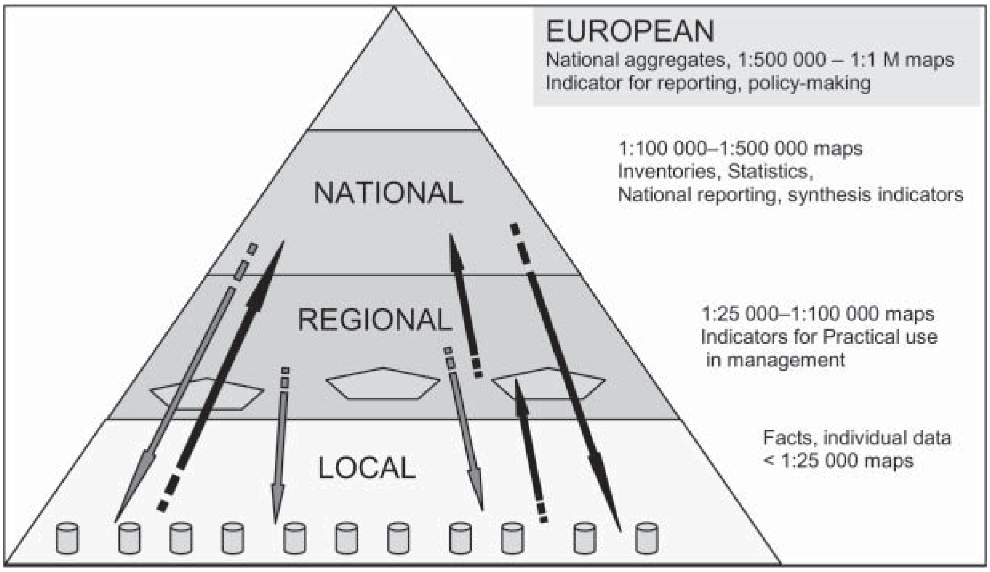

Figure 1. Geographical scopes, data, scales and indicators for the local, regional, national and European levels.

- Cause-effect relationships. This includes gaining knowledge on driving forces and pressures within the forestry sector as well as activities like agriculture, tourism, transport, etc., and their impacts on forest biodiversity

European level of decision-making may also need both detailed and general information for example: (1) data collected at local/regional level (therefore quite detailed and at fine scale) for the enforcement of legal rights and obligations when applicable or/and to show areas at risk; (2) inform on the general situation in a given geographic area on forest biodiversity overall situation.

The JRC is leading the indicator work-package in the project EON2000+ (Earth Observation for Natura 2000 plus at www.eon2000plus.org/Overview) which targets the information needs of organization acting at local/regional level. Project's partners have organized workshops with users to investigate their reporting and management needs for Natura2000 sites. Common information needs were collected for sites in rural landscapes, alpine landscapes and wetlands, namely:

- *Habitat and land cover geo-referenced maps* at scales ranging from 1:5000 to 1:50 000 to address composition and structural issues*;*
- *Vulnerability Status of protected site*: it is assessed through the knowledge on both, human and other pressures upon this site in terms of type and extent, and the ecological vulnerability (sensitiveness) of the protected habitats and ecosystems to these pressures.
- *Pressures in the wider countryside upon protected habitats*
- *Habitat suitability maps for targeted species*

Table 1 provides an overview of information and indicator requirements for forest habitats and forest types, indicators have been grouped into three groups depending on their purposes.

Cross-relation of indicators developed for local/regional administrators and site managers with European policy-making indicators must be ensured to facilitate a common reporting

Table 1. Information requirement for forest habitats protected in Natura 2000 sites from local/regional organizations in the UK, Germany, Italy, France (adapted from Estreguil et al. 2003).

Indicator	Additional information	Relevance to European set
Indicators to answer on the state and trends of forest biodiversity		
Forest in N2000 site and outside: - Distribution, area, Proportion - Natural/managed forest - Share broadleaf/coniferous and changes over time - Single/mixed species Forest Annex I Forest Habitat - Distribution, area, Proportion - Habitat suitability for species - Wetness/soil moisture	Forest map at 1ha Minimum Mapping Unit (MMU) Scales 1:25000, 1:50 000 Geo-location and area size, change over time Habitat + sub habitat species (birds, invertebrates, plants) – Forest map at 0.01 ha	BDIV1 BDIV12 MCPFE 4.1; BDIV2d MCPFE 4.3; BDIV 1d TELC4
Indicators to answer on the causes on the loss of biodiversity		
Forest in N2000 site - Continuity of canopy - Forest spatial pattern (Openness in forest, Landscape closure, Connectivity in forest, edges type) - Minimum area size, increase in woody vegetation area size - Disturbance from path and roads - Land cover pattern	Forest type and Annex I Forest habitat ; Map at 1 ha and 0.01 ha MMU; Map at 1:100 000.	MCPFE 4.7; BDIV6a TELC2 BDIV5
Indicators to answer on measures and their effectiveness		
Legal status and ownership: - Distribution of forest ownership in site - Proportion of forest under special protection regime - Human activities in site - Trend in forest conversion to other use	- Forest area (and changes) accordingly to ownership structure (private/state) - scales 1:25 000 and 1:100 000 strictly protected/managed - Land use map, information on clear cut, burnt area, silviculture practices, conversion to agriculture, infrastructure building, drained mire/peat extraction	BDIV10 BDIV13 TELC4

and coordination among organizations operating at local, regional, national and European levels (Estreguil et al. 2003). Within Table 1, the relevance of each indicator proposed at local/regional levels to the EEA and MCPFE European Core set of indicators is demonstrated for the three purposes:

1. Indicators to answer on the state and trends of forest biodiversity:
   MCPFE 4.1 and BDIV 2d: *Area of forest and other wooded land, classified by number of tree species occurring and by forest type.*
   MCPFE 4.3 and BDIV1d: *Area of forest and other wooded land, classified by "undisturbed by man", by "semi-natural" or by "plantations", by forest type*.

BDIV1+12: *Habitat diversity* (in and outside designated areas)
TELC4 *Land cover change*

2. Indicators to answer on the causes on the loss of biodiversity:
   MCPFE 4.7 and BDIV6a: *Landscape level spatial pattern of forest cover*
   BDIV5 *Threats to ecosystems*
   TELC4 *Land cover change*
   TELC2 *Fragmentation of ecosystems and habitats by transport infrastructure*
3. Indicators to answer on the measures taken to maintain biodiversity and on their effectiveness
   BDIV10 *Designated areas* (forest protection regime and protection instrument)
   BDIV13 *Human impacts on designated areas*
   TELC4 *Land cover change*

Finally, the implementation of available indicators at the European level is not a straightforward task despite programmes and projects that have increased the knowledge on forest biodiversity in the last years (Puumalainen et al. 2002; Estreguil et al. 2003). Pan-European biodiversity indicators require data availability and harmonization and clear definitions of targets and thresholds. However, soft or broad qualitative objectives are often provided (wording like to protect, to restore, to halt the loss, report on favourable conservation status, etc.) in policy document. Guidelines on how and what to measure in order to report on progress need to be written and coordinated among Member States. European forest Biodiversity can be promoted from different perspectives, among others: ethic, aesthetic, genetic diversity, potential uses of species, species richness or in association with sustainable management (MCPFE). These perspectives require different type of indicators and measurements. Also, surveys at local / regional levels outline that biodiversity objective for protected sites are not always clearly defined. Aggregates of different protection categories involving different permissions of human activities further render evaluation difficult.

Regarding target and threshold, they are rarely mentioned and are mostly reporting deadlines like for example, every 6 years requested in the Habitat Directive. We may comment that the temporal range of ecosystem response, apart for some specific habitats, may be at least 20-50 years and 6 yearly reporting frames would primarily capture human impacts and may therefore be more adapted to monitor pressure indicators. To capture ecosystem response over Europe, National Forest Inventories would be relevant but a pre-defined temporal frequency for all countries should be defined for their field surveys. Indeed their temporal resolution still varies between countries even if a ten year cycle is often applied. First assessment year ranges at the earliest in end of the 19th century (Denmark) and at the latest in 1997 (Flanders).

Regarding data, reporting on forest biodiversity and protected areas is heavily linked to the use of multi-scale geographical information (Figure 1) and require data collected at European, national, regional and local scales. Compilation of existing data for biodiversity assessments generally offers large scope for bias and confusion (Vanclay 1998). The compilation of the Natura2000 standard form relies on available field surveys and maps over the protected site. On the other hand, for a European wide assessment based on sampling-based surveys, forest inventories are a major source of information at least for traditional variables describing tree species composition, age, etc. Forest biodiversity related variables are however collected in different countries according to different measuring and sampling procedures, variables, definitions and standards during different time periods. Also the main goal to collect such data is mostly for timber production, even if in the last decade biodiversity related parameters have been introduced. Poor harmonization of National Forest Inventories data and the complete lack of biodiversity related data from some countries are of course a major drawback from the perspective to use national data as a sole basis for monitoring forest biodiversity at European scale (Puumalainen et al. 2002).

JRC research activities contributing to the implementation of indicators

The Joint Research Centre (JRC) focuses on providing spatially explicit forest biodiversity related information based on the use of remote sensing and GIS techniques. Remote sensing data and spatial metrics combined with other geo-information derived from field surveys and available digital data are particularly suitable to tackle large-scale forest biodiversity reporting. The contribution to the latter relies in assessing surrogate indicators (qualitatively connected) of biodiversity. Quantitative assessment of forest pattern, forest type and change would help to better understand the population decline of several plant and animal species in fragmented and/or disturbed landscape. JRC focuses on three main activities that will be introduced in the next sections:

1. Selection of relevant European Reporting Units
2. Compositional indicators: forest type, area and changes (MCPFE 4.1)
3. Structural indicators: Forest spatial pattern (MCPFE 4.7)

Selection of relevant European Reporting Units

The large diversity of the European forest ecosystem, accentuated in different ways due to variable natural and anthropogenic effects, its history and ownership structure render any biodiversity assessment at European level difficult. In case of wide areas, such as the European territory, the adoption of a regionalisation procedure is helpful to recognise homogeneous regions with respect to biotic and abiotic properties within which the same methodology for reporting can be applied. Moreover it provides a geographic framework to analyse ecosystem/sites in relation to their environment and in which similarities on anthropogenic and natural pressures as well as on responses like management strategies may be expected. A set of relevant European reporting units have been identified in addition to the traditional administrative units: the bio-geographic regions of the Habitat Directive (Noirfalise 1987), the forest type regions and strata (Larsson et al. 2001; FIRS 1995), the environmental classification of Europe (Metzger et al. 2003), the Digital Map of European Ecological Regions (DMEER; Bunce 1995)), the geobiological divisions of Europe (Ozenda 1994), the natural regions of Europe (Arnould et al. 1992). All the described land classifications differ by the approach used to determine boundaries of eco-regions (statistical, controlling factor, clustering, intuitive and expert knowledge), the considered parameters (latitude/longitude, landscape form, topography, climate, soil, natural potential vegetation, geo-botanical variables, forest management) and the scale of application (1: 10 M to 1: 1 M). A review of such classifications and their relevance for reporting trends in European forest biodiversity is currently being done at JRC.

Compositional indicators: forest type, area and changes

Compositional indicators such as forest type, forest area and changes require the development of robust methods to either discriminate forest types and/or both monitor acknowledged forest processes for the region of interest.

Within the Alpine region for example, major threats for forest biodiversity are associated with the general collapse of the agro-pastoral system after World War II and the expansion of tourism in the last decades. The latter leads to clear cuts and erosion processes while the former leads to recolonisation or invasion processes of two types: forest edges dynamics where the forest progresses at the expenses of meadows and within forest succession, where an open forest is closing up (Didier 2001). Few decades, even 10 to 15 years are considered sufficient for the development of dense woody pioneer formations (Guidi and Piussi 1993), 4–5 meters high, but vegetation dynamic can vary according to climate, lithology and pedology. Species requiring shadow conditions can employ up to 30–40 years to develop. High resolution (30 m) satellite remote sensing data available offer a maximum temporal interval of 16 years.

Forest types – however defined – reflect the specific local conditions of forests and the conditions of the biogeographic regions in Europe. The correspondence of remotely sensed derived nomenclatures must be worked out with the three following European classifications: Larsson et al. (2001) has defined within the EU BEAR project, a total of 33 forest types for Biodiversity assessment in Europe. Also to consider is the Pan-European system of classification of natural and semi-natural habitats (EUNIS Habitat Classification; Davies and Moss 1999) and the Habitat Annex I of the Habitat Directive.

Forest types discriminated within the European forest WIFS based map (GAF, 2001) which is the only European wide forest map available at scale 1:500 000, are very broad (predominantly broadleaved, conifers, mixed forest and other wooded land) with an overall accuracy of 78%. The newly released European Image2000 product (http://image2000.jrc.it/) offers a consistent European coverage based on orthorectified satellite imagery at 25 m (Landsat ETM) and is of interest to be evaluated for forest classification. Indeed, the better spatial and spectral resolutions of Landsat ETM (6 bands, 30 m) are expected to improve the information level available in the current WIFS based map (2 bands, 180 m). In the view of using this European Image2000 product, forest type monitoring and discrimination of forest processes, like increase of woody vegetation, are currently addressed for Alpine Natura2000 protected areas and further details can be found in Maggi et al. (2003) and Cerruti (2004).

Methodologies being tested are:

1. for classification purposes:
   - Traditional maximum likelihood algorithm (MLC);
   - Hybrid classification which combines a spectral box classification (Richards 1986) and spatial information known as seed region growing technique (Adams and Bischof 1994);
   - Integration in the classification process of multi-season information and DEM (altitude and slope). The map obtained over the Italian Val Grande site using this approach is illustrated in Figure 2;
2. for land cover change purposes
   - Simultaneous analysis of multi-temporal data by univariate normalized image differencing and production of a land cover change map with the following change classes: increase of woody vegetation, increase of herbaceous biomass, decrease of woody vegetation, from vegetation to mineral ground.
   - Comparative analysis of independently produced classifications for different dates, called post-classification comparison.

The feasibility to relate the Landsat derived nomenclature with four European classifications (the CORINE Land Cover nomenclature, the BEAR forest types, the EUNIS and the Habitat Annex I Classifications) is demonstrated for the Val Grande site in Table 2. This was achieved through downscaling the land cover map using the available IPLA (2001) map at 1: 25 000 scale.

Estimates derived for land cover change over a 15 years period (1985–2000) for three Alpine Natura 2000 site show that the area exposed to changes amounts for (1) over 30% of

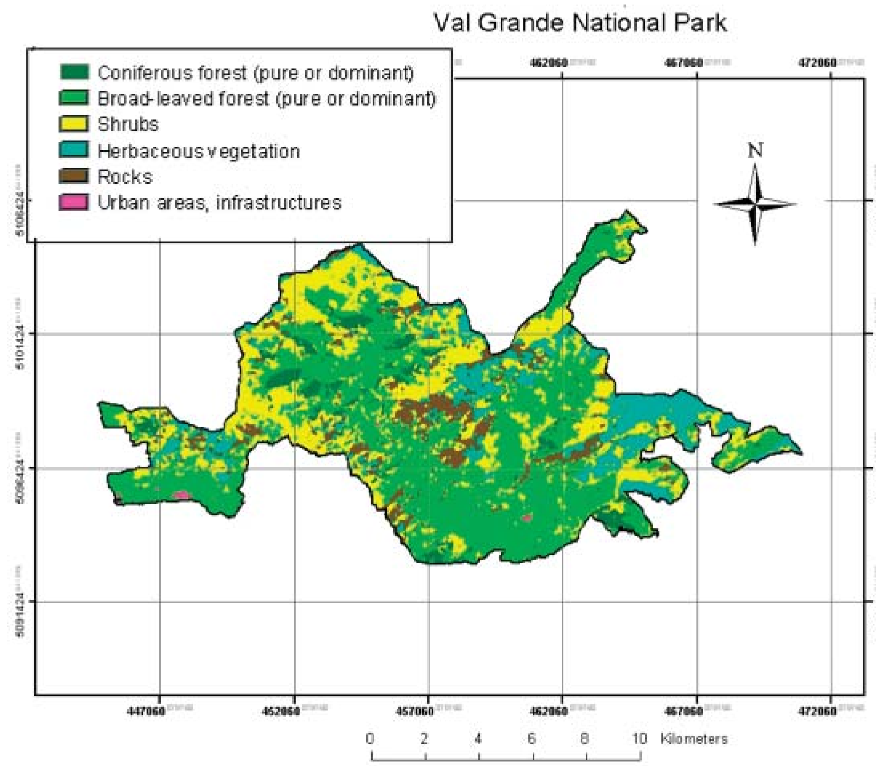

Figure 2. Land cover map of the Natura 2000 Val Grande National Park, based on multi-season classification of 3 Landsat TM, integrating information on altitude and slope (DEM 50 m).

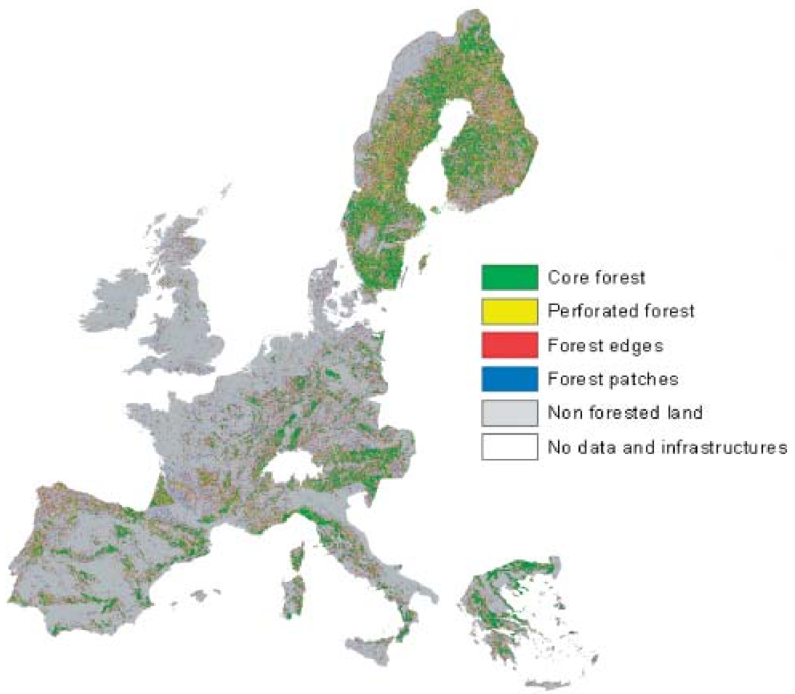

Figure 3. Forest spatial pattern indicators derived for the European forest WIFS based map (1996–1999).

Table 2. Correspondence of Landsat derived nomenclature with European classifications.

Landsat based Map	IPLA,2001 categories	CORINE Land Cover	EUNIS	Habitat	BEAR
Coniferous forest (pure or dominant)	*Abies alba*	Coniferous forest (**3.1.2**)	Acidophilous *Abies alba* forests (**G3.1**)	-	Fir and spruce woodland (**FT3N.1**)
	Pinus uncinata		*Pinus uncinata* woodland (**G3.3**)	*Pinus uncinata* forests (**4140**)	Alpine larch-Arolla and mountain pine woodland (**FT3N.2**)
	Pinus silvestris		South-western Alpine mesophile *Pinus sylvestris* forests (**G3.4**)	-	Scots pine woodland (**FT3N.3**)
	Picea abies		Alpine and Carpathian sub-alpine *Picea* forests (**G3.1**)	Acidophilus forests (*Vaccinio-Picetea*) (**9410**)	Fir and spruce woodland (**FT3N.1**)
Broad-leaved forest (pure or dominant)	*Acer pseudoplatanus, Tilia cordata, Fraxinus excelsior*	Broad-leaved forest (**3.1.1**)	Ravine and slope woodland (**G1.A**)	*Tilio-Acerion* ravine forests (**9180**)	-
	Alnus incana, Alnus glutinosa		Riverine *Fraxinus-Alnus* woodland, wet at high but not at low water (**G1.2**)	Residual alluvial forests (*Alnion glutinoso-incanae*) (**91E0**)	Fluvial and riparian woodland (**FT1N.2**)
	Castanea sativa		*Castanea sativa* woodland (**G1.7**)	*Chestnut woods (9260)*	Thermophilous deciduous woodland (**FT1N.4**)
	Quercus cerris		-	-	Acidophilus oak-dominated woodland (**FT1N.5**)
	Quercus petraea		Continental *Quercus petaea* forests (**G1.8A**)	-	
	Fagus Sylvatica		Medio-European acidophilus *Fagus* forests (**G1.6**)	*Luzulo-fagetum* beech forests (**9110**)	Mountain mixed beech forests (**FT1N.2**)
Shrubs	Pioneer vegetation (*Betula pendula, Corylus avellana*)	Transitional woodland/shrub (**3.2.4**)	Small broadleaved deciduous anthropogenic woodlands (**G5.2**)	-	-
	Rhododendrum ferrugineum with sparse larches (*Larix decidua*)	Moors and heathland (**3.2.2**)	Western *Larix*, mountain pine and *Pinus cembra* forests (**G3.2**)	Alpine forests with larch and *Pinus Cembra* (**9420**)	-
	Alnus viridis		Mountain *Alnus* brush (**F2.3**)	-	-
	Bushes		Alpine acidocline *Rhododendron* heaths (**F2.2**)	Alpine and subalpine heats (**4060**)	-
	Grazed bushes				-
Herbaceous vegetation	Unused grassland	Natural grassland (**3.2.1**)	*Nardus stricta* swards (**E1.7**)	Species-rich *Nardus* grasslands, on siliceous subsrtrates in mountain areas (and submountain areas, in continental Europe) (**6230**)	-
	Grassland	Land principally occupied by agriculture, with significant areas of natural veg.(**2.4.3**)			-
	Rupicolous grassland	Sparsely vegetated areas (**3.3.3**)	Alpine and subalpine grasslands (**E4**)	-	-
	Pastures	Pastures (**2.3.1**)	Permanent mesotrophic pastures and aftermath-grazed meadows (**E2.1**)	-	-
Rocks	Rocks	Bare rock (**3.3.2**)	Alpine siliceous screes (**H2.3**)	Siliceous (**6110**)	-
Urban areas, infrastructures	Urban areas, infrastructures	Continuous urban fabric (**1.1.1**)	Buildings of cities. towns and villages (**J1**)	-	-

the site where wilderness is the management priority and where relatively high temperatures and heavy rains support even more recolonisation processes, (2) approx 10% of the two other sites where agro-pastoral practices are enforced to keep the landscape open. Of the area interested by changes the greatest part corresponds to an increase of woody vegetation of over 60% outlining trends associated to land abandonment.

Structural indicators: Forest spatial pattern

Structural indicators related to forest spatial pattern refer to the assessment of forest connectivity, forest fragmentation, forest isolation, edge/interior forest. These parameters are determined following the approach described in Riitters et al. (2002). In short, the algorithm calculates the forest area density (*Pf*) and forest connectivity (*Pff*) for a neighborhood of a given size overlaying each pixel classified as forest. Applying pre-defined thresholds to the interrelation of Pf and Pff results in the classification of the forest spatial pattern classes core, perforated, edge, and patch. This method was successfully applied on European scale to the WIFS based forest map (GAF 2001), (Figure 3), and on local scale to the Landsat TM based map of the Val Grande Natura 2000 site.

This site and land cover maps derived for the years 1985 and 2000 were used to demonstrate the feasibility to assess trends in forest spatial pattern (ref. Figure 4). The results of this research, and this particular site, indicate an increase in core forest and patches sizes due to land abandonment and invasion processes accompanied by a decrease of the size and number of open areas in the forest (decrease of the perforated forest class).

Trends over the European territory will now be investigated over representative European forest strata and regions as proposed in the section on “European reporting Units”. For this purpose, historical data will rely on the CORINE Land Cover product for year 1990 and on the classification of Landsat TM for year 1985. Present situation will be derived from the classification of Landsat images following the more robust methodology among methods listed in the previous sub-section by using the IMAGE2000 database and ancillary data.

Conclusions

The European territory and its high complexity of landscape structures compel for a regional approach to discriminate homogenous regions to which the same forest biodiversity monitoring strategy and associated indicator set can be applied. A list of available regions and indicator sets has been provided. Here it is crucial to ensure the consistency and cross-relation of the indicator set defined by managers at local levels, and for European policy-making and reporting. The feasibility of this cross-link was demonstrated for forest habitats over selected sites.

A list of methods for mapping and monitoring of forest type was provided with some preliminary results. These results provide spatial information on forest main types that can be further link to European forest types and habitat nomenclatures. Forest spatial pattern (core, perforated, edge, patches) analysis in time and space was demonstrated. They contribute to the implementation of the indicators MCPFE 4.1 and EEA/ BDIV2d (Forest type area and changes) and MCPFE 4.7 and BDIV 6 (Landscape forest pattern and changes).

Present activities include the mapping and monitoring of forest spatial features and invasion processes over a 16 years time-frame using the Image 2000 European product and additional Landsat TM images from the year 1985.

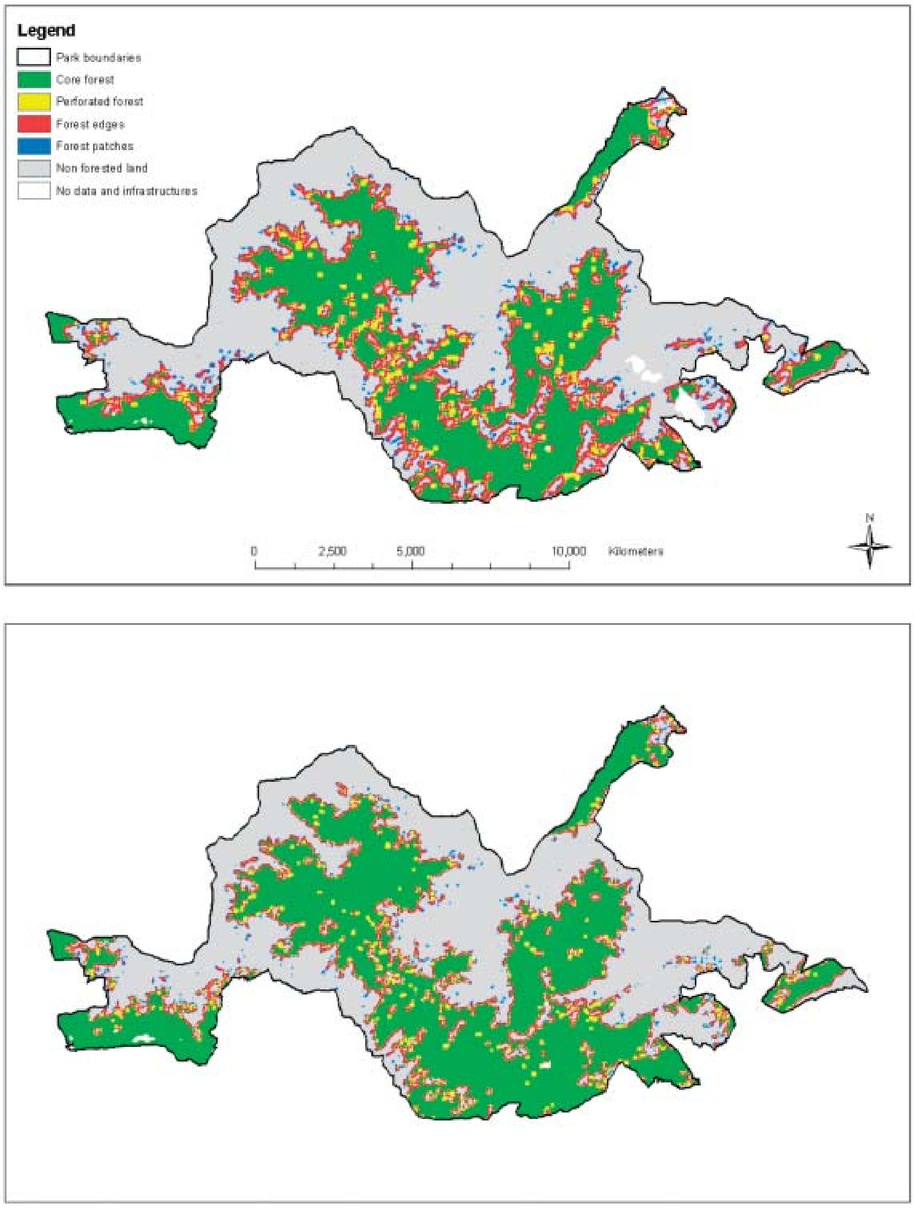

Figure 4. Changes in Forest landscape pattern 1985 (top) to 2000 (bottom) over the Val Grande Natura 2000 site.

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Session 2: Forest Classification and Assessment Strategies

Forest Types for Biodiversity Assessment (FTBAs) in Europe: the Revised Classification Scheme

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Abstract

European forest biodiversity is the result of two main jointed forces: a) the post-glacial recovery of forest species at the end of last Ice Age and b) the human impact. The aim of this work is to extend the covarage of classification to the forest presently growing in EU-25 countries and to increase the oparationality of the classification. The paper describes an implementation of forest type nomenclature for biodiversity assessment at a Pan European level. Based on the previous 33 Forest Types for Biodiversity Assessment (FTBAs) an updated scheme of classification has been proposed. A sample of top level classes has been discussed focusing on class definition, geografical ditribution, key factors (structural, compositional and functional factors), subtypes.

Keywords: forest types, biodiversity key factors, biodiversity assessment, forest classification system.

1. Introduction

The actual appearance of forest biodiversity in Europe is the result of the evolution of the communities under ecological factors (climate, edaphic condition), disturbances (e.g. fire) and a long history of anthropogenic influence. The present distribution of European forests, their composition and structure are the combined reflection of two interplaying driving forces:

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1. the post-glacial recovery of forest vegetation at the end of the Ice Age, leading to a primary differentiation of the major forest communities throughout the last 10000 years (Holocene);
2. a further differentiation under the human impact starting from the Neolithic age (8000–5000 BP), when anthropogenic disturbances (grazing, burning and agricultural clearance) started to interfere with natural forest ecosystems, leading to a modification of forest biota.

Both processes have left footprints which are still visible on the actual biodiversity of European forest, that can be outlined as follows:

A. The establishment of forest communities followed the regression of the icecap; boreal coniferous forest and oak-beech and beech-spruce forests of central Europe are the youngest European forests, only took shape some 5000–3500 years ago. Southern Europe's forests are much older. Some of these formations or forest species are even relics of Tertiary flora (e.g. Macaronesian Laurel forest, *Juniperus thurifera, Pinus peuce, Aesculus hippocastanus*). Southern European forests have been much less influenced by the glaciations resulting in a much higher species richness and diversity of their floristic associations. As a result, European forests show North-South floristic gradient; about 2/3 of the forest species of European flora grow in southern Europe; the Mediterranean and Macaronesian Regions, notably, are the richest in endemic species. Further floristic variability in Europe is found along the West-Eastern gradient from oceanic to continental influence, associated with a decrease in the richness of species and vegetation types.

B. The history of human impact on forests also follows a South-North gradient. The first extensive deforestation and intensive wood exploitation occurred in the Bronze Age and was strictly tied with the rise and fall of the first civilizations (Perlin 1991); these emerged early in Southern Europe in at the beginning of the Bronze Age (Crete, Minoan civilization, early second millennium B.C.) spreading throughout Europe in the Late Bronze Age (e.g. Mycean Greece, proto-celtic culture of Urnfield people).

The effect of past anthropogenic influence becomes evident when observing the differences, in terms of forest cover and forest types between the potential European vegetation, as mapped by Bohn et al. (2000; see map in Bradshaw and Møller, in this volume) and the actual forest vegetation as described in EUNIS Habitat classification (European Topic Centre on Nature Protection and Biodiversity 2003).

The anthropogenic footprint on actual forest biodiversity can be summarized as follows:

i) most of the area of natural European forest is lost; the largest forest complexes found in Fennoscandia and Central Europe (partly resulting from extensive afforestation) are stark contrast with fragmented forest areas of southern Europe (see also potential and current forest vegetation maps in Bradshaw and Møller, in this volume). According to an evaluation carried out by Smith and Gillett (2000), the current forest loss in Europe varies according to natural forest types as mapped by Bohn (2000), ranging from more than 90% in mesophytic deciduous oak and oak-ash woods to approximately 50% in boreal or oromediterranean coniferous forests.

ii) pristine or near pristine forests are absolutely rare in Europe and even more so in EU-25 countries; in addition pristine forest area continue to decline to an alarming rate (WWF 2003); purely artificial forest is not really abundant and the largest part of European forest are considered semi-natural. Under this general heading are therefore included most of European forests, whose biological diversity has been more or less modified by human activities (modification of species composition and forest structure by silviculture, reforestation, etc.).

This being said, the question here discussed is to find a scheme to classify European forests upon a limited number of types each being reasonably different from the others, and internally homogeneous, as regards the determinants of forest biodiversity. The idea has been suggested for the first time in the EU funded BEAR, which outlined a strategy for assessment of forest biodiversity in Europe building upon *key factors* (natural and anthropogenic determinants of forest biodiversity) and related indicators and on the use of a forest type stratification to optimise the assessment (Larsson et al. 2001). The suggested forest type stratification was based on 33 Forest Types for Biodiversity Assessment (FTBAs) including an heterogeneous mixture of actual and potential forest type in Europe, with sensible differences also in terms of geographical distribution (see also Bradshaw and Møller, this volume). The need of overcoming this limitation and of finalising the classification for practical use has prompted a revision of the BEAR FTBA classification. In this paper we present the new classification nomenclature and basic contents of the revised FTBAs. Further details on the revision are also given in this volume in the paper of Bradshaw and Møller.

2. Structure of the FTBAs review

2.1 Targets

Two main targets have be considered in the FTBAs review:

1. Classification coverage, to be extended to the forests presently growing in EU-25 countries.
   To this end a check-list of relevant forest ecosystems to be included in the classification has been compiled taking into consideration the following references:
   - EEA/Eunis Habitat classification (European Topic Centre on Nature Protection and Biodiversity 2003), being a comprehensive inventory and description of pan-European natural, semi-natural and anthropogenic habitats;
   - the list of forest types included in European Habitats Directive (Annex 1, Council Directive 92/43/EEC), as it covers forests important for biodiversity conservation to be protected under Natura 2000 network;
   - all relevant literature giving a comprehensive the description of forest vegetation in Europe (Rodwell et al. 2002; Ozenda 1994, Polunin and Walters 1987).
2. Increasing the operationality of the classification, by:
   2.a. a simplification of original BEAR FTBAs classification, into a more workable and internally coherent scheme; to this end original BEAR types have been clustered into new broader classes including also missing types as identified from the check-list. The revised classification structure is commented further on (see 2.2).
   2.b. a finalization of types descriptions, to allow a clear distinction of the unique set of key factors determining biodiversity variation among forest types, as derived from relevant literature. The contents of FTBAs description are summarized in Chapter 2.3.

2.2 FTBAs classification

The revised FTBAs classification scheme is structured into 14 top level classes, hereafter *FTBAs* or *categories* (Table 1).

Top level classes are conceived mainly as keys to enter the classification through a simplified nomenclature. The coverage of each class is given in Table 2. The categories are however too broad for an operational assessment of forest biodiversity at regional or local

Table 1. The new FTBAs nomenclature and cross-links with previous Bear FTBAs and to the natural forest vegetation types of Europe (Bohn et al. 2000).

FTBAs	Bear FTBAs	Natural vegetation of Europe
1. Boreal forest	2;3;4;5;6	C2;D1;D2;D3;D4;D5;D6;D10;D11
2. Hemiboreal and nemoral Scots pine forest	7	D8;D12
3. Alpine coniferous forest	8;1	C3;D9
4. Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest	9;10;17	F1;F2;P1
5. Oak-hornbeam forest	11	F3;F4
6. Lowland to submontane beech forest	12	F5a
7. Montane beech forest	13	F5b
8. Thermophilous deciduos forest	14;27	G2;G3;G4
9. Mediterranean and Macaronesian sclerophyllous forest	15;25	J1;J2;J3;J4;J5;J6;J7;J8;
10. Mediterranean and Macaronesian coniferous forest	16	K1;K2;K3;K4
11. Swamp forest	20;21;22	S3;T1;T2
12. Floodplain forest	23;24	U1;U2;U3;U4
13. Native plantations	27;28;29;30	-
14. Exotic plantations	31;32;33	-

scale; to this end *sub-types* have been identified for each FTBA, i.e. a suggested further stratification for optimising biodiversity assessment within the class.

The rational the revised classification structure is to categorise FTBAs based on:

i) *variation of forest building trees and/or biogeographical or site ecological conditions leading to a differentiation of forest communities at European scale*. These three factors are universally recognized criteria of forest vegetation classification in Europe and can be considered as the major natural determinants of variation in forest biological community on a large scale.

ii) *to separate purely anthropogenic forest types (plantations), typically intensively managed from other semi-natural types*; silviculture, notably forestry intensity, is a well known anthropogenic factor shaping the structural, compositional and functional features of European forests.

In plantation forests, a consistent rate of European forests, the alteration of original forest biodiversity is highest for structural and composition properties are monotonously uniform and natural processes are altered (e.g. natural regeneration, resistance to disturbances). In addition, in intensively managed plantations (short rotation forestry, productive forests) these key-factors are under an active and strict anthropogenic control and this has a major influence on the development of forest biodiversity. The distinction native vs exotic is based on the assumption that plantations of native species are better adapted to interact with local flora and fauna.

In all the other FTBAs despite anthropogenic influence (silviculture, agriculture and grazing) a ‘residual’ biological diversity of natural forest communities is left. This is more or less developed depending on the marginality of the environment where they occur and on local forest history. Old-growth forests, for instance, occur where a favourable combination of environmental factors and historical events exist for a long time: e.g. inaccessible areas like abrupt rocky slopes or inner ranges, State-owned forests, former ancient royal forests, included in protected areas.

Human activities have also produced man-made habitats or cultural landscapes of importance for biodiversity conservation (Britain wooded pastures, dehesas, old chestnut groves, etc.).

2.3 FTBAs descriptions

The updated FTBAs description is structured according to the following frame:

- *Class definition*: key to the identification of the FTBA; it is a general description of the type in terms of dominant forest species and biogeographical/ecological factors determining its appearance.
- *Geographical distribution*: actual distribution of the type referenced to EU Biogeographical Regions. Notes about level of forest cover fragmentation are also given.
- *Key factors*: description of the most relevant structural, compositional and functional factors responsible for the variation of forest biodiversity within the class (see Chapter 3).
- *Subtypes*: list and description of (lower level) classes suggested for a further FTBA stratification for biodiversity assessment. Cross-links with Annex 1 Habitat Directive forest habitats are also established at sub-type level.
- *Relationship to previous BEAR types*: Cross-links with the original 33 Bear parent units.
- *Examples*: Photographic gallery on the FTBA.
- *References*: Bibliographical sources used in the compilation of the revision of each FTBA

Focus on key factors

In order to give an overview of the main results of the FTBAs revision, we report in the following on biodiversity key factors for a selection of semi-natural FTBAs:

- Boreal forest (FTBA 1)
- Hemiboreal and nemoral Scots pine forest (FTBA 2)
- Alpine coniferous forest (FTBA 3)
- Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest (FTBA 4)
- Oak-hornbeam forest (FTBA 5)
- Beech forests (FTBAs 6, 7)
- Mediterranean and Macaronesian forests (FTBAs 8, 9, 10)

A description of the Floodplain forests (FTBA12) is given in the paper of Bradshaw and Møller (this volume).

For a better understanding of the role played by anthropogenic influence in shaping the actual biodiversity of each FTBA a brief summary of forest history is also given; this includes also estimates of the residual area left for each FTBA in Europe – expressed as percentage on its natural potential area – according to the evaluation carried out by Smith and Gillett (2000).

Boreal forests and hemiboreal forest (FTBA 1)

Forest history

Since the earliest time of human settlement boreal forests were used as an economic resource by rural populations for timber, charcoal and tar production. For a long time numerous small family holdings combined forestry with agriculture in much of the boreal region; this has had a relevant impact of the forest landscape as the rural populations relied on the practice of

Table 2. FTBAs classification structure: class description and subtypes.

FTBA name	Class description	Subtypes
1. Boreal forest	Boreal spruce, pine and birch forests of Fennoscandia. Boreal forest with an oceanic character can be found also in central and northeastern Grampians of Scotland.	1.1 *Picea* taiga forest, dominated by *P. abies* or in northernmost Fennoscandia, *Picea obovata* predominantly found on mesic to most fertile sites. 1.2 *Pine* taiga forest, dominated by *Pinus sylvestris* predominantly found in nutrient poor and dry sites. 1.3 Boreal birch forest, dominated by *Betula pubescens spp. czerepanovii* in highest altitudes and latitudes in Northern Scandinavia. *Betula verrucosa*, *B. pubescens* and *Sorbus aucuparia* forest are also found on fertile sites in central and south Fennoscandia.
2. Hemiboreal and nemoral Scots pine forest	The hemiboreal forest covers the transitional forest zone between the boreal coniferous and temperate deciduous ones: *Pinus sylvestris* and *Picea abies* are dominant, but are mixed with broadleaves. In EU-25 countries this forest zone covers: i) as a wide belt south-central Sweden, thinning out in southernmost parts of Norway and Finland; ii) most of Estonia, Latvia and NE Poland. Lowland to montane, non alpigenous, *Pinus sylvestris* dominated forest of the middle European nemoral zone alpine are also included.	2.1 Hemiboreal forest in mesic to fertile and moist site types in southern Sweden and southernmost Norway and Finland. 2.2 Nemoral *Pinus sylvestris* forest, pure o mixed with broadleaves (*Quercus sp.*, mainly) of sandy lowlands of Poland and Germany, Po terraces and of the (sub-continental) montane level of Central Massif.
3. Alpine coniferous forest	Coniferous forest of the alpine biogeographic region of Europe, Scandinavian Alps excluded. In EU-25 countries the FTBA covers several isolated sub-regions corresponding to the main European mountains: the Pyrenees, the Jura, the Hercynian ranges, the Alps, the Central-Apennine and western Carpathians.	3.1 Subalpine larch-stone pine and dwarf pine forests, covering the upper belt of subalpine forest vegetation and the transition to shrub vegetation. 3.2 Subalpine and montane spruce and montane mixed spruce-fir-forests; 3.3 Scots pine and Black pine forests, with an azonal distribution related to sites where other alpine coniferous climax species can not grow because of limiting site conditions (dry and poor limestone, dolomite or serpentine substratum).

Table 2. continued.

FTBA name	Class description	Subtypes
4. Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest	Atlantic and nemoral acidophilous oakwoods, Atlantic ashwoods and dune forest.	4.1 Acidophilous oak dominated (*Quercus petraea*, *Quercus robur)* forest, lowland to submontane forests, on poor acid soils of Atlantic and nemoral Europe; 4.2 Atlantic ashwoods, lowland to upland UK and Ireland ash, predominantly found on neutral and alkaline, often moist soils. 4.3 Atlantic dune forest, dune forests and plantations of southwestern France and the western Iberian peninsula, mainly of *Pinus pinaster ssp. atlantica* mixed with *Quercus ilex* and *Quercus suber or* otherwise, *Pinus pinea and Populus tremula; Quercus robur* and *Betula pendula* coastal forest of the North sea shore.
5. Oak-hornbeam forest	Mixed forest dominated by oak (*Quercus* spec.) and hornbeam (*Carpinus betulus*) forest of lowland to submontane levels in sub-atlantic to continental climates. On wet soils *Quercus robur* is predominantly found, on dry soils *Quercus petraea*.	5.1 Central European pedunculate oak (*Quercus robur*)–hornbeam (*Carpinus betulus*) forests on groundwater influenced soils. 5.2 Central European sessil oak (*Quercus petraea*) – hornbeam (*Carpinus betulus*) forests on moderalety humid soils without groundwater or draught extemes. 5.3 Pannonian pedunculate oak - (*Quercus robur*)–hornbeam (*Carpinus betulus*) forests on clay soils with high water-tables but rarely flooded. 5.4 Pannonian field maple (*Acer campestre*)–hornbeam (*Carpinus betulus*) forests on warm soils, mainly loess or tschernosem-soils.
6. Lowland to submontane beech forest	Lowland to submontane forest (0 to 600-800 a.s.l.) dominated by *Fagus sylvatica* or its transitional hybrids with *Fagus orientalis*.	6.1 Lowland beech forests of S-Scandinavia and north central Europe 6.2 Atlantic and subatlantic lowland beech forests 6.3 Subatlantic to atlanto Mediterranean submontane beech forests 6.4 Central European submontane beech forests 6.5 Carpathian submontane beech forests 6.6 Illyrian submontane beech forests 6.7 Moesian submontane beech forests

Table 2. continued.

FTBA name	Class description	Subtypes
7. Montane beech forest	Montane to altimontane beech forest partly mixed with conifers (*Abies alba* and/or *Picea abies*) firs of the main European mountain ranges.	7.1 South-Western European montane beech forests 7.2 Central European montane beech forests 7.3 Apennine-Corsican montane beech forests 7.4 Illyrian montane beech forests 7.5 Carpathian montane beech forests 7.6 Moesian montane beech forests
8. Thermophilous deciduos forest	Deciduous or semideciduous forest of thermophilous species found mainly in submediterranean climate regions and supramediterranean altitudinal levels; under local microclimatic or edaphic conditions, the type is are found also far north in the Atlantic region, Pannonic and Continental regions.	8.1 White oak (*Quercus pubescens*) forests 8.2 Pyrenean oak (*Quercus pyrenaica*) forests 8.3 Turkey oak (*Quercus cerris*) and/or *Quercus frainetto* forests 8.4 *Quercus faginea* and *Quercus canariensis* Iberian forests 8.5 *Quercus trojana* forests 8.6 Sub-xeric oak (*Quercus macrolepis, Q. brachyphylla*) forests 8.8 Chestnut forests (*Castanea sativa*) 8.9 Other deciduous mixed woods (*Fraxinus* spp., *Ostrya carpinifolia*, *Carpinus orientalis*, *Acer* spp., *Tilia* spp., *Carpinus betulus*, *Aesculus hippocastnus*, *Juglans regia*)
9. Mediterranean and Macaronesian sclerophyllous forest	Broadleaved sclerophyllous forest of the Mediterranean and Macaronesian Regions.	1. Thermomediterranean (open) woodlands and dominated by *Olea europea* or *Ceratonia siliqua* or palms groves (*Phoenix theophrasti*, Crete). 2. Mesomediterranean oakwoods dominated by *Q. ilex*, *Q. rotundifolia*, *Q. suber, Q. calliprinos*, *Q. alnifolia* 3. Macaronesian laurel forest. 4. *Ilex aquifolium* relict woods (supra-Mediterranean level of Sardinia and Corsica and in Atlantic mountains of northwestern Spain).

Table 2. continued.

FTBA name	Class description	Subtypes
10. Mediterranean and Macaronesian coniferous forest	Meso to oromediterranean pine, fir and cypress dominated forest and Macaronesian pinewoods.	9.1. Mesomediterranean pinewoods, including old naturalized plantations, dominated by *P. pinaster, P. pinea, P. halepensis* or *P. brutia*. 2. Supra-to oromediterranean forests dominated by pines of the *Pinus nigra* group (*P. salzmannii, P. laricio, P. pallasiana, P. leucodermis*). 3. *Pinus canariensis* forest. 4. Meso- to Oromediterranean fir forest dominated by *Abies pinsapo* or *Abies cephalonica*. 5. *Juniperus thurifera* or *Cupressus sempervirens* dominated woodland.
11. Swamp forest	Coniferous or broadleaved more or less open and developed woodland on mires of the boreal, zones in Fennoscandia or on peaty soils throughout Europe.	11.1 Boreal Pine or spruce dominated mires 11.2 Alder dominated swamp and fen forest 11.3 Birch dominated swamp and fen forest
12. Floodplain forest	European fluvial and riparian forests.	12.1 Riparian *Alnus, Populus* and *Salix* forest, on low-lying areas and organic soils frequently flooded close to river channels; 12.2 Fluvial *Fraxinus, Quercus, Ulmus* forest on less frequently flooded mineral soils. 12.3 Mediterranean and Macaronesian riparian woodland (similar to 12.1-2 with additions of local species e.g. *Fraxinus angustifolia, Platanus orientalis, Nerium oleander, Tamarix spp., Flueggea tinctoria, Phoenix canariensis*).
13. Native plantations	The FTBA includes two main types of plantations: 1. Reforestation with native species: i.e. plantations of native coniferous species established inside or near the present or recent natural range of the species. In some cases these are old established plantations accompanied by seminatural undergrowth. 2. Highly artificial forestry plantations: conifers plantation within their broad biogeographical area of occurrence, but outside of the conditions described in point 1, characterized by intensive exploitation for commercial purpose. The decisive character is the artificial forest condition with a considerably modified accompanying floristic cortege.	13.1 *Pinus sylvestris* or *Pinus nigra* reforestation 13.2 *Picea abies* reforestation 13.3 *Abies alba* reforestation 13.4 Higly artificial coniferous forestry plantation

Table 2. continued.

FTBA name	Class description	Subtypes
14. Exotic plantations	Plantation of forest species non native to Europe or outside of their broad biogeographical region of occurrence established for the production of wood; some species (Robinia, Eucaliptus) species are able to regenerate naturally competing successfully with autoctonous forest species.	14.1 Robinia plantations 14.2 Eucaliptus plantations 14.3 *Poplar* clone plantations 14.4 Exotic *Quercus* sp. plantations 14.5 *Picea sitkensis* plantations 14.6 *Pinus radiata* plantations 14.7 *Pinus contorta* plantations 14.8 *Pseudotsuga menziesii* plantations …

slash and burn cultivation for their livelihood which was still used in eastern Finland until the 1920s. With the evolution of agricultural techniques, farmland was and still is increasingly abandoned in marginal areas and the land returned to forest naturally or by afforestation.

After the Second World War forests have been extensively affected by clear-cutting for commercial forestry, which has gradually changed the age distribution in the forest landscape (close and homogeneous forest cover) and has reduced the amount of natural forest remnants. During the 1990s there has been a shift towards an increased consideration for biodiversity conservation. Currently the FTBA covers about half the area of its natural range.

Key factors

Structural

Both wildfires and slash and burn cultivation resulted in a mosaic-like forest landscape with forest patches in various stages of succession in the boreal zone of Scandinavian countries (Parviainen 2000).

The structure of boreal forests is characterised by several features:

- Forests stands can reach an high mean age and due to gap-dynamics have wide age and diameter distribution (Kuuluvainen 1994). Especially in northern areas, where there is no human intervention to the landscape it is possible to find large continuous forest areas with high mean age. Pine forest stands reach mean ages up to 300 years.
- Old trees and deadwood varies according to tree age; the volume of deadwood is lower than in many other more productive forest types but the proportion out of the total biomass is still high.
- Usually forests contain several age classes from young to old (Huse 1965).
- Due to adverse climatic and site conditions natural reproduction of spruce can be irregular, therefore many high-altitude forests have age structures in the form of a damped wave with certain dominating age classes.
- In pine forests it can be observed a considerable variation in the size of areas affected by fire and in the age distribution within the landscape.
- In spruce forests there is a rather clear distinction of succession stages.

Compositional

Even if the structure of boreal forest is similar throughout the region, differences in forest composition can be found according to a north-south gradient and soil characteristics.

The main forest tree building are either *Picea abies* (dominant in mesic to fertile and moist site types) and *Pinus sylvestris*. Only in the northernmost areas the forest is dominated by *Picea obovata* and towards north-east *Abies sibirica* and *Larix* sp. enter as dominating species. Throughout the northernmost Fennoscandia, birch *Betula pendula* spp. *tortuosa* is the dominating species, which also forms the timberline. In some sites a mixture occurs consisting of *Picea abies, Pinus sylvestris, Betula pubescens* spp. *pubescens, Betula pubescens* spp *czerepanovii.*

Boreal forests of early- and mid- successional stages also contains deciduous trees: *Betula sp., Populus tremula, Alnus spp.* and *Sorbus aucuparia.*

In middle and south spruce forests the following stages and species can be identified (e.g. Angelstam 1998 a, b) after fire disturbance:

- vascular plants e.g. *Geranium bohemicum/lanuginosum* and pyrophilous insect species;
- young forest with ericaceous species;
- deciduous succession with increasing amounts of deadwood with age;

- old-ageing forest with pine;
- old-growth forest, very old trees and large amounts of deadwood.

In the south on most fertile sites occasionally isolated remnants of temperate broad-leaved species can be observed, namely *Tilia cordata, Acer platanoides, Ulmus glabra* and *Fraxinus excelsior*.

Natural disturbances
In boreal forests, key factors for maintaining biodiversity are fires, wind storms, insects, pathogens, floods, and animals including the moose and beaver (Kuuluvainen 2002). Nevertheless human actions have been at an accelerating rate replacing natural disturbances, forest management becoming the main driving force of forest dynamics. Such management has decreased the within-stand variation with relation to structure and composition as well as decreased the variation of age classes at the landscape scale.

In the western part of each Scandinavian country forests are more vulnerable to wind disturbances and affected by the Atlantic climate. By contrast fire becomes more important towards the east and south of each Scandinavian country.

Hemiboreal and nemoral Scots pine forest (FTBA 2)

Like for boreal forests anthropogenic disturbance has profoundly influenced the dynamic of natural hemiboreal forest. This forest zone is characterised by complex dynamics of deciduous and coniferous trees and has been intensively exploited during historic times. Grazing of domestic animals has been important for several centuries, but has recently ceased to be relevant. Natural fire regimes were amplified, beginning about 1000 years ago and then suppressed during the last 150 years. In general the deciduous elements have decreased and *Picea abies* has increased in importance, driven by interactions between anthropogenic disturbance and climate change.

Original hemiboreal forests with large conifers and southern deciduous trees occur nowadays almost only in reserves such as the representative and famous Bia³owieza National Park in eastern Poland. Currently the FTBA covers approx. the 38% of the area of its natural range.

Key factors

Structural
There is considerable debate about the natural structure of hemiboreal forest. Actual forests have a more closed cover than in the recent past owing to fire suppression, cessation of grazing and increase in abundance of *Picea abies*. The amount and types of deadwood changes with succession. Old-growth forests in the natural sense are not common.

Nemoral *Pinus sylvestris* forests are typically light flooded high forest, pure or mixed with *Quercus sp*.

Compositional
Pinus sylvestris and *Picea abies* are dominant forest tree building species in the hemiboreal forest, but are associated with a mixture of deciduous species, in early- to mid- successional stages, such as *Betula spp, Populus tremula, Alnus, Sorbus aucuparia*. The dominance of *Picea* increases with age.

On most fertile sites isolated remnants from a previously wider distribution range of temperate broad-leaved tree species, such as *Tilia cordata*, *Acer platanoides*, *Ulmus glabra*, *Fraxinus excelsior* and *Quercus robur* can be found.

The dynamic of hemiboreal forest after fire disturbance is similar to that described for boreal forests.

Natural disturbances

Key factors maintaining forest biodiversity are a mixture of natural and cultural disturbances, strong effects of wind and seasonal waterlogging in the west and fire of increasing importance in drier sites to the east.

Alpine coniferous forest (FTBA 3)

Forest history

Thousands of years of human settlement in the Alpine Region have modified the spatial distribution and appearance of Alpine vegetation; intensive Alpine pasturing, notably, produced the clearing forest through fires; this has influenced forest stand structure but also the spatial distribution of the FTBA by lowering the upper timberline by 200–400 m. Larch was particularly favoured by fire, grazing and clear-cutting. According to pollen analysis (Kral 1979) it is presumed that the current range and proportion of larch is much larger than it was originally. The FTBA currently covers approximately the 27% of its natural range.

Alpine coniferous forest have been traditionally managed as even-aged high forest for timber production. Nowadays, only montane spruce and mixed spruce-silver fir forests are productive forest, while the rest of subtypes is managed for protective functions, applying low intensive silvicultural practices.

Key factors

Structural

Three main subtypes can be distinguished in Alpine coniferous forests:

These three subtypes have substantial differences in forest structure which can be summarized in:

1. Variability of forest cover and physiognomy: subalpine larch-stone pine forests and spruce forests and Central alpine scots pine forest are open forests. Tree canopies are uneven and stand density is relatively low. At the upper timberline of the central and eastern parts of the Alps, the *Larix decidua* and/or *Pinus cembra* – forests show a smooth transition to shrub vegetation of dwarf pine *Pinus mugo ssp. mugo*. More open forests of *P. mugo ssp. uncinata* are mainly found in the western and central parts of the Alps. On the opposite, montane pure spruce forests are much closed forests, with high stem-density, often with single layer structures.
2. Differences in forest structure and natural regeneration: in subalpine forests, close to the upper timberline, the horizontal structure is clustered due to specific regeneration dynamics caused by harsh micro-site conditions. Only the most favourable micro-sites (patches of some square meters or less) can support seedlings. Dead wood is very important to the natural regeneration of subalpine forests, as seeds and seedlings are likely to find adequate growth conditions on decaying wood. Because of permanent small-scale disturbances and regeneration in the subalpine zone, the pattern of development phases is

on a much finer scale than in montane forests. These are instead often affected by large windfalls or snowbreaks due to their even-aged structure, being a result of the silvicultural treatments. As a result montane forests show a more homogeneous horizontal structure and regeneration rich in individuals. Further differences in stand structure are caused by the proportions of shade tolerant silver fir, occuring partly in the montane subtypes, but not in the subalpine ones.

Compositional
Besides the differences in forest tree building species amongst the three subtypes, in the subalpine and montane mixed spruce forest of the Alps a clear gradient in the proportions of silver fir to spruce exists, due to different species migration patterns and forest history. Spruce dominates in the eastern Alps and silver fir in the western. Therefore, in the south-western Alps near to the Mediterranean Sea pure fir-forests occur, even close to the upper timberline.

In general, the proportion of spruce has increased mainly as a result of silvicultural treatment (clear-cutting and replanting of spruce) and as a result of decreased competitiveness of silver fir resulting from browsing by roe and red deer. In many regions, silver fir dominated sub-types have nearly disappeared and species diversity declined into pure spruce stands. Broad-leaved species rarely occur in these forests. Grazing, pasturing and clear-cutting have also increased the competitive ability of *Larix decidua*, which now has extended beyond its potential natural contribution in the area.

Natural disturbances
Barkbeetle outbreaks often fallowing windfall and snowbreaks (*Ips typographus*, *I. aminatus*, *Pythiogenes chalcographus*).

Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest (FTBA 4)

Subtypes 4.1 and 4.2

Forest history
The Atlantic forests have been profoundly modified by man throughout the past 10 centuries due to the old colonization of the region, one of the most densely populated in Europe. Estimated forest loss in Europe of Atlantic and nemoral oakwoods and ashwoods is strikingly high, 94.8% and 97.8% respectively; this is mainly due to the natural distribution of mesophyle oakwoods occurring on meso- and eutrophic soils proved to be the most interesting for agriculture. Human impact involved intensive wood exploitation as well; Atlantic oakwoods have been managed since from Middle Ages for many purposes: fuelwood, charcoal and timber production (*Q. petraea* and *Q. robur* were used for shipbuilding) and grazing (through pollarding, shreddeding oakwoods provided for leaves and acorns for livestock).

Structural
Human influence has shaped the actual structure of the acidophilous oakwoods and ashwoods, producing the differentiation of four main silvicultural types: high forest, coppice with standards, mixed coppice/high forest, coppice. Of purely cultural origin are wood-pastures, pollareded meadows with a savanna-like structure of ‘cultural’ origin of great importance for biodiversity conservation (see Humprey et al. in this volume; Rackam 1998). Forest continuity is also recognized as a critical factor for biodiversity conservation in

England (Bailey et al. 1998). England ancient woodland – sites that are believed to have remained wooded continuously since AD 1600 – are semi-natural communities retaining a wide range of original woodland species and habitat and are generally far more diverse in species than recent woods (most of them plantations with non site-native conifers).

According to French and UK NFIs data, old growth acidophilous oakwoods and ashwoods are also found in the class.

Compositional

The edaphic factor combined with the degree to which woodland were and still are subjected to coppicing and grazing affect the actual species composition of the subtypes and the distribution pattern of characteristic species assemblages.

Acidophilous oakwoods growing on poor acidic soils – with frequent degradation of the upper soil layer – are typically species poor communities. The forest is characteristically dominated by combination of Q. *petraea*, *Q. robu*r and *Betula* sp. Atlantic ashwoods allow the development of a more varied forest community harbouring a great diversity of plant species as well as a rich invertebrate fauna and lichen flora. Forest building species vary according to local site condition with *Fraxinus excelsior* mixed with different other mesophytic species.

Natural disturbances

The forest regeneration process can be locally severely limited by grazing (e.g. in the uplands of the UK) and acorn and seedling predators (e.g. deer, grey squirrel). Flooding in riparian areas is a common disturbance in ashwoods.

Subtype 4.3

Structural key factors

Atlantic dune forest is typically managed as even-aged high forest; structural variability within this subtype (vertical structure, canopy closure) is therefore chiefly influenced by age class distribution. Topography affects also soil condition and micro-climate variation at fine scale, creating heterogeneity in stand composition.

Oak-hornbeam forest (FTBA 5)

Forest history

Mixed oak-hornbeam forests occur naturally in sites where beech forests can not grow as a result of special local climatic conditions (frequent frost periods in early spring, bassins with temperature inversion) as well as macroclimatic areas with too low precipitation rates. The natural range of the FTBA show clearly a main centre in Central-Eastern Europe. The potential area in central and western Europe is tied to riverside flatlands with soils not suitable for beech and/or planar-colline lowlands of the great basins. Most of the natural oak-hornbeam forests have been transformed to very productive agricultural land; nowadays the FTBA cover just the 15% of its natural range.

Natural oak-hornbeam forests have been deeply modified by coppice management. In recent times, many stands have been or are being transformed into high forests, with the silvicultural aim of valuable oak timber production. Presently, only special sites are considered to be quite close to natural original woodlands and are protected in strict forest reserves.

Key factors

Mixed oak-hornbeam high forests are typically managed by intensive silviculture with long-term rotation. Forests show a characteristic two layered structure. Oak builds up the main layer and hornbeam is used as an auxiliary species in the dominated part of the stand. Most of the silvicultural activities during the rotation period of these stands aim to maintain the lead of oak against hornbeam. Regeneration of oak is mostly achieved through planting, as natural regeneration has been supressed as a result of browsing, grazing and competition from herbaceous vegetation.

Strong evidence exists that the structure of naturally developed stands is rather different to stands managed even by close to nature silvicultural practices. However there is little knowledge on how structures and stand dynamics under natural conditions work, as remainders of virgin forests are absent. Stands protected in existing strict forest reserves show a change of species composition from oak to hornbeam: stands are more ore less dominated by hornbeam, with oak as an subdominate tree-species (Mayer and Tichy 1979). This is partly due to the cessation of silvicultural treatment, but also to the pattern of small scale gaps created by natural disturbances seems to favour hornbeam leadership.

In the areas where coppicing had a long tradition, especially in the transition zone of beech or hornbeam dominated vegetation belts, the percentage of hornbeam and oak have increased due to their ability of sprouting. Through coppicing treelayer and understorey have been modified in many cases very effectively. In coppices the short-living shade-tolerant species hornbeam forms a high organized life community with the long-living light depending oak.

Lowland to submontane beech forest and Montane beech forest (FTBAs 6–7)

Beech, an heavy-seeded species with slow migration, immigrated very late to Northern Europe from its glacial refugia – in some areas as long as recent 1500–500 B.C. – whereas South Eastern Europe communities are much older, being established from 6 to 8 thousands years ago. In the NW, the arrival of beech interfered with the ever-increasing impacts caused by the presence of humans, so it has not been able to settle on all of its potential sites as a result of anthropogenic forest utilisation; in other cases, conversely, its spread was supported by human activities.

Throughout Europe, the natural distribution of beech forest was altered, since the earliest historical times, due to agricultural clearance, causing habitat destruction and fragmentation. Many potential beech forests were transformed to spruce plantations, frequently after agricultural use, or to oak, oak-hornbeam forests. Reforestation programmes in mountain areas supported the spread of spruce and fir pushing back beech.

Even where beech forest cover has been retained, species composition and structure has been changed dramatically by commercial forestry. Mountain beech forests were also intensively coppiced for firewood and charcoal production (e.g. in the Apennines and partly in the Alps). Most of these stands were turned to high forest in the 20th century.

Today, *Fagus* dominated forest retains approximately one third of its natural range.

Key factors

Structural and functional

Past and actual forest management has influenced considerably the actual variability of European beech forest encompassing a wide range of structural types: high forest (including also uneven-aged stands in mountain beech forest), coppice with standards, mixed coppice/

high forest, coppice, wood-pastures; a category of special interest for understanding how natural beech forests might have looked and their dynamics are the fragments of beech forest scattered throughout Europe, left unmanaged for a long time, which are considered to be quite close to natural original woodlands (see more on NatMan project website www.flec.kvl.dk/natman).

Structural types are primary determinant of biodiversity differentiation. High forest, the most common structural type, is traditionally managed as even aged forest with a 120–150 years rotation time. Beech is maintained in the optimal phase of its biological cycle, estimated to be 200–250 years long. The widely applied management systems (clear-fell or shelterwood with natural or artificial regeneration) use large, compared to natural patch size, compartments; as a result forest landscape is drastically modified, showing a coarse patch pattern especially in lowland and submontane forest types. In mountain areas orographic factors (direction of mountain ranges, exposition, elevation and steep altitudinal gradients) create more environmental variation in ecological conditions resulting in a higher spatial diversity of forest types and associated species at the landscape scale.

Typical alterations of natural forest biodiversity found in commercial high forests are:

i) close canopy closure and monolayered vertical structure; beech, being naturally an extremely shade tolerant species, casts dense shadow especially when it grows in large uniform stands. The understorey is sparse and in most sub-types shrub species are absent. In montane beech types the canopy layering is less uniform, since fir attains higher maximum height than beech.
ii) several patch types – the second half of natural forest cycle, early successional phases – are lacking, and new patch types – large uniform tracks of beech regeneration – have been introduced.
iii) decrease or elimination of admixing tree species and different forms of dead wood (snags and logs of different sizes and decay phases). Many forest dwelling organisms adapted to these structural features are hence extremely rare or completely missing. Montane beech forest are richer in dead wood, found large quantities, size and decay phases.

Compared to commercial forests, traditional woodlands like wood pastures, coppice with standards, or selection in small farm forests, scattered throughout Europe preserved much more of the original forest structures and served as refugia for many forest dependent species.

Under natural conditions forest development is governed by fine-scale gap dynamics.

In lowland and submontane areas, the death of one or a few old beech trees opens the space for advance regeneration, or newly established young beech trees, and also for other species. The dominance of this disturbance results in a fine-scale mosaic of forest developmental phases as it was described in virgin beech forest remnants by several authors (e.g. Korpel 1995; Prusa 1985).

In the mountains, the effects of snow and ice are more pronounced on stand development. Also, because of topography, when individual trees fall they often hit others like dominos, so somewhat larger elongated gaps can develop. Another important character of stand dynamics in mixed beech-fir forests originates from the different lifespan of these species. Silver fir (*Abies alba*) can live almost twice as long as beech (*Fagus sylvatica*). This means that beech can go through two developmental cycles, while fir accomplishes only one. As a result the proportion of the two main components of the tree layer changes dramatically through time in any one place, but the amount of overall standing crop is relatively stable (e.g. Boncina 1999).

Beech is also rather sensitive to the parasitic fungus Fomes fomentarius, which often is an important factor in gap-formation. In the submontane zone, ice deposition on beech can cause severe damage to the canopies, causing occasional opening or, rarely, uprooting of dominant trees on large areas. This leads to successional changes during which several otherwise suppressed species play important roles.

Compositional

Lowland and submontane beech forest are almost pure due to natural factors – close to the environmental *optimum* for the beech- and for the species has been favoured by forest management.

Other tree species can play a role where soils are too shallow and poor (*Quercus spp., Acer pseudoplatanus*), or locally dry or richer in nutrients (*Quercus spp., Acer spp. Fraxinus excelsior*), and in successional phases (*Acer pseudoplatanus, Ulmus glabra, Betula spp., Fraxinus excelsior*).

In montane beech forest *Fagus* is partly mixed with spruce and fir. Another important compositional feature of montane beech forests is the level of endemism in mountain floras (and faunas) much higher than in lowland and submontane regions.

The distinction of 14 beech forest subtypes is tied to the high bio-geographical differentiation within the two FTBAs; based on geographic differences, the subtypes show characteristic floristic differences, most pronounced in the composition of the herb layer. Some shrubs and trees also show distinct geographic patterns.

Mediterranean and Macaronesian forest types (FTBAs 8, 9, 10)

Forest history

Mediterranean forest have played a major role in the development of civilisations since the Bronze Age (e.g. Crete, Cyprus, Greek, Roman, Muslim). Thousand of years of human influence have drastically changed the distribution of natural forests which covered most of the Region following the last Ice Age. The need for arable land or pastures produced a gradation of clearing of forest cover: lowland forests were extensively deforested, hill forests were fragmented in a patchwork of fields, pastures and other tree crops (mainly olive, citrus and vine) and only the most inaccessible mountain areas remained relatively wooded.

A similar process took also place in Macaronesian archipelagos starting from the Portuguese colonization in the 15th century, leading to a sharp decline in the area originally covered by laurel forest.

Mediterranean forests cover presently a percentage of their natural range, which varies significantly with the inaccessibility and marginality of forest sites: broadleaved sclerophyllous forest (20–22%), thermophilous deciduous forest (23–28%), Juniper and cypress dominated woodland (39%), meso- oro mediterranean fir and pinewoods (approx. 50%).

Woods left from agricultural clearing were intensively harvested in the need to support farming. Oaks and chestnut woodlands (the latter established through plantations in Europe since the Roman times) were coppiced to yield a large number of assortments for agriculture (e.g. vineyards poles), but also fodder, leaves and acorns to feed livestock or rural populations (chestnut groves). Cork oak and holm oak woodlands were converted into agro-forestry systems known as “dehesas” (in Spain) or “montados” (in Portugal) and related wooded pastures in Sardinia, Greece. As long as wood was the main source of energy oakwoods were intensively logged to provide fuel wood and charcoal and juniper woodland as well. Coniferous forest were also intensively exploited both for timber production (fir, pinewoods) but also for resin extraction, pines and honey production and grazing.

Despite their importance as economic resources, Mediterranean forest have been over-exploited throughout Europe and still are in some regions. Intensive logging combined with recurrent fire and overgrazing has produced disastrous effects in terms of land degradation, particularly severe in regions with arid climate and soils prone to erosion, leading even to desertification.

All these processes had determined and still influence biodiversity condition in Mediterranean and Macaronesian forests. The variation of biodiversity within and between the FTBAs can be related to changes in one or more of the key factors described below.

Structural

1. *Variability of forest cover and physiognomy*: tree growth and forest cover range from low-growing and fragmentary broadleaved woodlands found mostly in xerophile/degraded sites, to open pioneer pinewoods, to scattered and full-sized trees in wooded pastures, to well developed and closed forests.

2. *Differentiation of forest structural types and development of old-growth trees/stands*: rather simplified forest structures shaped by traditional silvicultural systems are the most common in broadleaved woodlands: coppice, coppice with standards, mixed coppice/high forest. The typical physiognomy of Mediterranean and Macaronesian coniferous forest is even aged high forest.

Of purely cultural origin are chestnut-groves, today largely replaced by coppice-woods or left unmanaged. Some of them host ancient trees and are important habitats for species related to the specific qualities of old-growth forest (e.g. hole nesting birds, invertebrates living in thick bark, etc.).

Other remains of cultural woodland are *montado*s, *dehesas* and the all the cultural savannas of broadleaved and coniferous species scattered throughout Sardinia, Balearic islands, Greece and Crete cultivated through pollarding or coppicing. These communities, once were common Mediterranean landscapes, represent today vanishing important hot-spots of biodiversity (e.g. dehesas and montados home endangered species such as the Iberian lynx and black vulture), especially where grassland and ancient trees are present (Rackham 1998).

Further condition of structural differentiation within thermophilous and sclerophyllous woodland is the abandonment of forest cultivation; high forest-like structures develop naturally with the cessation of felling in coppices, process that is under particular circumstances actively managed through the conversion to high forest; a sort of infilled structure can be observed in unmanaged *Castanea sativa* groves with old big tree surrounded and/or overtopped by a crowd of local pioneer forest species.

Besides chestnut groves and savanna-like oak woods there are other pockets of Mediterranean forest stands developed to a phase of advanced maturity or even senescence scattered throughout Europe. In the case of broadleaved woodlands these originate either by the abandonment of coppices and from the maintenance of ancient chestnut-groves. Old-growth coniferous forests are also found in steep and isolated slopes of Mediterranean ranges where a combination of low human population and inaccessibility has kept some forest stands relatively intact and untreated for a long time.

Compositional

1. *Characteristic floristic differentiation of the Mediterranean region* (submediterranean, west-mediterranean, east-mediterranean clearly visible in the distribution of the species in *Quercus* or of *Pinus nigra* group) related both to the great climatic and physical variation within the Region (mountains, peninsulas, islands) and to paleobiogeographical reasons; in addition the complexity and compartmentalization of Mediterranean reliefs the complexity and compartmentalization of Mediterranean reliefs or Macaronesian archipelagos has favoured isolation and therefore

endemism. According to Ozenda (1994) the most important centres of endemism in the Mediterranean Region are found in Pyrenees, Sierra Nevada, Corse and Crete.

2. *Variation of (residual) richness forest communities as to forest building species*; most mediterranean forests are dominated by assemblages of one-two dominant native or naturalized tree species, accompanied by secondary species. Anthropogenic exploitation has modified the composition of original forests, leading in most cases to the elimination of species without a commercial interest or with poor resprouting capacity or, conversely, the development of new forest communities (chestnut forest, old established naturalized plantations of mesomediterranean pines). Some mixed broadleaved and coniferous forest types found in the FTBAs derive, for instance, by the "enrichment" of broadleaved coppices with conifers (mainly mediterranean *Pinus* sp. and *Cupressaceae*).

Natural disturbances
Fire: human induced wildfires can significantly influence the dynamics of forest regeneration favouring the persistance of resprouter species in broadleaved forest or the regeneration of pinewoods. Recurrent fires and overgrazing play a major role in the development of land degradation processes.

Biological disturbances: the forest regeneration process can be locally severely limited by acorn and seedling predators. Acorn-feeding mammals (wild boar, hares, woodmice) and livestock (goats and ibex) by consuming most acorns and seedlings constrain regeration via sexual reproduction (Gómez et al. 2003). Under these conditions, quite common in broadleaved forests, the only way that forest can regenerate is by resprouting.

3. Conclusions

In the last years both the European Ministers responsible for forests at the Vienna Conference (April 2003) as well as the European Ministers responsible for environment at the 5th Environment for Europe Ministerial Conference (Kiev, May 2003), endorsed a Framework for Co-operation between the Ministerial Conference on the Protection of Forests in Europe (MCPFE) and the Environment for Europe/Pan European Biological and Landscape Diversity Strategy (EfE/PEBLDS): one of the joint activities is the clarification between the Ecosystem Approach (EA) and sustainable forest management (SFM), building on: the work achieved so far by the MCPFE on SFM, the decisions taken by the Convention on Biological Diversity (CBD, Decision VII/11– 6th Conference of the Parties to the CBD, Expanded Work Programme on forest biological diversity, 2002) and the United Nations Forum on Forest (UNFF) (2003, third session, Resolution 3/4, para 8).

SFM has been defined in Resolution H1: *General Guidelines for Sustainable Management of Forests in Europe* of the Helsinki Conference (1993) and developed through all other decisions of the Ministerial Conferences (Strasburg 1990, Helsinki 1993, Lisbon, 1998 and Vienna, 2003: "*Sustainable Forest Management is the stewardship and use of forests and forest lands in a way, and at a rate, that maintains their biodiversity, productivity, regeneration capacity, vitality and their potential to fulfil, now and in the future, relevant ecological, economic and social functions, at local, national, and global levels, and that does not cause damage to other ecosystems*". The definition addressed the concern about biodiversity conservation issues, and the MCPFE process decided to develop criteria, indicators and forest management certification programmes, including protected areas: already at the Helsinki Conference (1993) the Ministers adopted general guidelines for the conservation of biodiversity of European forests (Resolution H 2).

In Vienna a set of criteria and improved indicators for sustainable forest management has been adopted, included Criterion 4: Maintenance, Conservation and Appropriate Enhancement of Biological Diversity in Forest Ecosystems: the list includes at least 6 indicators that specify the need to report data assessed by forest type. Even the TBFRA programme (Forest Resources Assessment) is committed to structure the information accordingly. Periodically measured indicators show a direction of change within each criterion.

The proposed nomenclature and preliminary analysis of the forest types here discussed could help for a correct and homogeneous interpretation of the forest biodiversity assessment issues at Pan European level.

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European Forest Types for Biodiversity Assessment – A Qualitative Approach

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Abstract

The BEAR project (Biodiversity Evaluation Tools for European Forests) proposed 33 forest types for biodiversity assessment (FTBAs), many of which could be identified on the general map of natural vegetation of Europe (Bohn et al. 2000). Eight of these forest types were purely of cultural origin yet types such as hedgerow and coppice also have a significant biodiversity value. We propose a simpler qualitative classification, which is a more appropriate tool for monitoring forest biodiversity. It comprises 14 European forest types and was obtained by combining some of the BEAR FTBAs. For example, five boreal BEAR FTBAs become one new boreal category and Mediterranean riverine woodlands are merged with flood plain forests.

We use floodplain forest as an example of the revised approach. This covers: species composition and typical forest structure; the relationship of the category to FTBAs and other European classification systems; a distribution map and description of representation along nutrient rich-poor gradients and moisture gradients. Key factors that distinguish the types are described, with emphasis on historical background and development, disturbance regimes and current status with regard to management and threats. Examples of typical locations are also presented.

This essentially qualitative approach to forest type classification has the advantage that it takes into account existing ecological knowledge and highlights particular communities such as swamp forests that are not widespread but have a high biodiversity value.

Keywords: floodplain forest, key factors, forest type maps.

Marco Marchetti (ed.)
Monitoring and Indicators of Forest Biodiversity in Europe – From Ideas to Operationality
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Introduction

There are two main arguments for classifying European forest vegetation to help with biodiversity assessment. Firstly there are scientific insights to be gained from seeking structure among the great variety of forest vegetation that exists in Europe. Certain groups of tree species tend to co-occur at stand-scale and have formed the basis for phytosociological classifications and national classifications of stand types (Peterken 1981; Påhlsson 1994). Several hundred to thousands of stand types could be recognised at the European scale so it is practical to group these into higher order categories that can be mapped with ease at the European scale. These higher order forest types, can be at a scale that reflects broad-scale continental, primarily climatic variation across Europe but will then be less sensitive to finer-scale variation such as soils and hydrology. They will differ from classical biomes (Holdridge 1947) in that they retain taxonomic variation and are not mere plant functional types. Such higher order types also have certain operational advantages in that 10–20 European types can be monitored more easily than 500–2000. 10–20 types are also the number that could potentially be resolved using advanced remote sensing technology. A potential problem with defining a limited number of forest types for biodiversity assessment is that diversity itself is concealed in the process of simplification. This problem can at least be minimised by constructing a clear hierarchy of types in the classification.

Classification must act on a data source. This is self-evident, but data sources used in the past have tended to be mixtures of potential and actual vegetation that can cause some confusion. Biome maps are expressions of potential vegetation, showing the situation in the absence of human impact past or present (Prentice et al. 1992). Bohn et al. (2000) presented a map of 699 potential European vegetation types but included some communities that were most likely the product of former human activities such as burning and grazing of domestic animals. Examples are some heathland and grassland communities (Figure 1). Päivinen et al. (2002) produced a map of European forest cover based on remote sensing and national forest inventories (Figure 2). This is clearly actual vegetation, but the techniques employed only permitted resolution of deciduous from coniferous forest, which is too coarse a resolution for monitoring biological change of any type except possibly patterns of carbon sequestration. CORINE a land cover classification system based on remote sensing also includes actual vegetation but among its 44 land cover types there are only 3 true forest types. In between the extremes of potential and actual forest vegetation classifications lie the European Nature Information System (EUNIS) with 377 forest types. The Ministerial Conference on the Protection of Forests in Europe (MCPFE) has set in motion a formal process to improve forest protection measures and has identified the need for a practical classification of European forest types that is more detailed than CORINE but more operational than EUNIS. In the EU Habitats Directive, 51 forest types are selected for their importance to nature conservation.

The BEAR concerted action aimed to identify indicators for monitoring and evaluation of forest biodiversity in Europe. These indicators were associated with 33 rather heterogeneous forest types (Larsson et al. 2001). This classification differed from previous divisions of forest types in that it was not primarily based on floristic composition. Instead the classification aimed to be process oriented and was based on so-called key factors. These could be structural factors such as dense, closed forest canopy; regeneration patterns; compositional factors such as dominance of *Fagus sylvatica* or high diversity of vascular plants; historical factors such as early association with human activity; or functional factors such as periodic flooding of the habitat. The classification system attempted to be scale independent and be applicable at regional, landscape and stand scales. The 33 forest types identified for biodiversity assessment ranged from mixed oak forest, a potential forest type

Figure 1. Map of 699 potential European vegetation types (after Bohn et al. 2000).

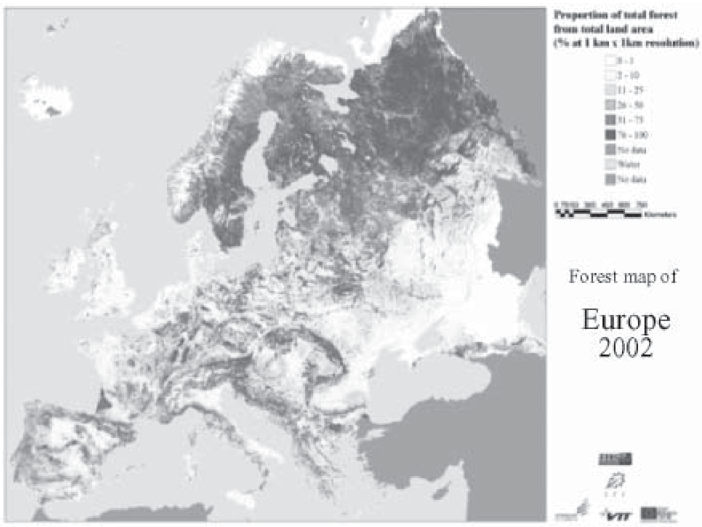

Figure 2. Map of European forest cover based on remote sensing and forest inventory data (after Päivinen et al. 2002). The forest area shown comprises over 1 000 000 000 ha representing 27% of the entire global forest area.

covering large parts of central and western Europe, to hedgerows and Chestnut coppice of purely anthropogenic origin and with rather restricted distributions.

The BEAR system provoked reactions from interested scientists. These indicated that the forest types were too heterogeneous for practical use and that the types included a mixture of actual and potential forest types. In this paper we aim to increase the operationality of the

Table 1. The new BEAR forest types and the number of EUNIS forest types that they contain.

BEAR type	Number of EUNIS types
Boreal forest	18
Hemiboreal and nemoral Scots pine forest	17
Alpine coniferous forest	46
Atlantic and nemoral oakwoods, Atlantic ashwoods and dune forest	46
Oak-hornbeam forest	26
Lowland to submontane beech forest	6
Montane beech forest	21
Thermophilous deciduos forest	52
Mediterranean and Macaronesian sclerophyllous forest	42
Mediterranean and Macaronesian coniferous forest	18
Swamp forest	24
Floodplain forest	37
Native plantations	11
Exotic plantations	13

BEAR types by making the categories more homogeneous and practical to use. We propose a merger of several of the previous BEAR types to yield 14 new categories (Table 1). A number of the 377 EUNIS forest types can be assigned to each BEAR type as shown in Table 1, illustrating the hierarchic nature of the classification.

We describe here characteristics of floodplain forest type to illustrate characteristic features of the new system.

Floodplain forest

Characteristics: composition and structure

European floodplains contain species-rich and productive forest ecosystems whose composition and structure depend upon the frequency of flooding (Penka et al. 1985; Schnitzler 1994a; Peterken 1996; Klimo and Hager 2001). Low-lying areas on organic soils that are frequently flooded and close to main river channels, are characteristically dominated by species of *Alnus, Populus* and *Salix*. Less frequently flooded mineral soils can support rich mixtures of *Quercus robur, Fraxinus excelsior, Prunus padus, Ulmus, Acer*. The common edaphic conditions and disturbance agencies throughout European floodplains contribute to comparable taxonomic composition with some local species additions (e.g. *Fraxinus angustifolia* in Portugal, Spain and Italy) or substitutions (e.g. *Lonicera caprifolium* for *L. xylostemum* in Italy). *Nerium oleander, Tamarix* and *Flueggea tinctoria* are important Mediterranean taxa.

Forests are often dense, productive and multi-layered with extensive growth of lianes (e.g. *Clematis vitalba, Hedera helix, Loranthus europraeus, Viscum album* and *Vitis sylvestris*). Shrub and herbaceous layers can be particularly rich in species. Characteristic herbaceous species include *Anemone nemorosa, Ajuga reptans, Brachypodium sylvaticum, Carex spp., Circaea lutetiana, Glechoma hederacea, Impatiens parviflora, Lysimachia nummularia,* *Paris quadrifolia, Phalaris arundinacea, Polygonatum multiflorum, Stachys sylvatica, Urtica dioica* and *Viola reichenbachiana.*

Relationship to previous BEAR and other systems

Floodplain forest as presented here comprises both the alluvial and riverine floodplain forests (type 23) and the Mediterranean and Macaronesian riverine woodlands and galley forests (type 24) of the BEAR Forest Types for Biodiversity Assessment (FTBA). Floodplain forest includes types G1.1 – G1.3 of the EUNIS III level forest types: riparian *Salix, Alnus* and *Betula* woodland, fluvial *Fraxinus, Alnus* and *Quercus, Ulmus, Fraxinus* woodland and Mediterranean *Populus, Fraxinus, Ulmus* and related riparian woodland and is part of the EUNIS II level type G1: broadleaved deciduous woodland. Floodplain forest matches type FT1N.1 in the tentative Pan-European forest type scheme: fluvial and riparian woodland.

Key factors

The fundamental processes of floodplain forests are dependent on river dynamics and consequent flooding frequencies. Flooding frequencies can vary from annual to intervals of over one century. Flood duration has a significant biological impact and the influence of flooding on soil properties is also a key factor. Floodplain soils tend to be rich in clays and silts and have a moderate to alkaline pH. The distance from the active channel is an important site variable. Former floodplains can turn into river terraces that may not flood any more, but retain the fertile soils characteristic of active floodplains.

Palaeoecological techniques can help indicate the baseline conditions prior to extensive anthropogenic modification. A study of floodplain forest history of the River Trent, N.E. England indicated that the forest vegetation between 5000 and 8000 years ago was similar to that of the Rhine today and indeed may have been part of the same flood-prone ecosystem prior to inundation of the North Sea about 8500 years ago (Brayshay and Dinnin 1999). Dead and decaying wood seems to have been particularly abundant in this forest type and the fossil beetle faunas from the Trent site was particularly diverse (Brayshay and Dinnin 1999). Particularly high species diversity of vascular plants, pteridophytes, bryophytes and insects is a key characteristic of floodplain forests. The woody flora alone, on sectors of the Rhine, comprises 21 families, 34 genera and 50 species, which is remarkable for the latitude. A comparable upland forest area in south Germany comprised 12 families, 18 genera and 21 woody species (Schnitzler 1994a). A contributory factor to high biodiversity in Boreal and Mediterranean regions is that these sites tend to be protected from frequent and intensive burning that tends to generate a specialised fire-adapted flora and fauna.

Floodplain forests occupy only a small fraction of their original and potential habitat. Areas originally covered by floodplain forests have were attractive for short-rotation coppice (Klimo and Hager 2001) and agricultural exploitation from at least the Middle Ages onwards. The first use was as productive grazing land for cattle and pigs. Regular mowing and use as hay meadows, represents an intensification of this process and finally crop cultivation is widespread on the drained, organic-rich soils (Penka et al. 1991). The current conservation status of floodplain forests is more influenced by the recent history of land reclamation, flood control through dyke construction and straightening or canalisation of rivers for improved navigation and regional drainage than by agricultural and forest history. Areal loss is extensive in parts of Britain and Ireland with some notable exceptions such as the Geeragh, S.W. Ireland (Kelly and Iremonger 1997). The Rhine is a typical case study for central

Figure 3. Distribution of European floodplain forests (after Bohn et al. 2000).

Europe (Schnitzler 1994b). An original estimated cover of 200 000 ha was reduced to 120 000 ha by the end of the 18th century. Straightening of the Rhine, which began in 1850, caused the loss of a further 50 000 ha. Canalisation operations during 1950–1970 reduced the area to less than 15 000 ha of which about 12 000 ha remain.

Examples

Danube, Rhine, Rhone, Loire, Tessin, Po, Shannon, Oder, Morava (Figure 3).

Concluding remarks

The qualitative approach to forest type classification outlined here utilises the skill and experience of the scientist and creates a practical division of European forest types that can form the basis for various types of monitoring programmes. The qualitative approach gives due regard to forest types of limited areal coverage but considerable biodiversity value such as floodplain forest. The key factor concept goes beyond mere compositional considerations and incorporates dynamic processes that can be characteristic of large forest areas with somewhat variable composition and at differing successional stages. Forest areas with similar histories can also be given consideration in the classification. The key factors are therefore useful tools in scaling up stand-scale classifications to the landscape/continental scale. Several of these key factors are weakly covered by classifications that are purely quantitative or only consider florisitic composition. However, as the forest types themselves are partially theoretical abstractions, a complementary quantitative approach is required if the extent of the forest types themselves is to be monitored.

To force the rich variety found in European forests into a mere 14 categories may appear to be a brutal over-simplification that works against the concept of biodiversity itself. The main point of this exercise however is to generate a practical number of types within which biodiversity indicators, such as dead wood, can be monitored. When scientific concepts are made operational through the political process, robust simplifications are inevitable (Failing and Gregory 2003). The scientific challenge is to select the least damaging simplifications.

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European Forest Types for Biodiversity Assessment – Quantitative Approaches

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Abstract

Future monitoring of European forests and their biodiversity is most logically based on quantitative surveys and it is practical if existing survey systems are modified for this purpose. We analyse quantitative data from two different types of survey that cover most of Europe: 1) National forest inventories and 2) ICP Forests Level 1 data. National forest inventory data were pooled at the scale of regional administrative units to generate a relatively homogeneous data set of percentage tree species composition. These data were classified using clustering routines to generate a set of major European forest types. The quantitative approach is considerably influenced by plantations of limited biodiversity and has difficulty resolving rare forest types of high diversity (e.g. floodplain forests), so a combination of approaches is desirable to faithfully describe European forest biodiversity.

Keywords: Forest type maps, national forest inventories, ICP Forests, classification.

1. Introduction

A sound classification of European forest types depends on a good data set, and in principal extensive databases on European forests exists. These have been compiled by, for example, UN-ECE/FAO (UN-ECE/FAO 2000), the European Environment Agency through its topic centres, EUROSTAT (EUROSTAT 1998), and by the European Forest Institute in the EEFR (EFI 2004a) and EFIDAS (EFI 2004b) databases. In addition, detailed forest data are collected in national forest inventories (NFIs) on a more or less regular basis in most European countries. However, no databases have yet been specifically used for quantitative mapping of European forest types. Therefore, the major aims of this paper are to assess the feasibility of forming databases for a forest type classification in Europe, performing classifications and assessing their value in the monitoring process of European forest biodiversity.

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In this paper we specifically develop an automated methodology for the classification of quantitative forest types based on actual tree species abundances. We present two classifications of forest types based on two different data sets. Our method could be used to generate different numbers of forest types dependent upon the defined degree of separation desired between categories. For reasons of operationality, we have set a rough target of between 10–20 types that can be practically displayed on a single map of Europe.

2. Data

2.1 National Forest Inventories

Data from national forest inventories (NFIs) were compiled. Contact was established with 24 European countries and relevant data were received in a variety of formats from 14 of these countries. All countries were requested to contribute data on tree species volume at a regional level. These were often administrative regions, e.g. "départements" in France and counties in the United Kingdom. Data at tree species level allows for classification of forest types as mixtures of species. Tree volume data were favoured rather than numbers of individuals or area as this was felt to give a better representation of the biological importance of the species on the landscape. Administrative regions are convenient units to map. However, all contributing countries could not meet even these specifications. Most countries had data available for dominant tree species only and two countries contributed with area data instead of volume data. Not all contributing countries provided data from administrative regions. Regions could also be forestry or state forest regions, e.g. in Switzerland and Poland.

Forest area is defined very differently in individual European countries (see Schuck et al. 2002). This is also the case in the NFIs that are part of this data set. Furthermore, data sampling methods vary hugely between the individual NFIs. The differences between NFIs in Europe have been outlined in a study on European forestry information (European Commission 1997), although practice has changed somewhat in several countries since this report was published.

The heterogeneity of the gathered NFI data called for harmonisation of taxa and normalisation of data. A common taxon list for the 14 countries was constructed as a list of lowest common denominators: Not all European tree species are monitored in an NFI. Instead most NFIs focus on tree species important in forestry. A taxon (genus or species) was included in the harmonised taxon list if it was listed (or could be proved not to exist naturally) in the NFI of at least 10 out of the 14 countries. A schematic example of harmonisation for *Betula* is shown (Fig. 1). The natural distribution of a taxon was determined using Atlas Flora Europaea (Jalas and Suominen 1973; Jalas and Suominen 1976; Jalas et al. 1999) and the taxon lists from the individual NFIs were sorted and interpreted, where possible with help from national experts. In NFI entries with two or more species, the first-mentioned species was considered to be the dominant and the entry was thus assigned to that taxon in the harmonised list. NFI entries that could neither be assigned to a specific taxon nor to a broad-leaved or coniferous group were discarded and not included in the total forest volume or area. Some taxon groups at the genus level were sub-divided further. An example is *Pinus*, which has been divided into two groups: southern and northern Europe, based on geographical distribution. This distribution follows Atlas Flora Europaea (Jalas and Suominen 1973; Jalas and Suominen 1976; Jalas et al. 1999). In Table 1 the NFI taxon list after harmonisation is shown. Data on tree species abundance (volume or area) were then normalised by transforming them into percentages of the total forest volume or area.

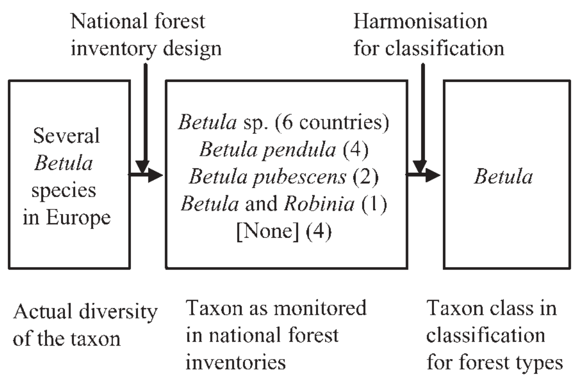

Figure 1. Schematic presentation of the harmonisation of taxon lists before classification of the national forest inventory data, example of genus *Betula.*

2.1 International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP)

Information on tree species abundance can also be found in the European-wide survey of forest condition in the ICP Forests Level I plots (UN-ECE 1998). In this study, data from the 1998 inventory were used for classification and mapping. The data set contains tree species counts on 115 tree species from 5513 plots in 30 European countries. These data have been interpolated between plots onto a regular grid and mapped as described in Köble and Seufert (2001). These tree species distributions we re-sampled and classified into forest types. The data presented in the maps from Köble and Seufert (2001) were at too high a resolution to be classified directly and were thus re-sampled using cubic convolution to 32×32 km resolution.

3. Methods

3.1 Classification of forest types

For classification of the harmonised and normalised NFI percentage data TWINSPAN (Hill 1979) vers. 2.2a was used. Analysis was performed in all possible combinations of: a) Alternative or default cut levels, b) alternative or default pseudo-species weightings, and c) omission or inclusion of outliers. Outliers were identified in PC-ORD® vers. 4, using 2 s.d. of the Chi-squared distance measure as a limit. A sister program to TWINSPAN vers. 2.2a, TWINDEND vers. 0.4 (unpublished) was used for identification of the most homogenous clusters. Clusters with a dispersion or within-group variance of <50% were chosen. Where no clusters down to the sixth level of division with <50% dispersion could be found, clusters at this last level of division were chosen in order to include all regions (except the outliers).

The ca. 8000 data points from the ICP data set were classified using TWINSPAN in PC-ORD® vers. 4. This version can handle larger amounts of data but not produce an output readable by TWINDEND. 16 groups resulting from four divisions were chosen. At this stage, the within-group variances have not been evaluated.

Table 1. The harmonised taxon list for the classification of national inventory data. South: Southern species of the genus; North: Northern species of the genus (see text). Exotic species: Not following the distribution of species given in Atlas Flora Europaea (Jalas and Suominen 1973; Jalas and Suominen 1976; Jalas et al. 1999).

Broadleaved taxa	Coniferous taxa
Acer sp.	*Abies* sp.
Alnus sp.	*Abies* Exotic species
Betula sp.	*Larix* sp.
Carpinus sp.	*Picea* sp.
Castanea sp.	*Picea sitchensis*
Eucalyptus sp.	*Pinus sylvestris*
Fagus sp.	*Pinus* Exotic species
Fraxinus sp.	*Pinus* South
Olea sp.	Other exotic pines
Ostrya sp.	Other coniferous
Populus sp.	
Quercus North	
Quercus South	
Robinia sp.	
Other broadleaved	

3.2 Mapping

The base map used for mapping of the NFI forest types is compiled from several sources. Most region outlines are from the ESRI package of ArcGIS®, but some region outlines did not correspond with the outlines given in the standard maps: The regions in the Czech Republic, Poland, Switzerland, Italy, and Portugal are all digitised from images. Mapping was performed in ArcGIS®. Re-sampling and mapping of the ICP forest types was performed in ArcView® and ArcInfo®.

4. Results

We have organised sufficient data to justify a provisional quantitative forest type classification at the European level. However, these data are very heterogeneous and were not easily accessible from all European countries. An overview of differences between the two data sets that were used for the classification is presented in Table 2.

In both the NFI and ICP data sets there are discernible clusters or forest types at this intermediary level. Many of these forest types make ecological sense as the mixture of species they contain can be recognised in the field on a larger spatial scale. Some of the resulting forest types from the classification of the NFI and ICP data sets are an ecologically unfamiliar mixture of species. This is partly due to the scale, partly to introduced species that have not been included in previous classifications of natural forest types.

From classification of the NFI data material 17 forest types were created, which are presented in Table 3 named by their most dominant (maximum three) species. These groups are the result of a classification using default settings and omitting outliers, a procedure that gave the groups with the least variance. In spite of the omission of outliers, some groups are very small only comprising between two and seven regions. These “rare” groups do not

Table 2. Comparison of the data from the national forest inventories and from the ICP Forests Level I plots. *: Comparability as a function of coverage, geographic scale, taxonomic resolution, forest definition, and data sampling procedure.

	National forest inventories	ICP Forests
Coverage	Most of (forested) area in contributing countries	5500 plots in major forests in monitored area
Geographic scale	Region	Plot – interpolated
Taxonomic resolution	Varying	Fine. 115 species
Comparability* between countries	Varying-low	Very high

Table 3. National forest inventory forest types.

Forest type numbered by TWINSPAN group	Dominant species	Occurrence in number of regions
00	*Picea, Pinus sylvestris*	98
11	*Pinus South, Quercus* South, *Eucalyptus*	43
0101	*Picea, Fagus, Abies*	27
1000	*Quercus* South, *Quercus* North, *Castanea*	3
1001	*Pinus* South, *Quercus* North, *Quercus* South	13
1011	*Pinus* South, *Quercus* South, *Pinus sylvestris*	29
01000	*Picea*	3
01001	*Pinus sylvestris, Larix, Abies*	3
01100	*Fagus, Picea*	15
01101	*Picea sitchensis,* other broad-leaved, *Quercus* North	33
10100	Other conifers, *Picea, Larix*	2
011100	*Quercus* North, other broadleaved, *Fraxinus*	38
010101	*Quercus* North	16
011110	*Quercus* North, *Castanea, Quercus* South	5
0111111	Other broad-leaved, *Quercus* North, *Fagus*	15
101010	Other broadleaved, *Pinus* South, *Quercus* South	7
101011	*Pinus* South, other broadleaved, other conifers	6

contain rare tree species or represent rare forest types. On the contrary, some well-known forest types with limited geographical distribution such as floodplain forests (see Bradshaw and Friis-Møller, this volume) are not to be found on this classification. The harmonisation of the taxon list does not give room for rare tree species, hence many forest types are characterised by mixed genus groups. Instead, forest types with exotic species, e.g. the forest type NFI forest type *Pinus* South-*Quercus* South-*Eucalyptus*, are found.

We describe here two NFI forest types dominated or co-dominated by *Fagus* as examples of ecologically familiar forest types. The distribution of both forest types in this data set is shown in Figure 2, but the actual distribution is somewhat obscured by the lack of data from, among other countries, Germany. The forest types are named by their three most dominant taxa, which in *Picea-Fagus-Abies* are represented as follows: *Picea* sp. 23%, *Fagus* sp. 21%, *Abies* sp. 16%. The forest type also includes *Quercus* North-species 15%, but no other taxa exceed 5%. In the forest type *Fagus-Picea*, the taxa that exceed 5% are: *Fagus* sp.29%, *Picea* sp. 21%, other broad-leaved species 9%, *Quercus* North 8%, other coniferous species 6%, and exotic species of *Abies* 6%.

Still following the principle of approximately 10–20 forest types, 16 forest types were taken as the result of a classification of the ICP data. These are presented in Figure 3. "Smaller" forest types, such as floodplain forest can not be identified from this data set either. In contrast to the NFI data set no taxon harmonisation was necessary, yielding a more diverse range of forest types with an especially broad diversity in the southern part of the European map.

Again, we show here the ICP forest types, in which *Fagus* plays a dominant role (see Figure 2). Here, also a mixture with *Picea sitchensis* is singled out. The geographic distribution of forest types in the map follows the actual forest area to some degree, but has been modified in the re-sampling procedure.

5. Discussion

The strictly quantitative approach to classification of forest types adopted here brings the advantages of objectivity and repeatability. The areal extent of the forest types can also be monitored through time and even compared with similar classifications based on pollen data showing forest type dynamics over thousands of years (Lindbladh et al. 2000). The data are clearly 'actual' vegetation and when mapped could be compared with potential vegetation maps such as that of Bohn et al. (2000). We compare the ICP map showing the three forest types dominated by *Fagus* with the NFI *Fagus* types and Bohn's potential distribution of *Fagus* forest types (Figure 2). The ICP data record *Fagus* plantations beyond its natural range limits in the British Isles. The ICP data also show less *Fagus* than is natural in Denmark, northern Poland and north-west Germany. This reflects both extensive forest clearance, but also the somewhat patchy nature of the ICP sampling network. Differences between the potential and actual vegetation maps show the extent of present deviation from 'natural' conditions. This concept could be of practical value in forest ecosystem restoration policy.

Several of the types generated by TWINSPAN are ecologically familiar from previous stand or regional scale classifications, such as the ICP forest type *Picea abies – Fagus sylvatica – Abies alba*; while others are clearly artificial such as the ICP forest type *Eucalyptus – Pinus pinaster – Pinus sp.* or *Fagus sylvatica – Picea sitchensis – Picea abies*. This reflects both regional 'autocorrelation', meaning that species associated at stand-scale can also be associated at regional scales and illustrates the extent to which artificial forest types have been created at landscape-continental scale. Thus mapping forest types at this scale is not purely an administrative abstraction, but reflects underlying ecological relationships and illustrates the scale of cultural landscape modification.

The major disadvantage of the quantitative approach, at present, lies in the lack of homogeneity of NFIs leading to significant loss of information when they are merged. The ICP data set is well standardised, but small, and was not originally designed as a measure of forest composition. Increased European co-operation and increased use of standardised data collection techniques will hopefully reduce this problem in the future. The increasing use of remotely-sensed forest surveys is clearly an important development. More homogenous, high resolution data sets will require new classification techniques. We have begun to experiment with neural network classifications of homogenous Scandinavian forest data and initial results are promising.

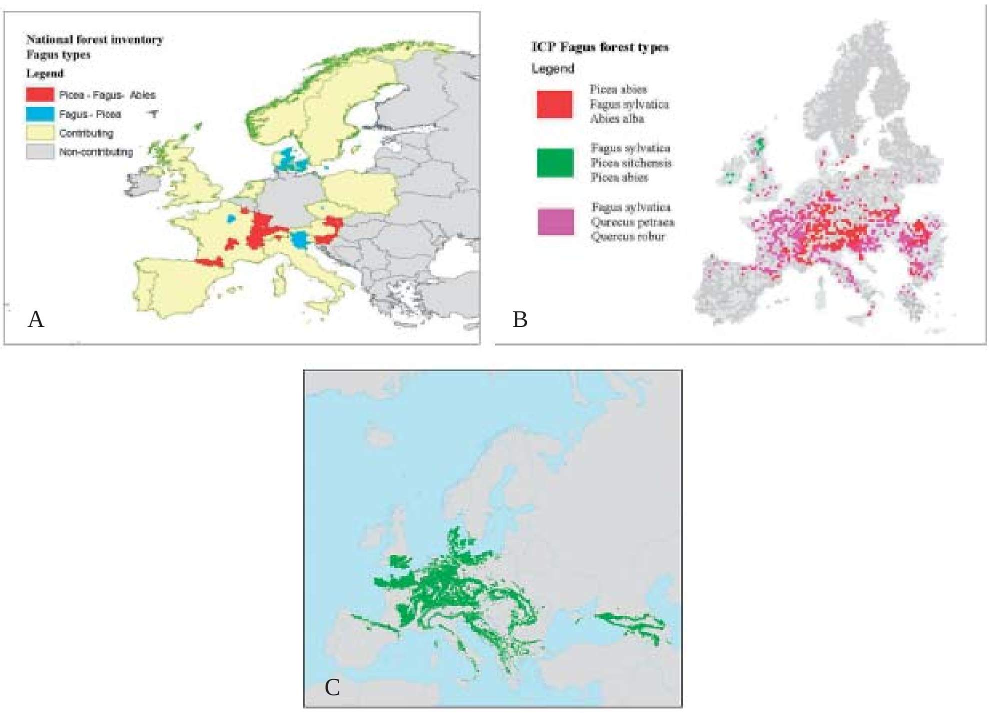

Figure 2. A) National forest inventory forest types with *Fagus*. B) ICP forest types with *Fagus*. C) Bohn (2000) forest types with *Fagus*.

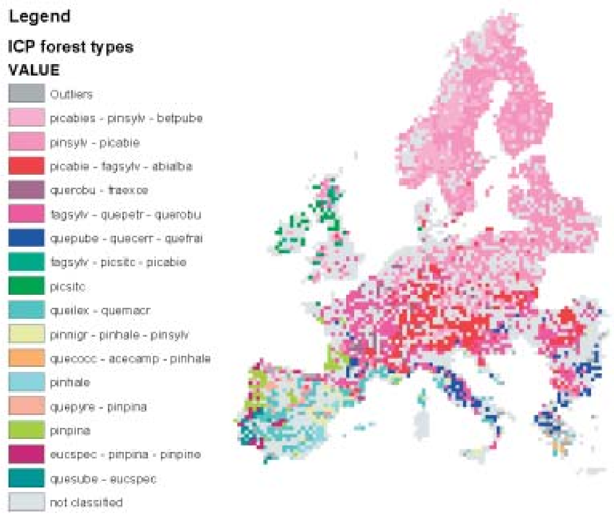

Figure 3. ICP forest types.

Acknowledgements

We would like to thank Jogeir Stokland for ideas and encouragement, Oriana Pfizer for initial mapping of the ICP forest types and Renate Köble for kindly making tree species distribution from the ICP Forests Level I data set available to us. Thanks also to Mette Palitzsch Lund for help with the Bohn map.

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Stand-level Forest Type Approach in Italy: Experiences from the Last Twenty Years

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Abstract

The concept of stand-level forest types embraces all those forest classification and nomenclature systems that on the basis of different disciplines allow a hierarchical assessment of forest stands by a synoptic evaluation of either vegetation aspects (both floristic and phytosociological) and of ecological-silvicultural ones. The scale of the assessed units (i.e. the stand-level forest types) is relative to the scale of forest resources management. That is, a forest type scheme is designed to distinguish among forest stands in a forest compartment those that need different consideration when planning sustainable management practices. Such a kind of forest classification schemes benefits a more practical implementation when compared with those based on a phytosociological approach or designed for land cover mapping. In this paper, a synthesis of experiences from Italy, where classification systems of stand-level forest types are well widespread on a regional administrative basis, is presented and discussed as a suitable, exemplificative case study.

Keywords: forest nomenclature systems, forest habitat, forest stand, sustainable forest management, forest mapping.

1. Introduction

All over the world, the practice of sustainable, multi-use forestry has progressively been taking place. A viable support is to link sustainability implementation and assessment to a "habitat" approach, adopting forest types definition and description as a reference criterion for management on a stand scale.

Marco Marchetti (ed.)
Monitoring and Indicators of Forest Biodiversity in Europe – From Ideas to Operationality
EFI Proceedings No. 51, 2004

In general terms, a forest type is "a category of forest defined by its composition, and/or site factors (locality), as categorized by each country in a system suitable to its situation" (Canadian Forest Service 1995). The boundaries are set up where spatial variability in forest diagnostic attributes (i.e. composition and/or site factors) causes qualitative shifts in forest structure and/or processes of concern for practical purposes (i.e. site productivity, habitat quality, type and quality of wood resources, forest health status, etc.).

Distinctively, the concept of stand-level forest types embraces all those forest classification and nomenclature systems that allow a hierarchical assessment of forest stands by a synoptic evaluation of either vegetation aspects (both floristic and phytosociological) and of ecological-silvicultural ones. The scale of the assessed units (i.e. the stand-level forest types) is relative to the scale of forest resources management. That is, a forest type scheme is designed to distinguish among forest stands in a forest compartment those that need different consideration when planning sustainable management practices (Del Favero 2000).

The meaningfulness with respect to biodiversity is that stand-level forest types are separated where the natural variability of forest attributes (mainly forest composition and/or site factors) or anthropogenic factors (past and actual management practices) causes qualitative shifts in biodiversity key factors, at the forest stand level. Forest stand types thus distinguish management scenarios which are significantly different as regards the targets of biodiversity conservation, that is the maintenance of processes and factors that maintain, generate or directly reflect the variation of forest biodiversity in the forest management unit (Barbati et al. 1999).

Forest typology plays an important role making easier the exchange of information among professionals and researchers, due to the language standardisation, and possible the comparison between experiences in order to make the better choices regarding forest planning and management prescriptions. As a matter of fact, stand-level forest types have a straight operational meaning, each type being provided with distinctive silvicultural prescriptions.

In this paper, we refer to the Italian experience as a suitable, exemplificative case study with regard to forest stand classification.

2. Stand-level forest type systems in Italy

Classification systems of stand-level forest types are well widespread in Italy, where the administrational/regulatory framework of forest management is ruled on a regional basis. Regional forest services set the basic guidelines of forest planning and supervise the conformance of forest management plans to these requirements. In this context, for instance, the forest legislation of the Friuli Venezia Giulia, Veneto and Piemonte Regions adopt stand-level forest types as a reference framework for normative prescriptions. Another relevant example of application concerns the assessment of priorities in silvicultural tendings to prevent forest fires in the Veneto region by the evaluation of pirological potential of each stand-level forest type.

Physiognomic features of the forest cover and its dominant species are used to discriminate the main forest formations of each administrative Region (e.g. *Picea abies* forests, *Fagus sylvatica* forests, *Castanea sativa* forests, *Quercus ilex* forests). Such classes correspond to the higher level of the classification, the category level. Each category splits up in a convenient group of forest stand types, which are basic unit of the system of nomenclature, the various types of each category under different management conditions. The peculiarity of typological studies is to split up the complex of forest system in small parts sufficiently

Table 1. Range of information provided by stand-level forest types systems in selected Italian Regions. The capital letter is referred to the Region within which the system is applied: A = Veneto; B = Piemonte; C = Friuli Venezia Giulia; D = Toscana; E = Sicilia; F = Lombardia; G = Provincia di Trento; H = Marche.

Topics	Stand-level forest types system							
	A	B	C	D	E	F	G	H
Number of forest types	106	84	102	88	60	97	110	37
Correspondences with other nature nomenclature systems	X	X	X	X	X	X		X
Geographic localization	X	X	X	X	X	X	X	X
Complete mapping (scale 1:10000 / 1:25000)			X					X
Geomorphological characterization	X	X	X	X	X	X	X	X
Geopedological characterization	X	X	X	X	X	X	X	X
Tree species composition	X	X	X	X		X	X	X
Composition of the shrub and herbaceous layers	X	X	X	X			X	X
Stand dynamics characterization	X	X	X	X		X		X
Main diseases and parasites	X					X		
Average biometric (dendrometric) parameters	X		X			X		X
Valuable landscape and naturalistic features	X		X			X		
Susceptivity to fires	X					X		
Silvicultural guidelines	X	X	X	X		X		X

homogeneous to be thouroughly characterized, but not so small to hinder a comprehensive operational management view.

A main target of the stand-level forest type approach as conceived in Italy is to have types characterizable by suitable indicators for aspects like forest health monitoring, forest fire assessment, soil conservation, nature conservation, timber production. However, the characterizing information connected to stand-level forest types is currently quite heterogeneous across administrative Regions in Italy (Table 1).

Generally, few indicators are provided about damages and fauna characterizing each forest type: the latter, if present, is focused on the relationships with ornithic species (e.g. Table 2) and ungulates. Relatively few information are given for the biometric features too (average growing stock, increments, etc.).

On the contrary, in most cases silvicultural guidelines are defined for each type, that constitute the essence of this kind of classification and make it significantly different from other nature classification tools.

Table 2. Examples of biodiversity information provided by the stand-level forest type system of the Veneto Region (modified from Barbati et al. 1999).

Indicators	Stand-level forest types		
	Esalpic beech mountain forest[(a)]	Fir forest on siliceous rocks[(b)]	Spruce high mountain forest on mesic soils[(c)]
Spatial pattern			
Widespreadness	High	High	High
Connectedness	Connected	Connected	Connected
Tree species contagion potential	Low	Low	High
Forest structure			
Uneven aged stands: percent of trees in dbh classes (<32.5, 32.5–47.5, 47.6–62.5, >62.5 cm)	-	60; 25; 10; 5	-
Even aged stands: number of development stages and average extent of each development stages (ha)	6 (22)	-	7 (19)
Ground vegetation layer			
Average number of herb/shrub species (min-max)	41 (18–62)	32 (25–41)	35 (16–49)
Average number of hemerophyta herb/shrub species	0.29	0.1	0.25
Dynamic trend in the number of herb/shrub species	Variable over time	Temporarily variable	Variable over time
Animal species			
Average number of bird species (min-max)	25 (23–27)	35 (33–37)	25 (23–27)
Overall naturalistic quality			
Flora[(d)]	1	0	1
Vegetation[(e)]	Medium	Medium	Medium
Macrofauna[(d)]	11	10	8

[(a)] Main tree species: *Fagus sylvatica;* secondary tree species: *Picea abies*; accessory tree species: *Abies alba, Laburnum alpinum, Acer pseudoplatanus, Sorbus aucuparia, Fraxinus excelsior, Ilex aquifolium, Populus tremula, Sorbus aria, Salix appendiculata, Prunus avium, Laburnum anagyroides.*
[(b)] Main tree species: *Picea abies, Abies alba;* accessory tree species: *Acer pseudoplatanus, Alnus viridis, Fagus sylvatica, Sorbus aucuparia, Larix decidua.*
[(c)] Main tree species: *Picea abies;* secondary tree species: *Larix decidua;* accessory tree species: *Abies alba, Sorbus aucuparia, Sorbus chamaemespilus, Fagus sylvatica, Alnus viridis, Salix appendiculata.*
[(d)] Number of protected, rare or endemic species.
[(e)] Phytogeographic relevance of the type (high, medium, low).

3. Relevant issues about typological studies

3.1 Typological vs. phytosociological approaches

Forest type definition is carried out using an approach integrating floristic knowledge with silvicultural understanding. Each type defines unique combination of forest vegetation properties and site factors (both environmental and anthropogenic) functionally interdependent. Such units are characterized by recurrent patterns in floristic composition and structural (spatial and architectural) properties. A typological unit may fully overlap with a phytosociological one, or may overlap only partially (in relation to specific management needs), or it may include more than one phytosociological unit in the case that vegetation differences at the considered level do not significantly affect management choices.

Most working teams involved in forest typological studies in Italy include vegetation scientists. Both the phytosociological and the forest typological approach consider the same object, the forest and the relationships among the flora and the environment, with different goals: the focus of the phytosociological approach is overall knowledge and interpretation, while the typological one concerns the management. However, in most Italian Regions phytosociological syntaxa and stand-level forest types have been generally found more correspondent than it was expected (Bernetti 1998).

3.2 Correspondence with other nature classification systems

Nature classification systems other than the phytosociological ones are widespreading in Europe, based mainly on physiognomical features. For instance, the sites included in the Natura 2000 network (see Habitat Directive 92/43/EU) have been classified by a system (Natura 2000 system, see Romao 1996) framed by biogeographic regions (Boreal, Atlantic, Macaronesian, Continental, Alpine and Mediterranean) and developed from the physiognomical-phytosociological Corine Biotopes system. Natura 2000 system is characterized by a relatively good level of comparability and correspondence with stand level forest types in Italy, albeit it shows considerable gaps (e.g. lack of habitats referred to *Abies alba* stands or to *Fraxinus-Ostrya* and *Ostrya-Quercus* stands).

EUNIS (European Nature Information System) is a more recent habitat classification system (Davies and Moss 1997). Distinctively, the EUNIS system is based on a hierarchical physiognomical classification up to six levels. The third level roughly corresponds to the categories and stand types of the typological systems in Italy.

However, there are many gaps in the correspondence due to the scattered detailness of the EUNIS system, for some classes quite high and for others too rough. Surprisingly the EUNIS system lacks classes for *Quercus cerris* stands, *Alnus viridis* stands, pure *Pinus cembra* stands as well as for the mixed *Picea-Fagus* stands. Beech forests of the Italian Alps are difficult to be classified because in an intermediate position among the middle European and the south European beech forests conceived by EUNIS. Also most *Pinus silvestris* forests have not a specific reference within EUNIS, albeit there is a unit for the steppic Alpine Scotch pine forests, a very rare type in Italy. Moreover, the significant distinction between montane and subalpine *Picea abies* forests is not conceived.

Why such differences of detailness? That is partly because there are real differences among forest formations in the different parts of Europe, but, on the whole, such a situation depends on the fact that the system is prevalently based on habitat recognitions by whom developed it, and probably the developers have not been evenly chosen from all the European countries.

3.3 Framework structure of typological system

The degree of detail of a classification system is determined not only by the adopted scale of reference, but also by the hierarchical structure of the system. For instance, the *Pinus halepensis* forests in the Apulia Region (Southern Italy) are classified by one class at the third

Table 3. Examples of the correspondence of selected stand forest types of the Veneto Region with the units from Natura 2000 and EUNIS classification systems.

Selected stand forest types in the Veneto Region	Natura 2000 units*	EUNIS units**
Flowering ash – European hophornbeam stands on gorges Flowering ash – European hophornbeam stands among rocks Flowering ash – European hophornbeam stands on detritic hillsides Typical Flowering ash – European hophornbeam stands	?	G1.7C1
Altimountain spruce stands on carbonatic rocks Subalpine spruce stands on carbonatic rocks	9411	G3.1B
Mountain spruce stands on xeric soils over siliceous rocks	9412	G3.1C
Altimountain spruce stands on xeric soils over siliceous rocks Subalpine spruce stands on xeric soils over siliceous rocks	9411-9412	G3.1B
Altimountain spruce stands on mesic soils over siliceous rocks Subalpine spruce stands on mesic soils over siliceous rocks	9411	G3.1C G3.1B
Secundary spruce stands	9412	G3.1J

* 9411 Alpine and Carpathian sub-alpine spruce forests, *Piceetum subalpinum*; 9412 Inner range montane spruce forests, *Piceetum montanum*
** (2.3 version, 28 february 2002): G1.7C1 - [*Ostrya carpinifolia*] woods; G3.1B - Alpine and Carpathian sub-alpine [*Picea*] forests; G3.1C - Inner range montane [*Picea*] forests; G3.1J - [*Picea abies*] reforestation

level of the EUNIS system and by four stand-level types according to the local nomenclature system: Aleppo pine forest of coastal cliff; Aleppo pine forest succeeding with *Pistacia lentiscus*; Aleppo pine forest succeeding with *Quercus ilex*; secundary Aleppo pine forest. It may seem obvious that a system set up on a continental level cannot have the same detailness than a system established on a local scale, but such detailness might be theoretically obtained also articulating the hierarchical structure down to finer levels. However, such an option leads to the risk of excessively complicating all the system for operational uses.

It is also to be stressed the use of classification systems carefully focusing the specific target they are conceived for. Consider a forest with *Picea abies* prevailing as cover percentage by 90%; the remaining cover is by *Fagus sylvatica*, more abundant as natural regeneration in the understorey, while natural regeneration from spruce lacks. This situation, although simplified, is quite frequent on the Italian Alps, distinctively in the belt between the esalpic and the mesalpic zones. Following a deterministic approach, such a forest might be classified as a spruce forest, because the cover of spruce exceeds a given threshold, say for instance 80%. This is a clear classification that can be easily delegated to an automatic device: e.g. such a classification could be useful in a kind of mapping by an algorithm based on photointerpretation and field inventoried data. If, on the contrary, thinking at the system functioning and management aspects, the relevance of the beech component is stressed, this forest should be framed within the type of *Piceo-Fagetum*. On the other hand, phytosociologists might classify that forest among the beech forest type since overall floristic composition and vegetation dynamics tend to that typical of a beech forest. None of these choices is wrong. However, stressing the managemental aim of typological studies, the silviculturist should consider this forest as a *Piceo-Fagetum* because, in the short period, his/her task will be to manage a mixed forest of spruce and beech.

4. Forest type mappability

Forest types mapping is a demanding operational perspective. In two administrative Regions (Friuli-Venezia Giulia and Marche) a complete mapping of forest types is already available, and most other Italian Regions are in the process of mapping the forests types. Mapping is widely recognized as a fundamental mean to effectively exploit the applicative potential of stand-level forest type classifications: such a knowledge could be fed back into the planning process, improving the capacity of local forestry administrations in setting provisions for forest practice and management.

However, the mappability of the type classes is, on principle, low, due to the fine spatial pattern of their diagnostic attributes and to the level of information required to distinguish among its classes (i.e. floristic knowledge). We cannot suppose to derive such information entirely from remote sensing. If types were detectable with sufficient accuracy, it would be feasible to use existing schemes to map the type level. Otherwise, aggregation of similar types should be tested until getting the required classification accuracy.

In principle, local classification of physiognomic attributes of forest cover requires images with different spectral, radiometric and geometric resolution. Data-fusion techniques could further improve the classification procedures. The last generation of very high-resolution satellites and the development of object-oriented classification techniques based on multiresolution segmentation seem also to effectively support such an effort (Chirici et al. 2003).

The mapping challenge has driven a recent experience of development of a forest type scheme in the Abruzzo Region (central Italy). Here, a pilot study has been undertaken in a forest area (about 110 000 ha), representative of the environmental conditions of the Italian Central Appennine. The kernel of the work is the design a forest type scheme which embodies, at root, mapping issues (Corona et al. 2001). Hence, if the general definition of classes satisfies the forest type concept (see Chapter 2), the mappability issues have been prominent in the selection of classifiers. That is, to find, accordingly: (i) a set of forest properties and environmental conditions which discriminates between forest landscapes suitable categories and, between categories, suitable forest stand types; (ii) a way to “materialize” these boundaries in a fine scale map (1:10 000), with the required accuracy (Figure 1). The first phase makes use of remote sensing capabilities to discriminate the physiognomic attributes of the forest cover. The power of multi-spectral/multi-temporal discrimination of satellite imagery (Landsat 7 and Spot 5), combined with the high geometric resolution of large scale (e.g. 1:10 000) aerial photos (i.e. B/W ortophotos, CIR aerial photographs), allows a successful (i.e. spatial accuracy) mapping of categories. Existing ancillary information (e.g. geographical databases on environmental features and/or forest resources) helps improving the thematic accuracy of mapped features. The availability of Colour Infrared Photos at suitable scale (not smaller than 1:20000) makes easier the classification at category level, permitting also a first delineation of some types. A ground survey phase completes the mapping process; its prominent role is to deepen the thematic content of the mapped features.

A Bayesian approach is proposed to fuse local classification with the understanding of the global spatial distribution of type classes as “encapsuled” in the forest type scheme (i.e. information on site limiting environmental factors for each type). By this, a probabilistic a priori model can be derived to draw the spatial distribution of environmental factors from existing geographical databases (e.g. geology, climate, soil and site data: see Tokola and Hekkila 1997, Chirici, 2003), as, for instance, carried out in the Regional Park of Colli Euganei in Veneto region (Del Favero 2001). If the main ecological factors that significantly interact with the forest tree species populations to suitably live and reproduce in a given range

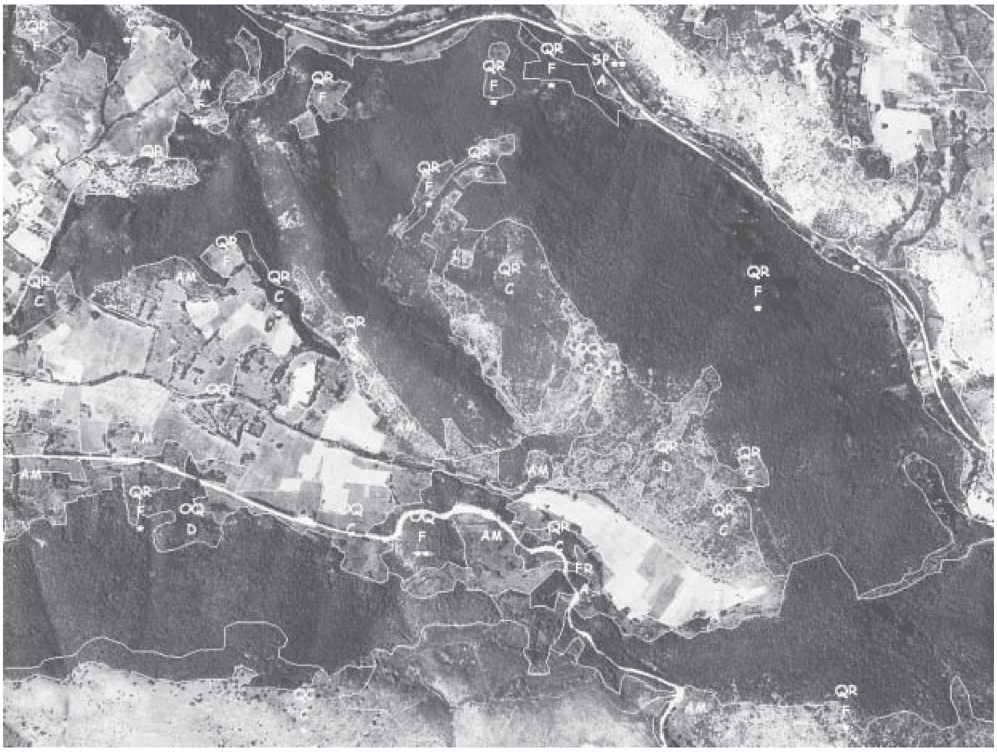

Figure 1. Example of stand-level forest types delineated on a digital panchromatic ortophoto from the Abruzzo Region. AM = thorny shrubs stands; FR = sub-montane beech forest; OQ = sub-montane European hophornbeam stands; QR = sub-montane Pubescent oak stands; SP = Black pine forest plantations.

of sites can be quantitavely assessed, then the combination in a GIS environment of the maps of each of such factors can restitute a probabilistic assessment of the ecological coherence of the given tree species within the given sites. This is not potential vegetation, rather it is predictive forest type mapping, *sensu* Vogiatzakis et al. (2003).

Such an assessment might be interpreted under a typological approach and overlapped with the map of actual forest species distribution to evidence the impact by forest management:

- stands with all the autochthonous and ecologically coherent tree species substituted by one or more species not ecologically coherent (e.g. pure *Robinia pseudoacacia* stands);
- stands where the ecologically coherent tree species composition has been modified only partially by introduction of new species (e.g. mixed *Quercus* and *Ostrya* stands invaded by *Robinia pseudoacacia*);
- stands where the ecologically coherent tree species composition is modified for the lack of one of the main species (e.g. *Castanea sativa* coppices where *Quercus petraea* tends to disappear);
- stands with light anomalies in the ecologically coherent tree species composition due to the lack of secondary species (like in the pseudo-maquis in the Veneto region where *Quercus* species should naturally be more widespread);
- stands with a good correspondence between current and ecologically coherent tree species compositions.

Another derivable representation could be that of spatial anomalies, i.e. a map of those patches characterized by shape, size and interdistance different from the "natural" situation. In fact, forest management tend to increase the size and the interdistance of the patches belonging to the same forest type as well as to simplify their shape. This map could give indication about where to intervene for re-establishing a more "natural" landscape biodiversity.

5. Remark

The benefit of stand-level forest typification is evident in reporting and when setting targets of sustainable forest management (Barbati et al. 1999). In Italy, the road has been open few decades ago. Any classification system, if well formulated, is certainly a progress for the comprehension of the world. However, the world is more complex than any algorithm we can state. There are no formulas that can explain the truth, the harmony, the simplicity of the world. Forest types boundaries (and the associated polygons in the forest maps) are somewhat arbitrary, but convenient to the application. Forest homogeneity found at the type level makes it a consistent frame for discriminating managementally relevant habitats.

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Combination of Structural and Compositional Factors for Describing Forest Types Using National Forest Inventory Data

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Abstract

For the first time in Portugal, simple variables describing the vertical structure and the composition of forests on the Portuguese mainland were included in the 2258 sample plots of the National Forest Inventory (DGF 2001). The vertical forest structure was assessed by percentage cover of seven height classes and the composition of the different layers was described using plant species, or groups of plant species, easily identifiable in the field.

Cluster analysis, in particular *K-means* statistics, was performed using combinations of vertical structure and compositional data, resulting in ten main natural groups or forest types: 1) *Quercus pyrenaica* forests; 2) Other deciduous oak forests; 3) *Arbutus unedo* forests; 4) *Cistus* shrubs; 5) *Cytisus* shrubs; 6) *Acacia* forests; 7) *Quercus suber* forests; 8) *Pinus pinaster* forests; 9) *Eucalyptus* forests; and 10) Other forests. The last four groups were further subdivided according to the vertical structure resulting in twenty final forest types. The geographical distribution of these forests types and the implications for biodiversity and other forest issues are presented and discussed.

Keywords: national forest inventory, composition, vertical structure, forest types, biodiversity.

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1. Introduction

Biological diversity in forests depends on their composition and vertical structure (Puumalainen 2001) and efforts have been made all over the world to include them in the criteria and indicators of forest biodiversity (Larson et al. 2001; Stork et al. 1997). The top to bottom description of forest composition, from tall to smaller trees, shrubs, grasses and other plants characterise its vertical structure. Among other features, a forest with a more diverse composition and vertical structure provides more habitats for animals where they can find food and cover, maintains moderate temperatures by reducing convective and radiative heat loss, thus providing climatic buffered areas, and has greater aesthetic and recreational value.

The aim of the traditional National Forest Inventory (NFI) was to describe the main features of Portuguese forests in terms of size, condition and change. But it was more concerned with their productive features than an extensive description of the forests. The last revision of the NFI (DGF 2001) aimed to go further, building on the information on productivity to describe the forests more completely. Despite adopting a simple key based on the cover of the dominant forest tree species it assessed, for the first time in Portugal, the cover of other plant species according to height classes. This information provided a useful tool to better understand our forests in their composition and vertical structure. Adopting a simple methodology, we obtained a new key of Portuguese forest types based on the vertical distribution of the forest plant species and a “tri-dimensional” view of our forest areas.

2. Methodology

To describe forest composition and vertical structure together, and to better understand the combined effect on the diversity of vertical forest structure, the percentage cover of forest species according to seven height classes are used as variables. These height classes were defined from ground level to 0.5 m, from 0.5 m to 1.0 m, from 1.0 m to 2.0 m, from 2.0 m to 4.0 m, from 4.0 m to 8.0 m, from 8.0 m to 16.0 m, and over 16.0 metres. The composition of the different layers was described by plant species, or groups of plant species, easily identifiable by the NFI team responsible for data collection on the ground.

The measurements were assessed from 2258 sample plots randomly distributed through the forest area and identified by their geographical coordinates. In every sample plot the percentage cover of the first three dominant plant species was assessed in decreasing order of importance (Figure1). The percentage cover of plant species, or groups of plant species, was assembled in a table in which each species according to height class was considered as *pseudo-species.* The connection established between the plant species and its respective layer, called a *pseudo-species,* allowed for an estimate of the proportion of cover that each plant species contributes to a specific height class.

Applying a *K-means* analysis (SPSS 2003) to the matrix of the *pseudo-species* percentage cover, natural groups based on the forest composition and vertical structure similarities were obtained. Following a sequence of attempts to determine the number of natural groups that better distributed the 2258 sample plots, ten groups were defined that effectively characterised Portuguese forests, with each natural group corresponding to a particular forest type. The result of the *K-means* analysis was the average percentage cover of plant species according to height class for each natural group, thus outlining the profiles of resulting forest types.

Once the number and the description of the forest types were determined, they were mapped by the Thiessen polygons method (Soares 2000) using the geographical coordinates of the 2258 sample plots.

	Total % Cover	Cover per species					
		Species	%	Species	%	Species	%
16	13	A	13				
8	22	A	14	B	8		
4	24	A	14	B	10		
2	25	B	20	A	5		
1	40	C	20	B	20		
0,5	40	C	30	E	10		
	50	C	30	D	10	E	10

Figure 1. Percentage cover of the first three dominant plant species.

3. Results and discussion

According to the NFI (DGF 2001), of the total area of the Portuguese mainland (8 879 862 ha), forest constitutes the main land use, occupying 38% of the territory, followed by 33% of agriculture land and 23% of uncultivated land including shrubs, natural meadows and abandoned land. The traditional forest area key is described as: maritime pine (31%), cork oak (22%), eucalyptus (21%), holm oak (14%), other oaks (4%), umbrella pine (3%), chestnut trees (1%), other broadleaves (3%) and other conifers (1%).

Based on the combination of composition and vertical structure, ten major forest types were identified: *Quercus pyrenaica* forest (0.4%), other oak forests (0.2%), *Arbutus unedo* forest (0.4%), *Cistus* shrubs (6.6%), *Cytisus* shrubs (4.5%), *Acacia* forest (0.8%), *Quercus suber* forest (8.5%), *Pinus pinaster* forest (29.5%), *Eucalyptus* forest (18.4%) and other forests (30.7%). Although the occupation areas of both keys are quite different, the forest types obtained by the *K*-means analysis give us complementary information.

Quercus pyrenaica forests

Quercus pyrenaica forests, distributed mainly in the north and the interior centre of Portugal, represent 0.4% of the Portuguese mainland forest area. They are very dense forests that occupy all the height classes. The *Quercus pyrenaica* has a very significant percentage of cover from the ground to 16 metres. The chestnut trees form, together with the *Quercus pyrenaica* some mixed stands where *Cytisus spp*, *Erica spp*, *Lavandula spp* and other groups of species occupy the understory up to 2 metres (Table 1).

Other oak forests

The other oak forests are distributed in three main small areas in the north, the central north and the centre of the Portuguese mainland. They represent only 0.2% of Portuguese forest area and are mainly dominated by *Quercus faginea*. It is a very dense forest of oak from the ground to 16 metres. In some stands, holm oak is present from 1 to 8 metres. *Rubus spp*, a typical climber, is abundant in these forests. The understory is also composed of *Cytisus spp* and other plant species (Table 1).

Arbutus unedo forests

These forests are distributed in small areas in the interior north and centre of Portugal but toward the south, these formations are larger. This forest type represents 0.4% of the Portuguese forest area. Its vertical structure shows that these formations are lower forests with a dense cover of *Arbutus unedo* from the ground to 4 metres, sometimes reaching 8 metres. The understory is also defined by significant amounts of *Ulex spp*, *Erica spp*, *Cistus ladanifer*, and other plants such as *Quercus coccifera*, *Juniperus spp* and *Pterospartum tridentatum*. The overstory is mainly composed of cork oak formations, sometimes forming mixed stands with maritime pine (Table 1).

Cistus shrubs

The *Cistus* shrubs are distributed in the interior north and centre of the Portuguese mainland but it is more abundant in the south, increasing from the coast to the interior. They represent 6.6% of the forest area. Although it is considered a forest land use class, it is important to notice that the most important cover does not exceed the 2–4 metres height class. It is composed mainly of *Cistus ladanifer*, *Lavandula spp* and other understory plants. The overstory is characterised by open formations of “montados” of cork oak and holm oak and this is the reason why they are considered forest areas. From our point of view, this forest type should be considered a shrub formation (Table 1).

Cytisus shrubs

This forest type is mainly distributed in the interior north and centre of the country and a significant area of *Cytisus* shrubs exists on the west coast of Portugal. It represents 4.5 % of the forest area. But, like *Cistus* shrubs, the most important cover does not exceed the 2-4 metres height class and is composed mainly by *Cytisus spp*, *Cistus ladanifer*, and other understory plants. The overstory is characterised by very open and tall formations of maritime pine and *Quercus pyrenaica*. We could also consider this forest type a shrub formation (Table 1).

Acacia forests

Even if we can find some significant acacia stands on the coast near Lisboa and the southwest coast of Portugal, this forest type is mostly distributed along the central and northwestern coasts of Portugal, in some regions reaching the interior north and centre of the country. *Acacia* forests represent 0.8% of the forest area. This forest type shows a dense cover from

Table 1. Average percentage cover of plant species, or groups of plant species, according to height class (C1 to C7) of forest types: *Quercus pyrenaica* forests, other oak forests, *Arbutus unedo* forests, *Cistus* shrubs, *Cytisus* shrubs and *Acacia* forests.

Forest type	*Quercus pyrenaica* Forests				Other oaks forests					*Arbutus unedo* forests						*Cistus* shrubs					*Cytisus* shrubs				*Acacia* forests				
Plant species composition according to height class	*Quercus pyrenaica*	*Castanea sativa*	*Cytisus spp*	Others	*Quercus spp*	*Quercus rotundifolia*	*Rubus spp*	*Cytisus spp*	Others	*Quercus suber*	*Arbutus unedo*	*Ulex spp*	*Erica spp*	*Cistus ladanifer*	Others	*Quercus suber*	*Quercus rotundifolia*	*Cistus ladanifer*	*Lavandula spp*	Others	*Pinus pinaster*	*Quercus pyrenaica*	*Cytisus spp*	Others	*Pinus pinaster*	*Pinus pinea*	*Eucalyptus spp*	*Acacia spp*	Others
C1 (>16m)				3.3																	0.3			0.5	2.8	1.5	0.8		1.8
C2 (8–16m)	10.8	0.9			17.5					3.4					1.5	3.0	1.3			2.6	2.2	1.0		2.0	4.8	4.7	2.1	9.4	0.3
C3 (4–8m)	26.9	1.4			47.5	1.5				8.3	1.8				1.0	8.5	6.5			4.2	3.4	2.5	0.6	6.1	6.6	5.6	2.4	18.5	0.3
C4 (2–4m)	36.7	2.8	0.1		46.3	1.5	6.3	2.0		7.5		2.0	0.3	0.3	0.5	3.9	5.7	4.5		4.1	3.0	2.8	6.4	5.8	2.2	3.5	1.8	25.9	0.6
C5 (1–2m)	42.2	3.9	0.7	0.7	37.5	0.5	8.8	5.0		4.0	26.3	3.5	1.3	1.2	3.2	1.4	2.4	17.1	0.1	5.2	2.7	2.2	22.0	4.9	1.5	2.9	0.6	26.5	0.4
C6 (0.5–1m)	34.7	3.9	2.1	2.3	30.0		15.0	2.5	13.8	2.5	29.0	7.0	2.5	1.5	10.5	0.4	1.5	30.8	1.9	8.2	2.1	1.9	38.1	7.7	1.2		0.6	26.5	3.1
C7 (0–0.5m)	35.6	4.4	4.1	6.7	27.5		16.3	3.8	21.3		30.5	6.0	4.0	4.5	8.0	0.5	1.4	37.3	5.1	12.5	1.8	1.8	42.8	11.2	1.8			26.0	9.5

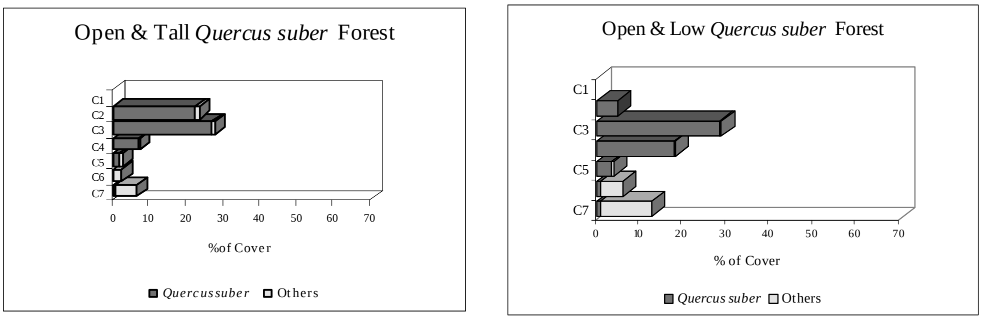

Figure 2. Vertical structure of open formations of *Quercus suber* forests.

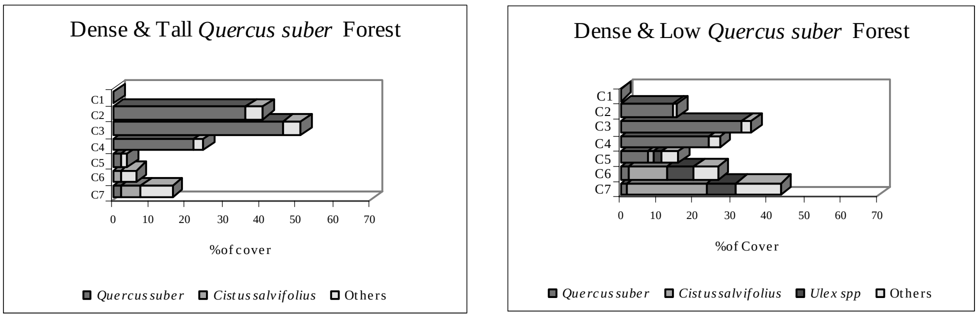

Figure 3. Vertical structure of dense formations of *Quercus suber* forests.

the ground level to 16 metres and, as an invader species, it spreads in open and tall maritime pine, umbrella pine, and eucalyptus stands (Table 1).

The forest types *Quercus suber* forests, *Pinus pinaster* forests, *Eucalyptus* forests and Other forests reveal a diverse vertical structure in their geographical distribution. To better characterise these forest types they were subdivided into four different groups according their vertical structure: open and tall formations, open and low formations, dense and tall formations, and dense and low formations.

Quercus suber forests

The *Quercus suber* forests are distributed mainly in the south of Portugal and they represent 8.5% of the forest area. The open formations of *Quercus suber*, representing 20% of this major forest type, are “montados” where cork oaks never reach 16 metres and the sparse understory is never higher than 1 metre, and composed of *Cistus salvifolius* and *Ulex spp* (Figure 2).

Dense formations reveal an overstory composed of cork oak sometimes mixed with *Quercus rotundifolia* and *Pinus pinea*. Dense and tall formations comprise 28% of this forest type and show an understory composed mainly of *Cistus salvifolius*, *Ulex spp* and *Erica spp*. Dense and low formations are more abundant, corresponding to 52% of the *Quercus suber* forests area; they have a denser understory composed of *Cistus salvifolius* and *Ulex spp*. *Cistus ladanifer* and *Lavandula spp* are also significant (Figure 3).

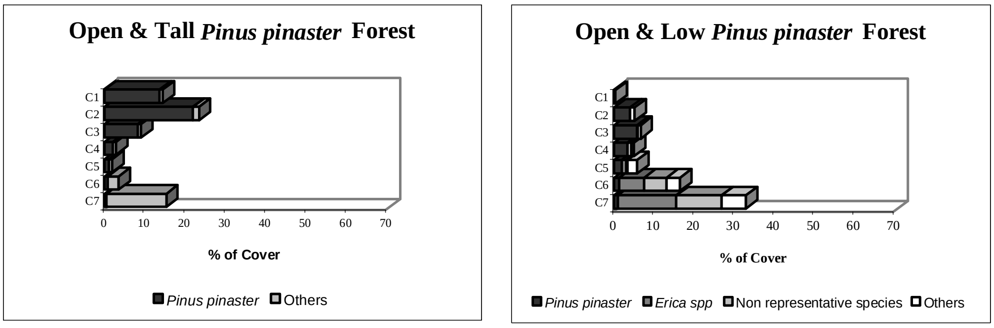

Figure 4. Vertical structure of open formations of *Pinus pinaster* forests.

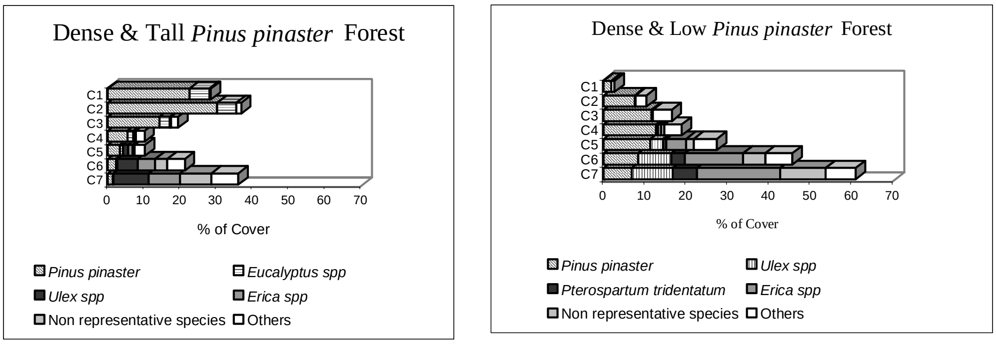

Figure 5. Vertical structure of dense formations of *Pinus pinaster* forests.

Pinus pinaster forests

Open and tall formations occupy 10% of this forest type and they are distributed in large patches mainly in the centre of Portugal from the coast to the interior. A significant large patch can be observed south of Lisboa at the Península de Setúbal. Some of these formations are mixed stands of maritime pine and eucalyptus. The understory is mainly composed of *Erica spp* and *Ulex spp*. Open and low formations represent only 4% of this forest type. They are distributed in the interior north and centre of the country, and south of the River Tejo with a large stand. The maritime pine overstory sometimes contains eucalyptus and the understory is composed mainly of *Erica spp*, *Ulex spp*, *Pterospartum tridentatum* and other plants (Figure 4).

Dense and tall formations of *Pinus pinaster* represent 32% of this forest type, distributed mainly in the north and centre, from the coast to the interior, and on the southwest coast of the country. Stands show a dense overstory of maritime pine, sometimes mixed with eucalyptus and other oaks. *Ulex spp, Erica spp, Cytisus spp, Pterospartum tridentatum* and other plants compose the understory. Dense and low formations characterise 54% of this forest type, mainly distributed in the north and centre, from the coast to the interior, spreading to the south of Portugal. The overstory of these stands is composed of maritime pine sometimes mixed with eucalyptus and cork oak. The dense understory is composed of *Erica spp, Ulex spp*, *Pterospartum tridentatum* and other plants (Figure 5).

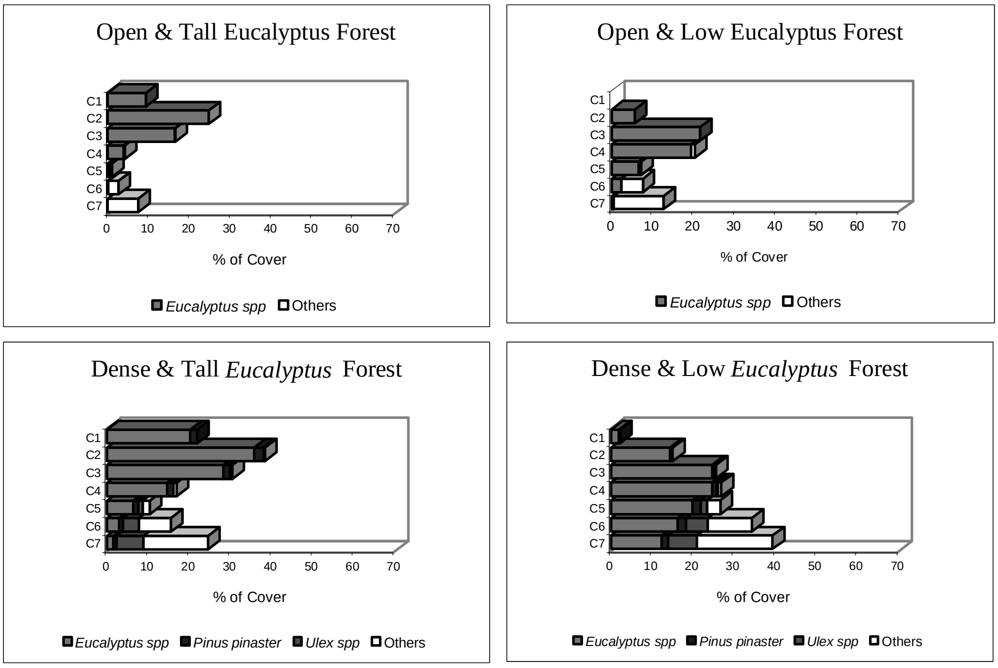

Figure 6. Vertical structure of *Eucalyptus* forests.

Eucalyptus forests

All the *Eucalyptus* forest formations follow the same geographical distribution pattern: from north to south generally near the coast, spreading to the interior in the centre of the country. Open formations, representing 22% of this forest type, show an overstory mainly composed of *Eucalyptus spp* and an understory of *Erica spp* and *Ulex spp* (Figure 6).

Dense formations represent 76% of *Eucalyptus* forests. Both tall and low formations show an overstory mainly composed of *Eucalyptus spp*, sometimes mixed with maritime pine. The tall formation understory is composed of *Ulex spp*, *Erica spp*, *Pterospartum tridentatum* and other plants. The low formation understory is denser and composed of *Ulex spp*, *Erica spp*, *Pterospartum tridentatum*, *Cistus ladanifer*, *Cistus salvifolius* and other plants (Figure 6).

Other forests

This main forest type covers a variety of situations representing a total of 30.7% of Portuguese forest. Open and tall formations, representing 25% of this forest type, are open stands of *Quercus suber* and *Quercus rotundifolia*, called “montados de sobro e azinho”, sometimes mixed with eucalyptus and maritime pine, with an understory mainly composed of *Ulex spp*. Open and low formations represent 50% of the Other forests. They are stands of *Quercus rotundifolia*, called “montados de azinho”, sometimes mixed with eucalyptus and maritime pine. The understory is mainly composed of *Ulex spp*, *Cytisus spp* and *Cistus spp* (Figure 7). Open formations are distributed in the south of Portugal.

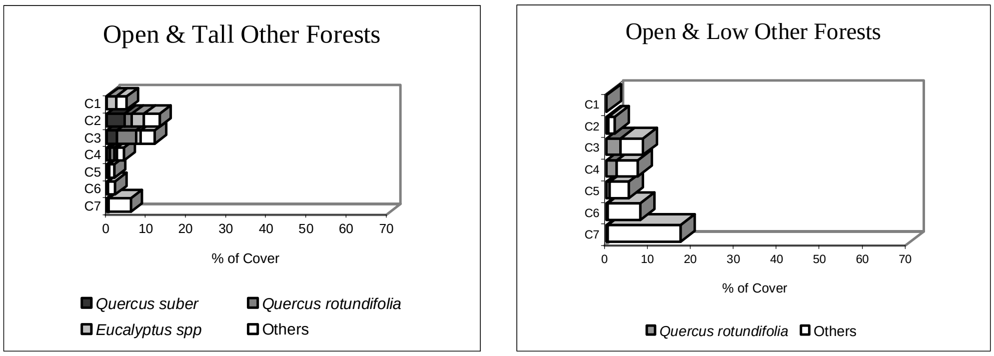

Figure 7. Vertical structure of open formations of Other Forests type: “montados de sobro e azinho” and “montados de azinho”.

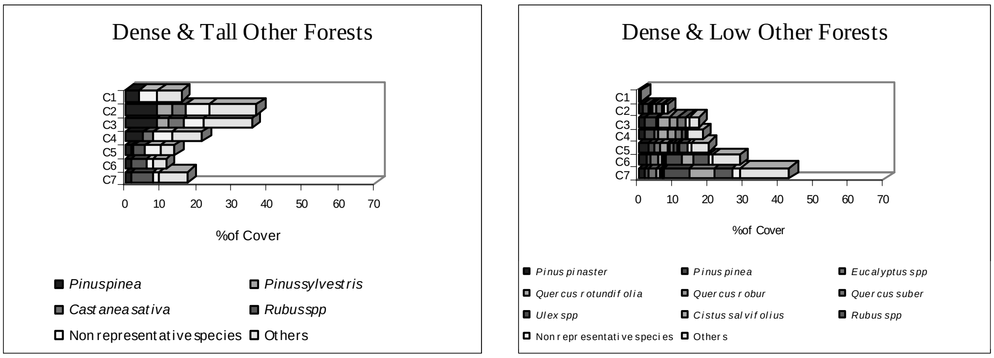

Figure 8. Vertical structure of dense formations of Other Forests type: umbrella pine and chestnut woods, and the very diverse Portuguese forest.

Dense and tall formations correspond to 4% of this forest type. They are stands of *Pinus pinea*, *Castanea sativa* and *Pinus sylvestris*, showing high diversity when mixed with maritime pine, eucalyptus and oaks. Their understory is composed mainly of *Rubus spp*, *Ulex spp* and other plants. Dense and low formations represent 21% of this forest type. They are very diverse stands with an overstory mainly composed of maritime pine, umbrella pine, eucalyptus, cork oak, holm oak and other trees. In the dense and diverse understory mainly composed of *Ulex* spp, *Rubus* spp, *Cistus salvifolius*, important species of *Quercus coccifera* and *Cytisus spp* can also be found (Figure 8). Dense formations are distributed from north to south of the country.

4. Conclusions

The map produced according to the forest types is a powerful tool to better understand the spatial distribution of vegetation structural diversity. The relationships between compositional and vertical structure of forest formations and the vertebrate and invertebrate fauna diversity can be studied. Research on the influence of these forest types on the distribution of reptile and amphibian richness in the Portuguese mainland is in progress. The good correlations obtained by this work suggest how sensitive this tool is. It can be applied to other biodiversity subjects and forest issues such as the evaluation of above-ground biomass, CO_2 sequestration, fuel loads and fire hazards.

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Biodiversity Conservation in Mediterranean Forest Ecosystems: from Theory to Operationality

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Abstract

The analysis of biodiversity in forest systems, and in particular of its interactions with management, must consider processes that cross multiple time and space scales. Silviculture and management affect biodiversity in many different and interrelated ways that are often difficult to quantify. The "biodiversity question" is usually confronted according to the classical scientific paradigm, i.e. the problem is reduced to singular aspects, and the solution is sought in terms of appropriate silvicultural treatment and utilization techniques. But the complexity of the "biodiversity question" does not allow simplistic solutions. The lack of coherence between theory and practice is the cause of the conflicts that very frequently surround forest management.

This is particularly true in the Mediterranean forest ecosystems where biodiversity is often the result of the long interaction of human activity with the environment. Several keystone concepts – relating to the scientific method, the relationship between classic silviculture and forest simplification, the time-space dimension and the value of biodiversity – are discussed with the aim of proposing silvicultural and management guidelines for the conservation of biodiversity in the Mediterranean forest ecosystems. Since there can be no good practice if there is no solid supporting theory, we conclude that sustainable forest management must be based on the theory of complexity.

Keywords: complexity, scientific paradigm, classic silviculture, systemic silviculture, sustainable forest management.

1. Introduction

Today it is universally accepted that the conservation of biodiversity is essential for sustainable forest management. Silviculture and management influence biodiversity in many

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different and interrelated ways that are difficult to quantify. This is particularly true in the Mediterranean forest ecosystems that are the result of the very long coevolution of society and the natural environment.

Here we review the theoretical assumptions that are at the base of forest management and discuss how these assumptions relate to biodiversity conservation in forest systems. This is because we believe progress in this field is possible only if practice is based on a solid theoretical framework. The lack of coherence between theory and practice is the cause of the conflicts that very frequently surround forest management. In brief, there can be no good practice if there is no solid theory behind it.

2. The forest: a sum of merchantable trees

Scientific research has generally confronted the biodiversity question according to the classic scientific paradigm where the problem is reduced to single components. With the aim of maintaining sustained wood production, classic silviculture and management have caused the simplification and uniformity of the forest: starting from the pre-scholastic period the forest has been viewed as a sum of merchantable trees, and it is still managed along these lines today.

Classic silviculture is based on the control of natural processes by means of cultivation techniques; silvicultural methods aim at forest regeneration according to a predefined model. Classic forest management is based on the continuity of production and on the theory of the regulated forest. The aim is a forest – that some consider ideal, others optimal – with a "regular" structure, without "abnormalities" in density, increment, and age classes for even aged stands, or size classes for uneven aged stands. A forest where all the variables are controlled (Ciancio et al. 1994: 1995).

In the cultivated forest one or more species are favoured, depending on particular characters, such as productivity, growth rate, quality and quantity of wood production, sprouting capacity, etc. Cultivation influences the structure of the forest, regeneration processes, distribution in age classes and, especially, maximum age (in relation to rotation or maximum felling diameter). Furthermore it can cause the fragmentation of habitats, the loss of organic substance from the soil and the onset of erosive phenomena.

The simplification of forest systems does not involve only the number of species, it also impacts the variety of structures and processes at different scales, from the stand to the landscape. In forest ecosystems the most evident symptoms of simplification are difficulties in natural regeneration and the reduction in the variety and quality of habitats (Nocentini 2000). Other, less obvious but still negative effects must also be taken into consideration, such as modifications in the geochemical processes and the alteration of soil micro-flora and micro-fauna.

In brief, classic silviculture and forest management have looked at the forest only as a sum of merchantable trees.

3. The forest: a list of species

Scientific progress in the last few decades and the awareness of the importance of biodiversity for the quality of life have brought about a different approach towards the forest. Thus biodiversity has become the catchword of any discourse on forest management.

Commonly, biodiversity is defined as the number and variety of species and silviculture is considered appropriate if it takes into account a series of indicators. But how is this concept interpreted and how does it risk impacting practical forest management?

The question is particularly interesting because it is a perfect example of the difficulty in arriving at a consistent and widely accepted definition of sustainable forest management.

The role of biodiversity in the functioning of forest ecosystems in relation to management is generally reduced to the impact that the diverse cultivation techniques have on a few parameters or indicators, such as the variation in the number of species of particular interest.

Following this logic, that in fact reduces the problem to an extremely limited number of variables, it is believed that once the relationship between cultivation techniques and ecological requirements of the various species is known, or when, for example, the "required thresholds" are met, then management will be able to deterministically sustain biologically diversity.

In other words, the forest is increasingly seen as a list of species or as a set of indicators. This approach is misleading for a series of reasons that we will try to explain.

First of all, as Marcot (1997) has written, unless radically simplified way beyond natural conditions, forest ecosystems are still too complex and unpredictable to allow for an accurately deterministic management of all their components.

In the second place, the history of each forest is dominated by processes that cross multiple space and time scales: the analysis of biodiversity in forest ecosystems and in particular of its interaction with management must consider processes that cross more dimensions. This is particularly important in understanding how diversity contributes to the resistance and resilience of ecosystems. When different species play similar roles at very different scales there is a functional strengthening of the ecosystem that increases its capacity of recuperating after disturbance factors (Holling et al. 2002): this redundancy is precisely what connects the diversity of an ecosystem to its functionality.

On the practical side the reductionist approach consists of a series of technical adjustments with the aim of reducing the impact of forest management. An example: the need to protect the diversity of species and processes supported by old, decaying and dead trees is answered by the prescription to allow a certain number of trees to grow old and die or to leave a certain number of cubic meters of dead wood per hectare, *without in any way changing the reference models of silviculture and management*. But in this way the fact that the reference models of classic silviculture and management are exactly what cause the simplification of forest ecosystems is completely ignored. In the end this approach does not resolve the lack of coherence between the theoretical reference paradigm – the forest reduced to a list of species or a set of indicators – and the aim – biodiversity conservation. Finally, the reductionist considers the forest as a closed system, and takes no account of interactions with other systems such as the social system, the economic system or the cultural system.

4. The forest: a complex biological system

If we accept the fact that the forest is a complex, adaptive and resilient biological system, then it is indeed possible to go from theory to practice following management guidelines that are truly coherent with the aim of conserving biodiversity. This is the only way the great number of research projects on quantifying and monitoring biodiversity can make a truly useful contribution to forest management.

The key aspects of the paradigm we are proposing are:

1. the awareness that the forest reacts to any natural event or human action with a series of reactions that are a synthesis of interactions and interconnections (Ciancio and Nocentini 1996; 1997): this means accepting the fact that processes seem linear and states appear constant, and thus *predictable*, only in a limited space and time scale (Mladenoff and Pastor 1993);
2. the awareness that the value of the forest is more than the sum of the various functions that are assigned to the forest from time to time (production, soil protection, biodiversity, carbon sequestration, landscape enhancement, etc.): this means recognizing that the forest has *intrinsic value*.

To obtain the overall functionality of forest systems and truly guarantee biodiversity conservation it is necessary to favour the rehabilitation of natural processes, i.e. the natural self-regulating and self-perpetuating mechanisms of the system that increase its resistance and resilience. Biodiversity must be connected to complexity.

In practice, this means adopting systemic silviculture, i.e. non-linear, extensive silviculture, based on the principle of autopoiesis (Ciancio and Nocentini 1996). This means maximizing the contribution of natural energy to the functioning of the system and minimizing artificial energy inputs (Allen and Hoekstra 1992). This means working together with, and not against, the natural self-regulating processes. *Forest management is sustainable if, and only if, it does not reduce the ecosystem's resiliency.*

Management must be based on constant control of the reactions of the system to cultivation, following an adaptive approach, based on the scientific method of trial and error. This means that management does not tend towards a predetermined model, but instead acts according to a coevolutionary continuum between cultivation and reaction of the ecosystem with the aim of increasing the system's capacity of self-regulation and self-perpetuation. In this approach, indicators are useful in the monitoring process that must sustain adaptive forest management, but not for predetermining "optimal" levels, e.g. of deadwood or other biodiversity indicators.

5. Biodiversity conservation the in Mediterranean forest ecosystems

The countries around the Mediterranean basin are characterized by a long history of heavy exploitation of natural resources. The forest has disappeared over vast areas or has been relegated on the poorest soils. In the first case, in many areas, important reafforestation projects carried out during the last century have produced even aged pure conifer stands. In the second case the original forest has been often transformed into coppices. On the best sites, where cultivation has maintained high forest stands, the application of silvicultural models oriented towards maximizing wood production has generally favoured even aged structures and the prevalence of one or few tree species (Nocentini 2001).

But the long interaction of man and the environment has also produced landscapes that are rich in a diversity which today has not only aesthetic value but also cultural and anthropologic relevance.

Biodiversity conservation in the Mediterranean forest ecosystems must therefore be based on a strong action aimed at:

1. preserving the last remnants of the primeval forest;
2. renaturalising forest systems that have been simplified by past management;
3. maintaining traditional forms of forest use where these are truly a part of the local culture and traditional knowledge.

It is especially in managing simplified forest systems that biodiversity conservation must be based on the coherence between theory and practice: here management must not be oriented towards recreating a supposed “natural” state, but should sustain natural evolutionary processes that tend to increase the system’s diversity and complexity.

6. Biodiversity conservation and Lewis Carroll’s universe

There is no doubt that biodiversity must be appraised through the space-time dimension. In purely scientific terms only two models are known: Newton’s universe and Einstein’s universe. In the first time is absolute and space is relative; in the second space and time are relative.

However, there is yet another model, Lewis Carroll’s universe. In this universe, space is absolute and time is relative. In the world of Carroll the Queen of Hearts tells Alice: “run, run as fast as you can only to stay in the same place”. With regard to biodiversity conservation, technique may run, run, but in the end it remains in the same place: space is absolute and time is relative. The future of the forest is unpredictable: it depends on the changing of time.

Allow us a metaphor: if a lady first wears a Dior dress, then a Versace gown and then an Armani suit, the style changes, but she is still the same person. Thus, the forest has been interpreted in different ways in different time periods: as a sum of merchantable trees, as a list of species or a set of indicators, now as a carbon sink and so on. But the forest does not need more or less fashionable dresses. What is important is to capture its essence – that of a complex biological system.

7. Conclusions

Many of the things that have been discussed above are also mentioned in these conference proceedings, albeit separately and leading to different conclusions and mostly to new questions. What seems to be lacking is a solid theory on which to build coherent management practices. We believe that sustainable forest management must be based on the theory of complexity.

Normally, scientific explanations are sought through analysis, which is an expression of continuum, and not by examining discontinuum that is the heritage of all ecosystems. One of the mainstays of the positivist culture has failed now that we have understood that complexity cannot be interpreted and explained with a scientific language based on predictability and the linear relationship between cause and effect. The explanation of forest ecosystem complexity and biodiversity will be convincing only when a different scientific language will be constructed and used.

In the meantime, we must be aware of our ignorance and act responsibly: this, ultimately, is the only idea of science we can conceive.

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Session 3: Stand-Level Indicators: Dead Wood

Characterisation of Coarse Woody Debris in Two Scots Pine Forests in Spain

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Abstract

Coarse Woody Debris (CWD) has become an important component in the study of forest ecosystems, being a key factor in the nutrient cycle as well as a habitat for many species. CWD dynamics varies greatly from primeval to managed forests. The aim of this study is to further our knowledge regards the changes in CWD dynamics resulting from the silviculture applied in Scots pine forests.

A chronosequence trial was established in two Scots pine forests in the Central mountain range in Spain. Although the shelterwood system has been applied in both forests, one has received more intensive silviculture, whereas in the other, regeneration has been much more gradual and the thinning regime has not been so intensive. In order to study CWD, eight categories were defined according to size and CWD type (dead standing trees, stumps, logs, fallen branches and litter), and five rot classes (from sound dead wood, to wood soft with the outer surface being difficult to distinguish). The volume of branches and logs (estimated from length and mean diameter) and the number of stumps by size class and rot class are used to characterise the CWD. The size and number of plots used for each class of CWD depends on the scale of the spatial arrangement of the items, ranging from 5000 m^2 for the larger standing trees, stumps and logs to 1 m^2 where litter was weighed and its thickness measured.

The most notable differences between the two silvicultural systems are peaks in the graph for temporary distribution of larger logs and stumps when intensive silviculture is applied. The CWD observed in the forests studied is mainly produced by logging, the maximum volume of logs and branches above 5 cm of diameter is 43.25 m^3/ha after regeneration felling whereas in Valsaín reach a maximum of 16.30 m^3/ha at 60 years, when the stand has just been thinned. The large stumps mean an important fraction of CWD biomass in these forests (about 15.00 Mg/ha just after the regeneration felling).

Keywords: Coarse Woody Debris, Scots pine, forest structure, forest dynamics.

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Introduction

Coarse woody debris plays an important role in forest ecosystems. It is not only a key factor in the nutrient cycle (Harmon et al. 1986) but also the habitat for many animals, plants and fungi (Similä et al. 2003; Bissonette and Sherburne 1993; Sippola and Renvall 1999; Ferris et al. 2000). CWD can therefore be a structural indicator, as a surrogate for decomposition processes and habitat availability (Ferris and Humphrey 1999). Certain aspects of CWD are well known characteristics of old-growth forests (Siitonen et al. 2000).

Coarse woody debris refers to dead wood, such as logs or branches on the ground, stumps and snags or dead standing trees, which go through a complex decomposition process. The minimum size considered coarse debris varies from 2.5 cm in diameter (Harmon 1986) to 10 cm in diameter (Spies and Franklin 1988). In managed forests the volume of CWD is between 2–30% (normally <10%) of the quantity found in unmanaged forests (Siitonen 2000). One of the main objectives of the new, biodiversity orientated forestry is to reduce this difference.

The amount of CWD in any stand is related to the incorporation rate, the decay process, which in turn depends on species, site conditions and exposure (Harmon and Hua 1991), slope (Rubino and McCarthy 2003) as well as other physical and chemical agents which may be present. Forest management changes the amount and quality of downed woody debris (Fraver et al. 2002). Interventions such as harvesting or pruning not only mean the extraction of woody material before it starts to decay, but also the loss of decaying ground wood caused by machinery during the work. In the Mediterranean forest ecosystems, dead woody material has traditionally been removed to reduce the risk of fire. In order to conserve the biodiversity, it is important to maintain adequate quantities of deadwood with varying characteristics and at different stages of decay. To achieve this aim, it is necessary to understand the dynamics of dead wood as well as the quantitative and qualitative fluctuations during the stand life. Although several studies deal with these key factors (Tinkter and Knight 2001; Ranius et al. 2003), much work still needs to be carried out. The next step is to integrate CWD into models with a view to optimising both timber yield and the conservation of ecological values under economic constraints, as in Wikström and Eriksson (2000).

The aim of this study is to analyse the structure and dynamics of CWD during the life cycle of Scots pine stands, and the effects of the silvicultural treatment applied. CWD has been characterized in different age class stands in two Scots pine forests in the Central mountain range in Spain.

Material and methods

Study site

Pinar de Valsaín and Pinar de Navafría are two Scots pine forests on the northern face of the Sierra de Guadarrama in the Central mountain range in Spain. Altitude ranges from 1200 to 2268 m. Annual rainfall is over 1190 mm. These mountains are composed of granite and gneiss, giving rise to acid soils (Rojo and Montero 1996). Pinar de Valsaín has had a management plan since 1889, Navafría since 1899. In both forests, the uniform shelterwood system was initially used in permanent blocks with a 20-year regeneration period (Donés 1994; García 1994) and in Navafría this management system is still used. The trees in the regeneration block are removed in two phases, leaving 30 to 40 trees/ha after the first felling, which is followed by soil preparation and seedling plantation. An intensive thinning regime is also applied mainly in the younger stages of the stand. Rotation is 100 years. In Valsaín the

Table 1. Characterisation of stand structure within experimental plots (extent 0.5 ha).

	Navafría					Valsaín					
Plot	1	2	3	4	5	1	2	3	4	5	6
Age (years)	10–30	30–50	50–70	70–90	90–110	0–20	20–40	40–60	60–80	80–100	100–120
Dbh (cm)	6.9	14.1	32.9	40.4	42.5	3.7	11.9	20.5	30.5	34.6	38.3
Hm (m)	5.9	12.0	22.0	20.5	22.1	3.7	11.4	16.7	23.5	22.2	24.0
G (m^2/ha)	27.8	41.8	60.6	48.2	44.2	34.4	41.0	48.5	53.3	54.0	46.3
Nº trees/ha	3261	1244	340	182	152	4139	1316	661	343	275	192

Dbh: diameter at breast height, Hm: mean height, G: basal area.

management system has changed several times (Montes et al. 2003). Since 1988, the group shelterwood system has been applied in ‘floating periodic blocks’. The uniform shelterwood felling at Valsaín comprises more than two phases and regeneration is allowed to take place naturally. Thinning is carried out when the stand has reached the stem exclusion stage. Rotation in Valsaín is 120 years, which has not changed since the management plan of 1889. A plot of 0.5 ha was established for each age class (0–20 years old, 20–40, 40–60, 60–80, 80–100 and 100–120) in a stand with a site index of approximately 23 m at an age of 100 years (Rojo 1996). The stand characteristics in each plot are synthesized in Table 1.

Methods

In order to study CWD, 8 categories of elements (Table 2) and 5 rot classes (Table 3) were defined. The most common method for rot classification identifies five classes based on observable characteristics (Pyle and Brown 1998; Siitonen 2000). This qualitative classification is important because the different wood-inhabiting organisms are adapted to different elements of different sizes (Jonsell et al. 1998), usually in a determined decay class (Söderström 1988). Branches and logs are classified into size (diameter at mid point) and rot classes and volume is estimated from length and midpoint diameter (Grove 2001). For pieces falling partially outside the boundary of the plot, only that portion lying within the plot was measured. The stumps are similarly summarised by number per size class and rot class. Two perpendicular diameters at 0.30 m height and two perpendicular diameters at breast height (*dbh*) were measured from a sample of 150 living trees within the 0.5 ha plots, in order to estimate the stump and main root biomass (*Br*) using the regression model proposed by Montero et al. (2003) for Scots pine in Valsaín forest (eq. 1).

$$\ln(Br) = -4.56044 + 2.62841 \cdot \ln(dbh) \quad R^2=0.972 \qquad (1)$$

Pieces of branch smaller than 5 cm in diameter and litter are quantified by dry weight. The overall sample was weighted in field and a subsample of each category was collected and weighted after oven-dried at 80ºC to constant weight to obtain dry weight. The dry weight of the sample was calculated by multiplying its wet weight by the ratio of dry weight to wet weight of its respective subsample.

Table 2. Characteristics of wood and subplots size of each fraction of coarse woody debris (CWD) considered.

CWD type	Size	Subplot size (m^2)	Nº subplots/plot
Dead standing trees	∅≥10cm	5000	1
	10cm≥∅>5cm	100	4
Stumps	∅≥30cm	5000	1
	30cm>∅≥5cm	100	4
Logs and branches fallen in the floor	∅≥10cm	5000	1
	10cm>∅≥5cm	100	4
	5cm>∅	25	4
Litter	-	1	16

Table 3. Rot classes of coarse woody debris.

Rot class	Stage of decomposition
I	Freshly dead, at most one year old
II	Wood hard, bark partly loose but >50% remaining
III	Wood hard or soft in the surface, <50% of the bark remaining
IV	Wood soft in the surface or throughout
V	Wood soft throughout or with a hard core only, the outer surface hard to distinguish or completely or partly covered by forest floor mosses

The size and number of inventory plots were determined according to the abundance and distribution of each category (Table 2). The larger size classes; standing dead trees, logs, branches and stumps were measured throughout the 0.5 ha plot. Within this 0.5 ha plot, four subplots of 100 m^2 were set to sample the smaller classes. Branches less than 5 cm in diameter were collected from smaller, 25 m^2 subplots situated within each 100 m^2 plot, in which soil cover data from a 1 × 1m grid had been taken prior to the inventory. Litter was collected in four 1 m^2 subplots nested in each 25 m^2 plot.

Results

The dynamics of fallen logs and branches are analysed in Figures 1, 2 and 3. The effects of intensive management lead in Navafría to a 'saw-tooth' graphic distribution for logs and branches over 10cm in diameter across the successive age classes at 20-year intervals (Figure 1a). Two plots had large amounts of CWD in this category, 13.50 m^3 in the 60 year plot, 74% belonging to the rot classes IV and V, and 12.97 m^3/ha at the age of 100 years, 95% within rot classes I, II and III. Some wood in late decomposition stages remained in the plot located in the 20-year stand, whereas the other plots had a notable lack of logs and branches in this size class. In Valsaín (Figure 1b) the volume distribution reached a maximum of 6.10 m^3/ha at 60 years, but much lower levels than for Navafría were found in the last age classes of the rotation.

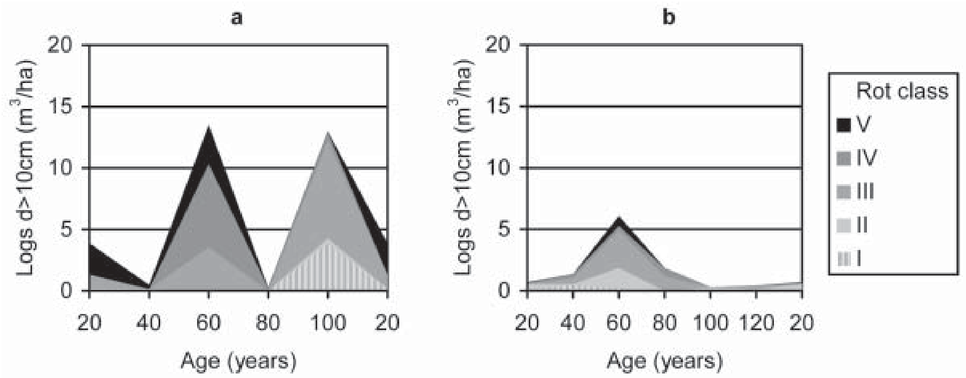

Figure 1. Amount of logs and branches fallen on the floor of diameter (*d*) over 10 cm across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

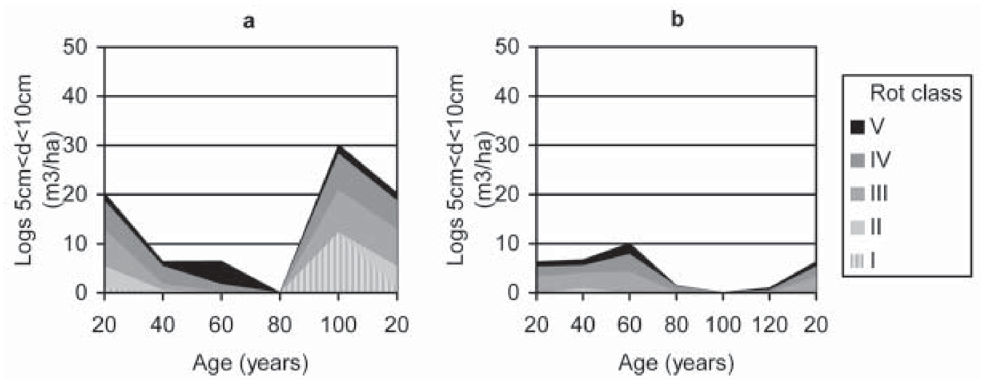

Figure 2. Amount of logs and branches fallen on the floor of diameter (*d*) between 5 and 10 cm across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

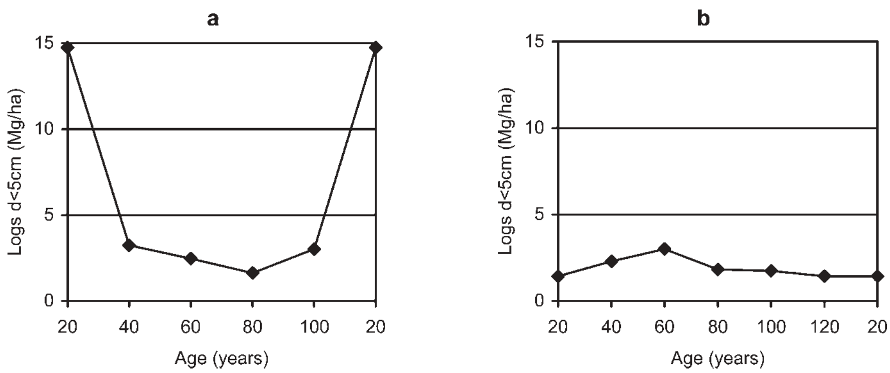

Figure 3. Amount of branches fallen on the floor of diameter (*d*) lesser than 5 cm across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

Logs and branches between 5 and 10 cm in diameter show a "U-shaped" pattern in both forests (Figures 2a and 2b), in Navafría the maximum is reached in 100 years plot (30.28 m^3/ha). In Valsaín the levels reached after regeneration cuttings are maintained until the stand is 60 years old and quantities are less than in Navafría (10.20 m^3/ha). In the case of those branches with a diameter less than 5 cm, the pattern is flat in both forests (Figure 3a and 3b) except for the 20-year plot in Navafría, which was completely covered (14.66 Mg/ha) with biomass resulting from precommercial thinning.

The dynamics of stumps are analysed in Figures 4 and 5. The number of stumps over 30 cm in diameter gives a "U-shaped" pattern both in Navafría (Figure 4a) and Valsaín (Figure 4b). In Navafría maximum is reached at 100 years plot, where there are 194 stumps per ha, most in decay classes I and II. This amount of stumps yield approximately a biomass of 14.43 Mg/ha of stump and main root system, estimating the biomass of each stump with eq. 1. The 20-year plot only had stumps in decay class V and large stumps had disappeared in the 40 year plot. In Valsaín we found large stumps of all decay classes in every plot, giving rise to a much more continuous pattern, ranging from approximately 15.08 Mg/ha to 3.46 Mg/ha. The number of stumps between 5 and 30 cm in diameter peak at 40 years in Navafría (Figure 5a) and at 60 years in Valsaín (Figure 5b), being 0.22 Mg/ha and 0.38 Mg/ha the respective biomass estimation.

The amount of litter on the forest floor shows an increasing trend across the successive age classes up to the regeneration felling (Figure 6). Biomass of litter ranges from 7.73 Mg/ha to 18.72 Mg/ha in Navafría and from 4.41 Mg/ha to 10.06 Mg/ha in Valsaín. In Navafría the trees are quite spaced in the greatest age classes, favouring the growth of a herb layer over the soil (Figure 7a). In Valsaín, where stands are kept very dense during the earlier development stages, part of the soil is covered by mosses (Figure 7b).

Discussion and concluding remarks

This study underlines the significant relationship between CWD dynamics and the silvicultural system applied. We chose an areal sampling method, but several other useful methods such as line intersect sampling (Ringvall and Stahl 1999) or multistage point relascope sampling (Gove et al. 2002) have been developed to study CWD.

Several studies have described a "U-shaped" temporal pattern for the amount of CWD in even-aged forest chronosequences (Sturtevant et al. 1997; Ranius 2003). In our chronosequences, under two different silvicultural systems, this pattern only appears for some types of CWD. The "U shaped" pattern is due to the low level of input in the younger stages as the decay process leads to a decline in accumulated debris, whereas the input tends to increase from when the stand reaches the stem exclusion stage. In managed forests the input depends on the silvicultural system and the way in which it is applied. This not only affects the frequency and quantity, but also the quality of CWD accumulation. The effects of silviculture lead to a disturbed pattern with peaks of accumulation of certain types of CWD after thinning or regeneration felling. The 'saw tooth' pattern found in the case of large logs and branches as well as in the distribution of large stumps during the life cycle of trees in the Navafría forest may lead to changes in habitat availability. The presence of all developmental stages throughout the forest means the temporal continuity of habitat availability, but the location of these habitats changes periodically. Some key structural features of natural forest, such as snags or large fallen trees, are replaced by others which are typical of managed forests, such as stumps. In the Valsaín forest, where the regeneration felling is more gradual and the thinning regime less intensive, there is an input of larger stumps from the residual

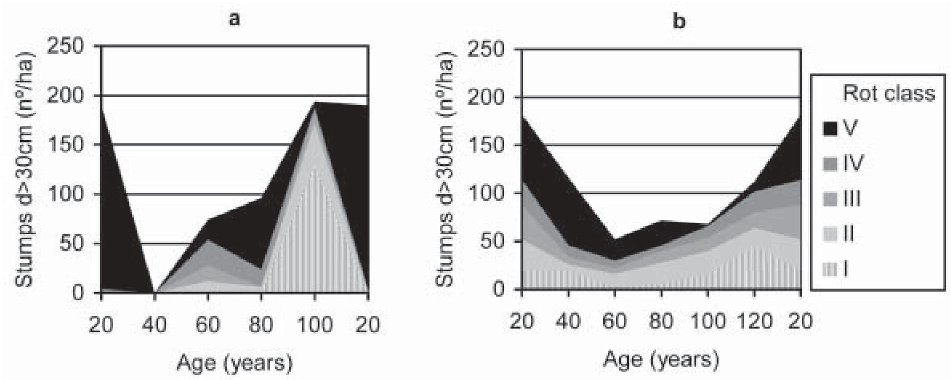

Figure 4. Amount of stumps of diameter (*d*) over 30 cm across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

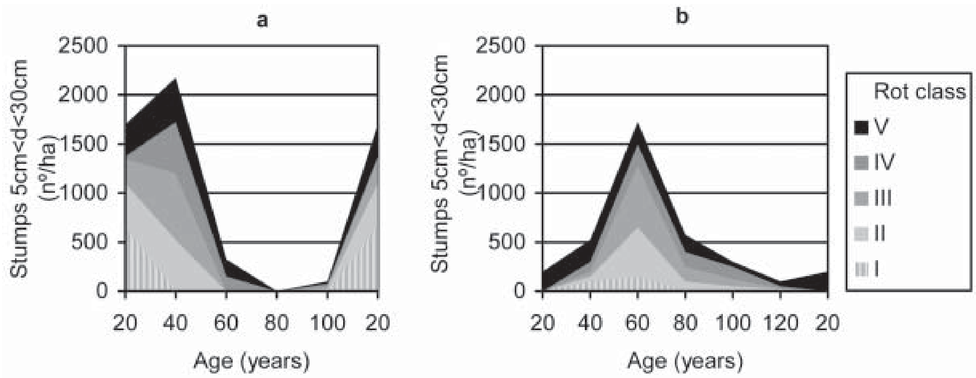

Figure 5. Amount of stumps of diameter (*d*) between 5 and 30 cm across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

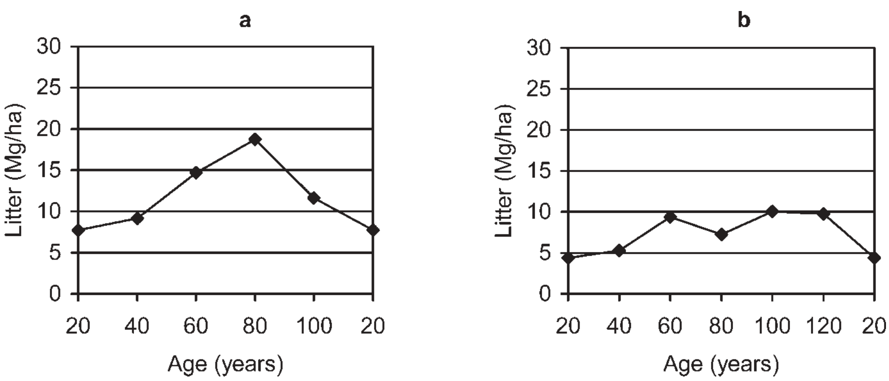

Figure 6. Amount of litter on the floor across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

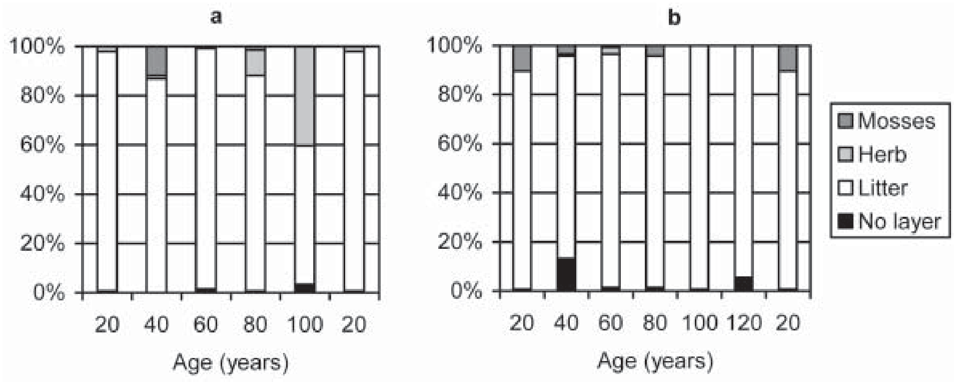

Figure 7. Percentage of litter, herb and mosses in the ground cover across the successive age classes of 20 years wide in Navafría (**a**) and Valsaín (**b**).

upper strata during the younger stages. After a decline in the abundance of stumps at intermediate ages, the trend increases towards the end of the life of the stand, leading to a "U-shaped" temporal pattern. Nevertheless, the main natural accumulation of CWD would arise long after the period covered in the study (Spies 1988). The CWD observed in the forests studied is mainly produced by logging and the volume of logs and branches above 10 cm of diameter, although is fairly small when compared with coniferous forests of North-west of U.S.A. (Spies 1988), where volume of logs ranges from 77 m^3/ha to 346 m^3/ha, it is similar to the amount reported by Ranius (2003) for Central Sweden managed forests of Norway spruce (9.7 m^3/ha). In Valsaín the distribution of larger logs and branches peaks at stem exclusion stage, when thinning was first carried out, but the extraction of firewood reduced the presence of this source of dead wood, especially in mature stands. Firewood extraction must be taken into account when modelling CWD dynamics since it reduces the accumulation of biomass after a man-made or natural disturbance (Carmona, et al. 2002).

Most of the logs and branches of 5–10 cm in diameter are the result of regeneration felling in Navafría forest. Naturally deposited logs and branches of this size are not common because of early thinning and pruning. However, such thinning is not common practice in young stands at the Valsaín forest, so there is a certain input of intermediate sized logs and branches until the stem exclusion stage, when the first thinning is carried out. There was a notable lack of information about CWD of less than 10 cm in diameter, known as fine woody debris or FWD (Currie and Nadelhoffer 2002), because most studies tend to focus on the larger pieces of wood debris.

Small stumps (less than 30 cm in diameter) are generally the result of thinning. In Navafría there are a considerable number of small stumps in the first two age classes plots, whereas in Valsaín, an upward trend starts at the second age class and reaches a maximum at the third age class. Nevertheless, except for 40 aged plot in Navafría, where there are not large stumps at all, small stumps means a small fraction of total stumps biomass.

Our data suggest that the decay process varies in the different categories of CWD, and that decay rates differ between the two forests. Stand density is one of the probable factors determining decay rate, whilst the frequency of man-made disturbances, which partially destroy accumulated CWD, may reduce the time spent in the decay process (Ranius 2003). The proportion of CWD in decay class V was smaller than would have been expected from the results obtained in other studies (Grove 2001). This may be due to the relatively small size of the woody material collected in both pinewoods and to the difficulty in detecting class V CWD, which is often hidden under leaves (Spetich et al. 2002).

The silvicultural system employed and the density of the stand will modify the decay process of ground litter. Dense stands favour the development of mosses, whereas the contrary will lead to the establishment of a herb layer. Not only are these differences important for small invertebrates, but also, the formation of a herb layer can be a serious obstacle to successful natural regeneration.

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Dead Wood in European Forest Reserves – A Reference for Forest Management

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Abstract

The amount of dead wood in managed forest is an important indicator for sustainability and biodiversity conservation and dead wood is one of nine pan-European biodiversity indicators for sustainable forest management. However, the knowledge on 'natural' amounts of dead wood in European forests is very limited. Based on published and unpublished studies from 116 forest reserves we analysed dead wood data in relation to forest types. Dead wood amounts varied with forest type, with typical average volumes ranging from 59 $m^3\,ha^{-1}$ in northern boreal coniferous forests to 216 $m^3\,ha^{-1}$ in mixed mountain forest in central Europe. These values were comparable to dead wood volumes in similar forest types in eastern North-America. In contrast, typical dead wood in managed forests is 2–20 $m^3\,ha^{-1}$ due to wood extraction and short rotations. Main factors influencing dead wood amounts are site productivity, decomposition rate and disturbance regime, which are all included in the concept of forest types. Because forest type plays a major role for potential dead wood quantities, we suggest a differentiated approach to dead wood as an indicator for sustainable forest management.

Keywords: coarse woody debris (CWD), indicators, forest management, sustainability.

1. Introduction

Dead wood is an operational and quantitative stand level indicator. Lately, dead wood has become an important indicator for sustainability and biodiversity conservation in managed forests as one of nine pan-European biodiversity indicators for sustainable forest management. However, we need a reference from natural forests or forest reserves covering major European forest types for the development and application of dead wood as an

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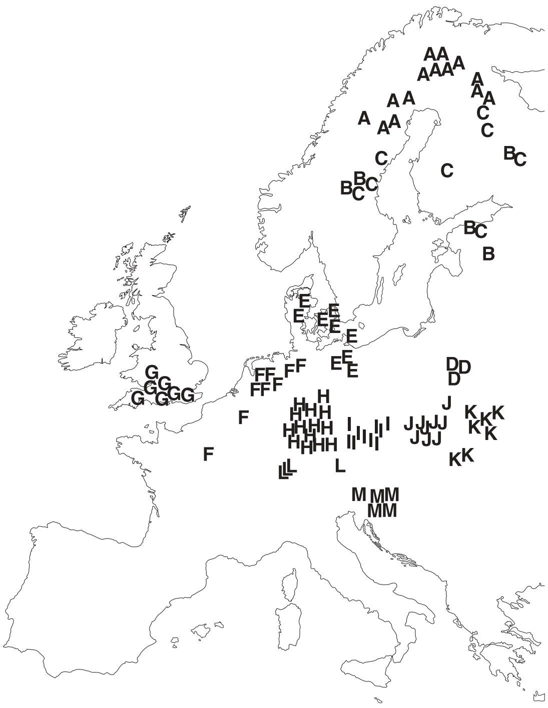

Figure 1. Location of the 116 sites included in the study. The letters on the map (A–M) refer to the different forest types as described in Table 1.

indicator. Presently, the knowledge on the 'natural' amount of dead wood in European forest is fragmented with some forest types being intensively studied whereas others are sparsely researched. Thorough, regional studies of the volume of dead wood in relation to forest type have been done in e.g. Canada (Stevens 1997) but to our knowledge a Europe-wide comparison of dead wood volume and forest types has not yet been carried out. The aim of this study is to analyse dead wood volume in relation to forest types across Europe, to discuss possible explanations of the variation in dead wood volumes among forest types and to provide preliminary estimates of dead wood to be used as a reference for sustainable forest management.

2. Methods and materials

Based on a combination of field research and literature studies we collected information on dead wood volumes in 116 forest reserves and other protected forest areas throughout Europe, representing major forest types (Figure 1). We divided data into 13 forest types (A–M) (see Table 1), based on a combination of geographical and ecological information (Ellenberg 1996; Larsson 2001). Due to sparse information on dead wood in forest reserves on the Iberian Peninsula and in the Mediterranean region, those forest types were not included in the study. We also excluded forest reserves with less than 50 years of strict protection in order to ensure the reference value of the study. However, some compromises were made for lowland Western Europe, where strict long-term forest reserves are sparse. Only dead wood classified as coarse woody debris (CWD) was included in the study, leaving estimates of small-dimension dead wood and dead wood in the form of dead branches or dead cores of living trees out of the dataset. Information on standing volume, stand productivity, main tree species and forest type was combined with the dead wood data. A dead wood ratio was calculated as (dead wood volume (m^3 ha^{-1}) / living wood volume (m^3 ha^{-1}))*100. Information on dead and living wood volumes were analysed in relation to forest types.

3. Results

Naturally, both living volume and dead wood volume varied between as well as within forest types. The living volume ranged from an average low of 217 m^3 ha^{-1} in the northern boreal forests (forest type A) to an average high of 682 m^3 ha^{-1} in sub-montane mixed forests in south-central Europe (forest type I). The dead wood volume generally followed the same pattern, ranging from an average low of 59 m^3 ha^{-1} in the northern boreal forests (forest type A) to an average high of 216 m^3 ha^{-1} in southern sub-montane mixed forests (forest type J) (Table 1). The deciduous forests in Southern England represent an exception with an average high of 256 m^3 ha^{-1}. This extreme is mainly due to recent major storm events in the region.

The overall trend of high dead wood volume corresponding to high living volume per forest type is illustrated in Figure 2. When calculating the difference in dead wood volumes between forest types on an absolute scale, there is an almost 5-fold difference in absolute dead wood values from the lowest to the highest average. This difference is reduced to a 2-fold difference between forest types when calculating the relative dead wood ratio. On average, the relative dead wood ratio ranges between 15 and 37% of living volume depending on forest type (Table 1).

4. Discussion

Our results show that forest type has major influence on dead wood volume in forest reserves, with a gradient from low dead wood accumulation in northern boreal forests to high levels in central-European mixed forest types. The relation between dead wood volume and forest type is however more complex, as different forest types are characterised by different species composition, site productivity, climate, soils and disturbance regimes. Generally, site productivity in combination with a decomposition rate determines the long-term average dead wood supply, whereas the regional or local disturbance patterns cause temporal pulses of dead wood input to the stand (Siitonen 2001).

Site productivity

Site productivity is strongly influenced by climate, soil conditions and water availability. Mean annual increment on poor sites in boreal coniferous forests (forest type A) may be as low as 1-2 m^3 ha^{-1} (Kuusela 1994), whereas mixed deciduous forests on rich soils (forest type E) may have a mean annual increment of 8–10 m^3 ha^{-1} (Henriksen 1988), reaching a high of 16–18 m^3 ha^{-1} in the submontane forest types (I-K) (Mayer 1977).

Table 1. The species composition, living and dead wood volume (average volume, min-max volume, number of sites), and dead wood ratio (dead wood/living wood) in relation to forest types.

Forest type	Main tree species	Living volume (m^3 ha^{-1})	Dead wood volume (m^3 ha^{-1})	Dead wood ratio (%)	References
A: Northern boreal forest, pine and spruce	*Pic, Pin, Bet, Pop.*	217 (138–338) n=10	59 (23–79) n=10	27	Linder et al. 1997, Sippola et al. 2001
B: Southern boreal forest, spruce	*Pic, Pin, Pop, Bet, Aln.*	440 (342–511) n=3	80 (31–133) n=5	18	Köster et al. 2003, Linder 1998, Linder et al. 1997, Rouvinen & Kouki 2002
C: Southern boreal forest, pine	*Pin, Pic, Bet, Pop.*	242 (151–333) n=7	79 (9–155) n=12	33	Karjalainen & Kuuluvainen 2002, Köster et al. 2003, Linder 1998, Linder et al. 1997, Rouvinen 2002, Rouvinen et al. 2002, Rouvinen & Kouki 2002
D: Boreonemoral mixed forest	*Que, Til, Ace, Cor, Pin, Pic*	450 (395–516) n=3	132 (106–151) n=3	29	Bobiec 2002
E: Baltic nemoral forest, beech	*Fag, Que, Fra, Ace, Car, Pin, Til, Pin.*	479 (201–674) n=9	148 (73–234) n=9	31	Christensen & Hahn 2003, Winter 2002 pers. comm., Schmaltz & Lange 1999
F: Alantic nemoral forest, beech and oak	*Fag, Que, Fra, Car*	525 (259–794) n=10	128 (44–269) n=10	24	Berge et al. 1990, Christensen & Hahn 2003, De Keersmaeker et al. 2002, Koop and Hilgen 1987, Odor et al. 2003, Wijdeven pers. comm.
G: Atlantic nemoral forest in England	*Fag, Que, Fra, Til*	–	256 (83–501) n=7	–	Christensen & Hahn 2003, Green & Peterken 1997, Mountford et al. 1999, Mountford in prep., Mountford & Peterken 2000
H: Central European mixed deciduous forest	*Fag, Fra, Que, Ace, Car, Pin, Ulm, Bet, Pic, Pin.*	561 (333–780) n=11	84 (10–177) n=12	15	Bantle 2002, Beneke 2002, Beneke & Manning 2003, Kegler 1999, Kölbel 1999, Labudda et al. 1999, Labudda & Staehle 1999, Nowack 2002, Nüss 1999, Seiler 2001,

Table 1. continued.

Forest type	Main tree species	Living volume (m^3 ha^{-1})	Dead wood volume (m^3 ha^{-1})	Dead wood ratio (%)	References
I: Submontane mixed forest in Bayischer Wald, Sumava Mnt. and Central Czech Rep.	*Fag, Pic, Abi, Ace, Ulm, Fra, Aln.*	682 (567–876) n=8	140 (73–234) n=9	21	Kölbel 1999, Nüss 1999, Odehnalova 2001, Prusa 1982, 1985, Vrska et al. 2000a, Vrska et al. 2001b, Vrska et al. 2001c
J: Submontane mixed conifer–beech forest in Carpatian Mnt. and adherent areas	*Fag, Abi, Pic, Ace.*	581 (473–741) n=10	216 (133–286) n=10	37	Jaworski et al. 1995, Jaworski et al. 1999, Jaworski & Paluch 2001, Korpel 1997, Saniga & Schütz 2001a, Saniga & Schütz 2001b, Saniga 1999, Vrska 1998, Vrska et al. 1999, Vrska et al. 2000b, Vrska et al. 2001a
K: Submontane beech forest in Carpatian Mnt. and adherent areas	*Fag, Ace, Que, Fra, Til, Pic, Abi*	631 (454–816) n=7	141 (99–183) n=7	22	Korpel 1995, Korpel 1997, Jaworski et al. 2002, Odor pers. comm., Saniga & Schütz 2001
L: Submontane–montane forest in the Alps, Swarzwald and S Bayern		498 (423–547) n=4	133 (60–296) n=4	27	Hanke 1998, Keller & Riedel 2000, Labudda 2000, Mayer & Neumann 1981 ao.
M: Submontane mixed beech-fir forest Slovenia	*Fag, Abi, Ace, Pic.*	664 (525–813) n=5	207 (69–568) n=5	31	Christensen & Hahn 2003, Debeljak 1999, Hartman 1987, Kovac & Joze 1999, Konecnik & Zaplotnik 2001, Papez 1997, Rozenbergar et al. 2003

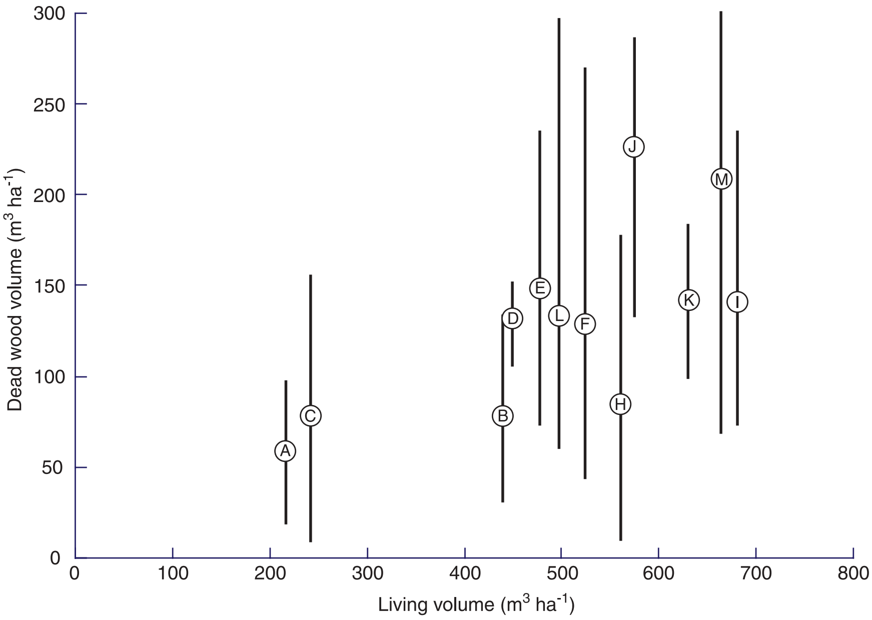

Figure 2. The relationship between dead wood level and living volume by forest type.

Decay rates

Decay rate is determined by a combination of tree species, climate, size and shape, so that some species decompose faster than others, warm and mild climates and nutrient rich soils increase the decomposition, small wood pieces decompose faster than large ones, and fallen logs decompose faster than standing dead trees. The tree species can generally be divided into three groups: Species with fast decomposition (*Betula*, *Populus*, *Tilia* etc.), species with intermediate speed of decomposition (*Fraxinus*, *Fagus*, *Picea* etc.) and species with slow decomposition (*Quercus*, *Pinus*, *Abies*, *Larix* etc.). The decay of a relative volume of dead wood can be described by a negative exponential function $V_t = V_0 * e^{-kt}$, where V_0 is relative start volume, *t* is time in years, V_t is volume after *t*, and *k* is a decay constant which is species specific (Figure 3A). The decay constant *k* is relatively high in areas with moist climates (>0.1) but very low in high mountain areas and in the northern boreal zone (<0.01) (Stokland 2001). Siitonen (2001) used decay rate together with information on productivity to predict a theoretically equilibrium volume of dead wood in Fennoscandian forest types (Figure 3B). The higher complexity of the central European forest and limited information on decay rates did not allow discussing the theoretical dead wood volume for all European forest types. But the general trend of higher productivity in mountain forests in Southern and Central Europe is without doubt the main explanation for the higher volume of dead wood, even where a generally higher decay rate will affect the volume negatively. High stand productivity in combination with a low decomposition rate therefore results in high dead wood volumes (Figure 3).

Disturbance

In contrast to the long-term average difference in dead wood volumes, which is primarily related to a combination of input rates and decay rates, the disturbance regime causes

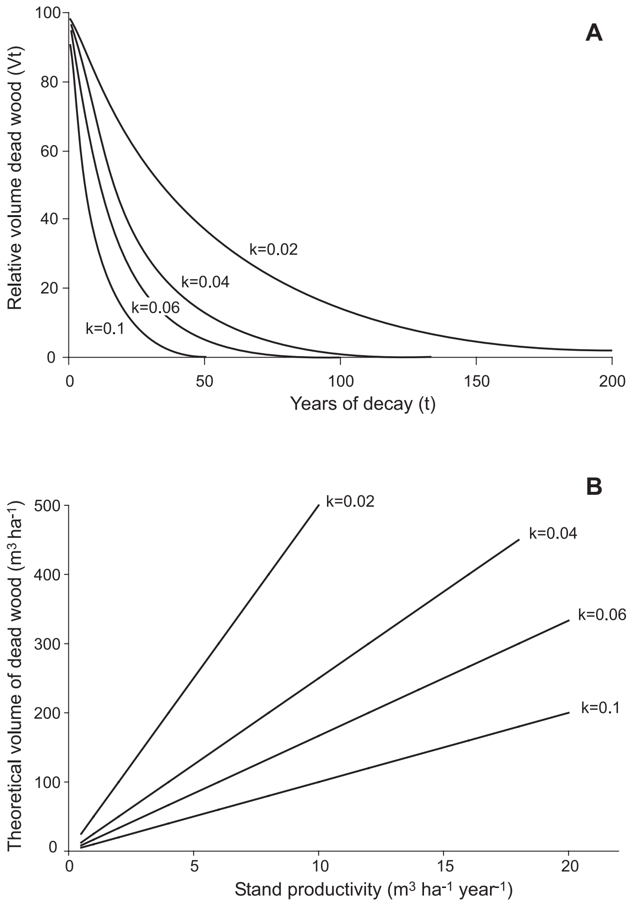

Figure 3. The relative dead wood volume (Vt) can be described as a function of decay rate (k) over time (t) so that a high decay rate causes a fast decomposition of the dead wood (A). When combining the decay rate (k) with the stand productivity, a theoretical dead wood volume can be obtained. Here, for example, a low decay rate in combination with high stand productivity causes an overall high theoretical dead wood volume (B). Based on Siitonen (2001) and Stokland (2001).

temporal pulses of high dead wood volumes. Typical disturbance-related pulses of dead wood are windthrow, icebreak, fire, insects and fungal attacks, which leave various amounts of standing and lying dead wood among the remaining living trees in the stand. In contrast to this, fire disturbances are characterised by an immediate consumption of a proportion of the dead (and living) wood.

How good is our reference?

The question of how close to a reliable reference for sustainable forest management our data from European forest reserves are is highly interesting. One important limitation to the data material is the lack of data from Southern Europe, which means that guidelines are only for north and central European forest types. Another constraint is the influence of former anthropogenic activities, which has altered forest structure and species composition. An example of this is infrequent high pulses of dead wood in English forests (Atlantic, nemoral forest type), where a skewed size class distribution in combination with strong storms have blown down large volumes.

Further, a harmonisation of sampling design, including minimum diameter sampled and sample size would increase the comparability of data, possibly in cooperation with coordination of sampling in national forest inventories.

Dead wood studies in other regions

One method of evaluating the reliability of the data is to compare it to similar forest types of other regions. Here we find that in general, the dead wood volumes as well as the variation between forest types in Europe are comparable with those of eastern North-America, where similar forest types, but with a higher degree of naturalness, are found. Here dead wood volume averages 121 m^3/ha (55–207 m^3/ha) in old-growth nemoral forest (Tyrrell and Crow 1994), and 73–129 m^3/ha in old-growth boreal forest (Stewart et al. 2003).

Dead wood in managed forests

In contrast, dead wood levels in managed forests of Europe are much lower than in unmanaged forests, as clearcutting, thinning operations and short rotations all lower the dead wood production. In e.g. Sweden, the average dead wood volume is 6.1 m^3/ha, (Fridman and Walheim 2000), and in England it is less than 20 m^3/ha (Kirby et al. 1997).

5. Conclusion

The relative dead wood ratio varies with forest type and the amount of dead wood and should therefore be seen in relation to forest type, standing volume and decay rates. Moreover, the local or regional disturbance regime is a key determinant of natural dead wood fluctuations, with respect to both frequency and intensity. The resulting natural pulses of dead wood indicate that an interval rather than an average value for dead wood should be applied in forest management.

Forest reserves are important as a reference for sustainable forest management, but we should take into consideration that the naturalness of forest reserves is higher in Eastern than Northwestern Europe. This should be considered when applying indicator values for dead wood levels or ratios

Finally, there is a need for further European-scale research on dead wood, including references from Southern Europe. The present low comparability of data should be improved, including a coordinated approach to sampling designs, sample sizes, and minimum diameter measured.

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Deadwood as an Indicator of Biodiversity in European Forests: From Theory to Operational Guidance

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Abstract

Decaying wood habitats associated with dead, dying and old trees are a key component of biodiversity in European forests and recognised as potential indicators for assessing and monitoring biodiversity. This paper reviews the link between different types of deadwood and the diversity of dependant flora and fauna in a range of European forest types. Coarse woody debris (CWD – logs and snags) is an important indicator of biodiversity in conifer-dominated forests in the Atlantic and Boreal biogeographical zones, but is less applicable to Mediterranean forests and wood pasture systems. Ancient and veteran trees are of key importance for rare and threatened saproxylic species in all forests, especially wood pasture, but to date have not been included in monitoring and assessment schemes at the European level. We propose a set of harmonised measures relating to CWD and ancient/veteran trees which could be adopted for use as biodiversity indicators in European forests, and outline an assessment framework for managers and policy-makers.

Keywords: saproxylic species; coarse woody debris; ancient trees; veteran trees.

1. Introduction

Dead and dying wood plays a key role in the functioning and productivity of forest ecosystems and provides important habitat for small vertebrates, cavity-nesting birds, and a

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host of lichens, bryophytes, polypores and other saproxylic fungi and invertebrates (Samuelsson et al. 1994; Esseen et al. 1997; Butler et al. 2001). Decaying wood habitats occur as logs, snags and stumps, and in the form of rot holes, dead limbs and roots, decay columns, heart rot and hollowing in living ancient or veteran trees (Read 1999). All these habitats have their own associated flora and fauna (Alexander 1999; Speight 1989; Samuelsson et al. 1994).

In natural forests, droughts, storms, fungal pathogens, insect disease, fire and mammals are the most important factors (including natural thinning) which result in decay, death and the creation of deadwood (Kirby et al. 1998). These disturbance factors vary in scale and intensity leading to a patchy distribution of deadwood at the stand and landscape scales with greater accumulations near canopy gaps and in old-growth stands (Humphrey et al. 2002; Sippola et al. 1998). The key old-growth forests in Europe for saproxylic diversity are found in boreal Fennoscandia and Russia (Esseen et al. 1997) and throughout Europe in mosaic woodland types such as wood pastures with a historic continuity of ancient trees (Butler et al. 2001). In managed forests deadwood has historically been removed either for wood fuel, or as a hygiene measure to protect the timber resource from what were perceived to be dangerous threats from insect and fungal attack (Butler et al. 2002; Bader et al. 1995) thus severely restricting the capacity of managed forest ecosystems to provide habitats for saproxylic species (Ranius and Jansson 2000; Speight 1989).

Recognition of the ecological importance of decaying wood has led to the incorporation of quantitative measures of deadwood in national forest inventories (e.g. Fridman and Walheim 2000) and as biodiversity indicators for use in monitoring programmes at the European level (MCPFE 2002; Kristensen 2003). The Ministerial Conference on the Protection of Forest in Europe (MCPFE) includes deadwood as one of 9 Pan-European sustainability indicators; the European Environment Agency (EEA) includes deadwood as one of its 15 core indicators of biodiversity (Kristensen 2003). Both these schemes tend to focus on deadwood as separate from living trees, for practical reasons, but the lack of recognition of the significance of dead wood in live veteran and ancient trees needs to be addressed.

The deadwood indicator is placed in the EEA category of “Long term-indicators”. Indicators in this category are judged to have high policy relevance (e.g. halting the decline of biodiversity) but assessment methodologies are little developed or unclear, data are either scarce, or difficult to aggregate, or yet to be defined. Given the ecological importance of decaying wood there is an urgent need therefore to develop an assessment framework and define data requirements as steps in the process of operationalising deadwood as a biodiversity indicator at the European level. This paper suggest ways in which this might be done with reference to different types of forest in Europe. We review the link between deadwood and dependant flora and fauna in a selection of forest types and identify measures which might have potential as biodiversity indicators. Where possible we include ranges of values for these indicators with respect to forest type, and propose an assessment framework for managers and policy-makers in Europe.

2. Selection of forest types and structure of the review

There are a number of different forest classification systems currently in use in Europe, ranging from potential vegetation type systems to the development of the UNIS/CORINE classification (for a review see Larsson 2001). Work is progressing to produce a forest type classification based on actual vegetation composition (Bradshaw et al. this volume). In this paper, we follow the BEAR project approach, where key factors of biodiversity are identified

Table 1. Forest Types for Biodiversity Assessment (FTBAs), occurring in European biogeographic regions. Reproduced with permission from Larsson (2001). A=Alps; P=Pyrenees C=Carpathian S=Scandinavian Alps; • = of major importance in the biogeographic region (•) = of minor importance in the biogeographic region.

Forest Types for Biodiversity Assessment FTBA	Occurs in the following Biogeographic Region (European Union 1992)					
	Boreal	Atlantic	Continental	Alpine	Mediterranean	Macaronesian
1. Subalpine conifer vegetation in nemoral zone				A, P, C		
2. North boreal spruce forest	•			S		
3. North boreal pine forest	•			S		
4. Middle boreal spruce forest	•			S		
5. Middle and south boreal and hemiboreal pine forest	•	•		S		
6. South boreal forest	•					
7. Hemiboreal spruce and fir-spruce forests	•		(•)			
8. Mixed spruce and fir forest			•	A, C		
9. Mixed oak forest	•	•	•			
10. Ashwood		•				
11. Mixed oak-hornbeam forest		•	•			
12. Lowland and submontane beech forest		•	•	A, P, C		
13. Montane beech and mixed beech-fir spruce forest			•	A, P, C	•	
14. Mediterranean and submediterranean mixed oak forest		•	(•)		•	
15. Mediterranean broad-leaved sclerophyllous forests and shrub					•	•
16. Mediterranean and Macaronesian coniferous forests, woodlands					•	•
17. Atlantic dune forest		•				
18. Ombrotrophic mires	•	•				
19. Arctic-subarctic mires				(S)		
20. Minerotrophic mires incl. swamp forest	•					
21. Swamp and fen forests, alder	•	•	•	(A)		
22. Swamp and fen forests, birch	(•)	•				
23. Flood plain (alluvial and riverine) forests		•	•			
24. Mediterranean and Macaronesian riverine woodlands and gallery forests					•	•
25. Laurel forest						•
26. Hedgerow		•	(•)		•	•
27. Chestnut coppice		(•)		(A)(P)(C)	(•)	
28. Pine plantation	•	•	•		•	•
29. Spruce plantation	•	•	•			
30. Poplar plantation		•	•			
31. *Robinia* plantation			•			
32. *Eucalyptus* plantation					•	•
33. Other plantation		•			•	

within different Forest Types for Biodiversity Assessment – FTBAs – (Larsson 2001). The FTBAs are stratified by biogeographic regions (Table 1), and deadwood is considered to be of major importance as a stand-level indicator in the majority of these. Here we concentrate on a selection of FTBAs found in the Boreal (FTBAs 2,3,4), Atlantic (FTBAs

5,9,10,11,12,28,29), Alpine (FTBAs 1, 6, 8,12, 13) and Mediterranean (FTBA 14) zones. Although not strictly separate FTBAs, wood pastures and parkland are also included as they are of such high importance for saproxlylic diversity. For the purposes of this review, the common term coarse woody debris (CWD) is defined as all fallen deadwood (logs), standing dead trees (snags) and stumps, but not including ancient and veteran trees. Ancient trees are those which have passed biological maturity, have entered a period of retrenchment and contain a mix of living, dying and deadwood (Read 1999). Veteran trees exhibit some or all of the habitat features of ancient trees, without necessarily being of great antiquity.

3. Deadwood in Atlantic forest types

3.1 Hemiboreal pine forest in the Scottish Highlands (FTBAs 5, 28)

Historically, management has reduced the amount of deadwood in hemi-boreal native pine *Pinus sylvestris* woods and old pine plantations in the Scottish Highlands; typical volumes range from 30–50m^3 ha^{-1} in total for snags and logs. Records from the least disturbed remnants suggest that volumes in excess of 100m^3 ha^{-1} might be a more natural figure. Typically, mature or ‘granny’ trees die *in situ*, creating the characteristic stag-headed trees which provide stable long-term habitat. More rarely, whole trees blow over creating larger logs which tend to rot very slowly. Snags, logs, stumps and crevices in veteran pines (and occasionally alder *Alnus glutinosa*, aspen *Populus tremula* and birch *Betula* spp.) provide niches for epixylic lichens (e.g. *Chaenotheca* spp.) including many rare and threatened species.

Saproxylic beetles depend on a continuity of supply of large diameter snags and logs, for example the timber man beetle *Aconthocinus aedilis* is an early colonist of new deadwood. Rot holes in stumps and snags are important for the larvae of rare hoverflies (e.g. *Callicera rufa*). Birds, such as the crested tit *Parus cristatus* nest in well-decayed snags and stumps > 30 cm in diameter; quantities of about 10 snags per ha are recommended (Humphrey et al. 2002). Other species such as wryneck *Jynx torquilla* nest in old woodpecker holes in snags, and capercaillie *Tetrao urogallus* roost in ancient/veteran pines. Birch, willow, aspen and juniper *Juniperus communis* also provide key habitats. Veteran aspen, for example supports a distinctive saproxylic ecosystem in its own right. Other habitats include bog woodlands where stunted pine, birch and juniper form a distinctive habitat. Fallen trees are also important in streams, ponds and other water features.

3.2 Spruce plantations (FTBA 29)

Deadwood in (often non site-native) spruce *Picea* spp. plantations can have value for saproxylic species, especially in well wooded landscapes where there are sources of colonising species (Humphrey et al. 2004). Important habitats for hoverflies, other saproxylic diptera and bryophytes include large diameter (= 20 cm) moderately decayed fallen deadwood and crevices/rot holes in large trees (Humphrey et al. 2002). As in pinewoods, stumps provide habitat for epixylic lichens, and snags are used by hole-nesters such as the greater spotted woodpecker *Dendrocopos major* and the willow tit *Parus montanus*. The retention of stands (over at least 1% of the forest area) with 30–40 large (80–100 cm diameter), old (100 years +) trees per ha is recommended to enhance deadwood habitat in spruce plantations (Humphrey 2004).

3.3 Broadleaved high forest woodland types (FTBAs 9,10,11,12)

In the oceanic north and west of the Atlantic zone the most common broadleaved types are oak *Quercus* spp.-birchwoods and mixed ash *Fraxinus excelsior* woods. In the south and east, oak, beech *Fagus sylvatica* and oak-hornbeam *Carpinus Betulus* types predominate. Native broadleaved woodland which has been managed historically as high forest or coppice is generally lacking in deadwood, although the absence of management over the past 80–100 years in some woods has allowed the build up of dead and decaying limbs, logs and snags through competition and windthrow. In western oceanic climates, logs are important as a substrate for bryophytes. Many ancient woodland bryophytes need a constant regime of humidity and shade, e.g. liverworts (*Lejeunea* spp). Well-decayed (i.e. where the bark has been lost, and the wood is beginning to break down) large diameter (= 20 cm) fallen deadwood appears to be the most valuable substrate (Humphrey et al. 2002). On birch snags, bracket fungi (e.g. *Fomes fomentarius*) support their own specialised saproxylic ecosystem. Several moth species need veteran birch trees, for example the Rannoch sprawler *Brachionycha nubeculosa*. In lowland high forests, fallen and standing deadwood and hollowing trees and limbs are very important for wood-decaying fungi, which in turn support a high diversity of invertebrates such as fungus gnats, mycophagous and predatory beetles. Some wood decaying fungi are specific to particular tree species creating their own characteristic types of decaying wood, but many are associated with a variety of tree species.

3.4 Broadleaved wood pasture/parkland

Wood pasture and parkland within the Atlantic zone is characterised by large, ancient open-grown (often pollarded) trees at various densities, in a matrix of grazed flower-rich grassland or heathland (Alexander 1999). There have been repeated calls to expand the Natura 2000 series of priority habitat types (European Union 1992) to include wood pasture (Andersson 2001). Wood pasture should also be included as an FTBA within the BEAR system and any forthcoming habitat classification.

Some of the rarest and most threatened species and communities remaining in Europe today are associated with ancient trees of wood pasture; the larger, older and more numerous the ancient/veteran trees the richer the biodiversity of the habitat (Butler et al. 2001). The greatest diversity appears to be associated with trees in open structured stands. Disturbance and grazing by large herbivores helps maintain populations of saproxylic species by preventing the regeneration of trees and shrubs which otherwise would shade out important habitats on veteran and ancient trees (Alexander 1999; Applequist et al. 2001). Hedgerows (FTBA 26) can also be of key importance in wooded landscapes as they can support relict populations of ancient trees.

The key part of the wood decay process in living trees is heartwood fungal decay (Alexander 1999) which creates habitat for many of Europe's rarest saproxylic invertebrates such as *Limoniscus violaceus*, and *Osmoderma eremita*. As nectar sources are important for invertebrates, it is essential that they can find flowering shrubs close by. One of the key structural indicators is continuity of light demanding tree and shrub species and the full range of tree species appropriate for that habitat as some fungi and invertebrates are tree species specific (Alexander 1999). Trees such as oak support a huge range of saproxylic coleoptera, and epixylic lichens, whilst beech is more important as habitat for diptera (Rotheray et. 2001). Other important decaying wood habitats are found in snags, 'phoenix' trees, layering trees, ancient pollards and coppice stools. CWD is not generally considered important except as very large diameter logs.

Spatial and temporal continuity of ancient trees is a key requirement. Some species of lichens have been shown only to move across to trees of more than 250 years old, therefore if old trees die before the next suitable cohort is available it is possible that species will die out. Another key aspect is the density of live trees, as distance of flight in key saproxylic beetles appears to be very limited. Species such as the hermit beetle *Osmoderma eremita* and male stag beetles *Lucanus cervus* have home ranges of 1 ha or less, and fly only up to 200 m (Ranius 2003; Sprechter 2003). This implies that the density of protected trees should be of the order of 1 tree per hectare minimum to conserve these and similar species. There is also evidence of a link between tree diameter and saproxylic species diversity.

4. Deadwood in boreal forest types

4.1 North boreal pine forest in Scandinavia (FTBA3)

Pine *P. sylvestris* forests in the northern boreal zone are characterized by slow tree growth and slow decomposition of CWD. The mean diameter of 300 year old pine is in the range of 21–40 cm (Ilvessalo 1970). CWD volumes in natural old-growth stands range from 19 $m^3 ha^{-1}$ at the timberline to 67–111 $m^3 ha^{-1}$ in more southern parts of the zone (Linder et al.1997; Sippola et al. 1998) and from 1.7 to 9.7 $m^3 ha^{-1}$ in managed forests (Kruys et al. 1999; Fridman and Waldheim 2000). In old-growth stands, 60–80% of total CWD volume is logs and 20–35% is made up of snags (Sippola et al. 1998).

Standing dead pines and snags provide nesting places for hole-nesting birds (about 20–34% of the total bird fauna). Black woodpecker *Dryocopus martius* requires relatively large-diameter trees for its nest; the holes are later used by other species such as hawk owl *Surnia ulula* and, near wateredges, goldeneye *Bucephala clangula* and smew *Mergus albellus*. Large-diameter veteran pines offer bases for the nests of golden eagles *Aquila chrysaetos* and ospreys *Pandion haliaetus*, and broken-top snags are used for nesting by great grey owl *Strix nebulosa* (Hautala et al. 1978).

Large-diameter (> 25 cm) pine logs, especially those in mid and late decay stages, are important to the diversity of wood-decomposing fungi and saproxylic invertebrates (Renvall 1995; Esseen et al. 1997; Sippola and Renvall 1999). Dry, hard, barkless standing dead pines are characteristic of north boreal old-growth pine forests and host a specific species composition of wood-decomposing fungi, including many threatened species such as *Antrodia crassa* (Renvall and Niemelä 1994). Fire has been the most common natural disturbance in boreal pine-dominated stands, with a return interval of 50–120 years on average in pristine forests (Zackrisson 1977). Charred CWD hosts some characteristic invertebrates, e.g., saproxylic beetles, and many heteroptera and diptera species which mainly inhabit microfungi growing on charred wood (Wikars 1992). Many of these species have declined strongly in boreal forests due to effective fire suppression (Esseen et al. 1997).

Birch is usually found as admixture in boreal pine forests, and standing veteran birch trees, snags and logs support a large number of saproxylic invertebrates, wood-decomposing fungi and epiphytic and epixylic lichens (e.g. Kaila et al. 1994; Sippola and Renvall 1999; Halonen and Jääskeläinen 2003). Well-decayed trunks offer nesting places for small hole-nesting birds such as Siberian tit *Parus cinctus*.

Of the deadwood dependant species groups, many polypore and beetle species seem to be relatively well adapted to the open forest structure and frequent disturbances typical for northern pine forests (Sippola and Renvall 1999, Sippola et al. 2002) and benefit from the retention of trees in managed stands. The ability of different species to persist in forest stands containing various amounts of decaying wood probably varies by a few orders of magnitude, according to, for example, the substrate requirements, life history and dispersal ability of species. However, volumes of approximately 20 m^3 ha^{-1}, which is the average amount of CWD found in the northernmost natural pine and spruce-dominated stands (Sippola et al. 1998), could be set as a rough minimum goal in boreal pinewoods (see also section 4.2).

4.2 North and middle boreal spruce forests (FTBAs 2, 4)

The mean volumes of CWD in old-growth Norway spruce *Picea abies* forests varies from 19 m^3 ha^{-1} in the north boreal zone near the timberline, to 201 m^3 ha^{-1} in the southern part of the middle boreal zone (Linder et al. 1997; Sippola et al. 1998). In managed forests, the volumes are on average only about 10% of the volumes found in natural stands, the greatest reduction being in the amount of large-diameter CWD (Siitonen 2001). Birch, aspen, rowan *Sorbus aucuparia* and bird cherry *Prunus padus* occur as admixture trees. Aspen has special importance for biodiversity in the boreal zone, supporting a large variety of host-specific cryptogams, beetles and lepidoptera (e.g. Bernes 1994) as well as many rare epiphytic lichens such as *Lobaria pulmonaria.* Well-decayed snags and basal trunks of old birch trees are an important substrate for several rare epiphytic species, e.g. *Chaenotheca gracillima* (Halonen and Jääskeläinen 2003).

In natural conditions, spruce forests mostly regenerate by windthrows and small-scale gap dynamics, and many saproxylic beetles, fungi and lichens are adapted to long forest continuity with a steady, humid microclimate and closed forest canopy (Esseen et al. 1997). Holes and cavities in standing dead spruce trunks offer nesting places for hole-nesters such as the three-toed woodpecker *Picoides tridactylus*; pygmy owl *Glaucidium passerinum* and Tengmalm's owl *Aegolius funereus*; both owls use woodpecker holes for nesting (Hautala et al. 1978). Several tit species inhabit spruce-dominated stands, and nest in soft, well-decayed birch snags. Flying squirrel *Pteromys volans*, which occurs at its western distribution margin in Finland uses large-diameter dead aspens for nesting and shelter (Reunanen 2001).

Spruce logs host a large variety of cryptogams and invertebrates. Many characteristic wood-decomposing fungi inhabit spruce; some of them are restricted to, or are more abundant in northern and middle boreal zones than in more southern latitudes. Large-diameter spruce logs in early decay stages host many rare corticioid fungi (e.g. *Laurilia sulcata)* whereas mid and late decay stages are important for many rare and threatened polypores such as *Amylocystis lapponica* (Niemelä 2003); red listed species show a strong preference for mid/ late decayed spruce logs of over 25 cm diameter (Renvall 1995; Kruys et al. 1999). Red-listed bryophyte species, such as *Lophozia ascendens* (Frisvoll and Prestø 1997), and a high number of rare and threatened beetle species, many of which require a steady moist microclimate and/or long forest continuity (e.g. *Pytho kolwensis*) also prefer mid/late decay stages (Siitonen and Saaristo 2000). These species benefit from patches which are left uncut in managed forests as woodland key habitats. In contrast, leaving deadwood on clear-cuts benefits species which prefer or tolerate sun-exposed habitats (e.g. Kaila et al. 1997). About 20 m^3 of CWD per hectare appears to be a stand-level threshold value at least for the occurrence of threatened polypore species in boreal spruce forests (Penttilä et al. 2004).

5. Dead wood material and associated organisms in Alpine (FTBA 1, 6, 8, 12, 13) and Mediterranean forest ecosystems (FTBA 14).

Alpine landscapes are a dense mosaic of very small ecosystems. Recent alternations of ice ages and thaws have left many "relict" boreo-alpine species. Deadwood plays a strong role in the conservation of habitat for more than 25% of the mountain forest fauna and flora (Dajoz

Table 2. Ecological characteristics of different saproxylic insects.

Micro-habitat	Duration	Functional group	Population	Dispersal range	Taxon	Author(s)
Recently dead wood	Very short (<1 year)	Xylophagous	Very dense	High (10–171 km)	Scolytidae Curculionidae	(Nilssen 1984)
Lignicolous fungi	Short (<10 years)	Mycetophilous		< 1 km	Beetles and flies	(Jonsell et al. 1999)
Logs	Short to long	Saproxylic	More diverse at 150 m scale		Beetles and flies	(Schiegg 2000)
Hollow tree	Very long (several years)	Saproxylophagous	16–18 adults in each occupied hollow	Short (30–190 m) for 6–28% of the population	*Osmoderma eremita* (Scarabaeidae)	Population size in Ranius (2000) Dispersal range in Ranius and Hedin (2001)

1965; Dodelin et al. 2003; Marage and Lempérière 2004). For the flora, one of the most important roles of deadwood is to provide sites for the germination and establishment of trees, especially of coniferous species. Logs also provide a substratum for most species of lichens and mosses. For the fauna, deadwood is a temporary habitat for hole-nesting vertebrates. An example is the strong association between Black Woodpecker *Dryocopus martius* and Tengmalm's Owl *Aegolius funereus*. In winter, deadwood provides a large web of paths under the snow for small mammals.

Saproxylic species use deadwood as a permanent habitat. They feed on sound wood (xylophagous, lignicolous fungi), decaying wood (saproxylophagous) or other saproxylic organisms (e.g. zoophagous parasites, mycetophagous species). For instance, European silver fir *Abies alba* hosts the rare fungus *Ischnoderma benzoinum* which in turn hosts the strictly associated beetles *Mycetoma suturale* and *Bolitophagus reticulatus* themselves eaten up by small Staphylinids. An ecological specificity of the saproxylic fauna is their variable dispersal ability which decreases from species breeding on recent deadwood to those living in strongly decayed wood (Table 2). The ability to colonise new substrates is less important for saproxylophagous species than for xylophagous ones. This can be explained by the persistence of deadwood habitats, which is short when the deadwood is fresh and longer when rot is progressing. In the French mountain forests, recent deadwood is a rare and scattered habitat at the stand level while well-decayed wood is more frequent.

Of the rare forest saproxylic beetles of the French Alps 40% are linked to conifer deadwood; 34% to deciduous deadwood and 25%% are common to both. In 11 studies in mountain forests (FTBA 1, 6, 8, 12 & 13) 166 types of deadwood were recorded in total, but with less than 57 different types in individual forests (depending on tree species, management history and site productivity). On average, 20% of the total number of pieces were large logs with diameter > 40

Table 3. Micro-habitats used by rare and threatened beetles in the French Alps (data in Brustel 2001)

Specific micro-habitats	Number of rare species (beetles) hosted (incl. strictly associated spp. in parenthesis)
Large fallen deadwood	36 (17)
Standing deadwood	2 (2)
Stumps	3 (0)
Bark	16 (6)
Lignicolous fungi	10 (8)
Other or unknown	51 (10)
Total	118 (43)

cm hosting rare species of beetles (Table 3) or fungi (Froidevaux 1975) threatened by forestry practices. However, in general, the more diverse the deadwood is, the more diverse and interesting the related biodiversity is. In the North Alpine area at least 20 m^3ha^{-1} of dead material, should be left to provide a suitable habitat for species living on deadwood in mountain forests. This may include 40 to 60 types of dead wood in mixed forests and a total of 40 pieces of deadwood per ha of which 20% should be large diameter logs.

In the Mediterranean region, the risk of fire is a limiting factor for the conservation of deadwood in conifer forests and consequently CWD is removed from the forest when possible. However, Mediterranean forest ecosystems such as oak forests with veteran trees provide habitats for large numbers of saproxylic invertebrates (Lempérière et al.1999).

The French Forest Office (ONF 1998) recommend a minimum of one cavity/ancient tree ha^{-1} in managed forests, although 5 to 10 ancient/cavity trees ha^{-1} would be a more applicable threshold for sustaining saproxlic diversity. Different scales must be considered for dead wood management. Regarding the dynamics of dead wood, a recruitment rate of 1 $m^3ha^{-1}year^{-1}$ over a 10 km^2 scale should be suffient to ensure temporal continuity of habitat.

6. Conclusions

6.1 potential deadwood indicators for use in European forests

Measures of deadwood clearly have the potential as indicators of saproxylic and epixylic diversity in both managed and natural forests in Europe. Table 4 provides a summary of the key measures relating to the 16 FTBAs reviewed in this paper. These measures include volumes and frequency of logs and snags, together with frequency and age ranges of ancient trees. However, there is inevitably some over-simplification of these to allow comparability between broad woodland types. It is also important to note that Continental FTBAs are not covered in this paper.

One of the main principles underlying the EEA Core Biodiversity Indicator concept is that core indicators should provide clear priorities for environmental data collection initiatives as these are expensive and involve long-lead times between conception and delivery. Ancient trees appear to be crucial for saproxylic/epixylic diversity in all the studied FTBAs (Table 4) yet have not been included in indicator schemes in the past (MCPFE 2002). Enumeration of ancient trees should be a priority for data collection in any future schemes for monitoring biodiversity in the EU.

Of the remaining measures of deadwood, not all are relevant to all FTBAs, for example CWD appears not a key measure in wood pasture nor in Mediterranean systems (although more research is needed to substantiate this in the latter case). The option is available therefore to exclude CWD from surveys of these woodland types, on the basis that no extra information would be gained from their inclusion over and above that obtained from a count of ancient trees. Similarly, stumps are locally important in some woodland types, but not generally across all type (based on current knowledge). The other important point relating to CWD is that larger-diameter material (in a range of decay states) appears to be most valuable for saproxyls and epixyls (although small dimension material is also important for some species groups – Kruys and Jonsson 1999). There are subtle differences in the lower cut off size, but harmonization across Europe is possible if the lowest common measures were selected (e.g. 20 cm diameter for both logs and snags). Variants such as volume of charred CWD or decorticated dry, hard snags and logs could be added for boreal forests, but would not be necessary in other forest types.

To be useful for monitoring trends in biodiversity it is necessary to ascribe quantitative values to the indicators and to have some idea what the consequences in changes in those values might be for dependant flora and fauna. Thus a range of possible values are included for each measure in Table 4. The upper limit of the range is defined by reference to information from natural/virgin stands where these exist, and from “best” examples. It is important to note that valuable areas for biodiversity can be found in places that have had a strong human impact for hundreds or thousands of years such as wood pasture.

The lower range limit for the different measures is more difficult to define, since there is very little information available on threshold volumes and frequencies of deadwood for sustaining key populations of species. The precautionary approach suggests that less than 1 ancient tree or cavity/veteran tree per ha, and less than $20 m^3 ha^{-1}$ of larger diameter CWD (all sizes of CWD in the boreal and hemi-boreal zone) would seriously compromise the potential saproxylic interest of most mature stands (Table 4). Even on clear-fell sites, there is a good case for retaining deadwood (either snags or logs) and other structural legacies after felling (Kaila et al. 1997). The retention of key tree species, like oak, beech, sweet chestnut birch and aspen will also provide important and distinctive deadwood habitats (Table 4).

6.2 Implementation

It is beyond the scope of this paper to review the ways in which deadwood is currently assessed in different European countries (see Schuck et al. in this volume for more information), but it is clear that national forest inventories provide the easiest instrument for undertaking large-scale assessments of deadwood. We suggest the following measures could be adopted in forest inventories and also used by managers at the local level to give a broad snapshot of the value of particular stands.

1. Assessment of volume, by decay class and species, of large (=20cm diameter) CWD (snags/logs). Standard transect and plot methods are available for the assessment of logs and snags (Kirby et al. 1998). It is suggested that a simplified decay scale is used: e.g. 1=fresh; 2=intermediate-loss of bark; 3=well decayed – loss of heartwood; more detailed scales would seem to be unjustifiable in terms of the specificity of information available.
2. Assessment of ancient and veteran trees by counts per ha by tree species. Ancient trees are an excellent surrogate indicator, so there should not be a requirement to inventory decaying wood habitats on individual trees. However, there would be some merit in measuring diameter or girth of ancient trees as this factor appears to be correlated with saproxylic value.

Table 4. Measures of deadwood for consideration as potential indicators of saproxylic and epixylic diversity in a selection of different European forest types in different biogeoclimatic zones. Information is summarised from preceding sections and from Humphrey et al. (2002) and Butler et al. (2002). Broad ranges in values for the indicator are given in parenthesis. Volumes are in $m^3 ha^{-1}$. LI = deadwood component is of lower importance. Ancient trees are defined in the broadest sense (Read 1999) to include: veteran trees, phoenix trees, ancient coppice stools with regrowth etc.

	Atlantic			Boreal		Alpine	Mediterranean
Deadwood component	Broadleaved wood pasture	Broadleaved high forest (FTBAs)	Hemi-boreal pine (+ spruce plantations)	Boreal spruce	Boreal pine	Beech and mixed beech-fir forest	Mixed oak forest
Stumps	LI	LI	Volume ≥ 20 cm diameter (20–40)	LI	LI	LI	LI
Snags	Volume: LI	Volume ≥ 20 cm diameter (20–40)	Volume ≥ 30 cm diameter (20–100)	Volume all diameters (4–35)	Volume all diameters (4–23)	Volume: LI	LI
	No ha ≥ 20 cm diameter (1–5)	No ha^{-1} (3–10)	No ha^{-1} (3–10)			No. ha^{-1} >20 cm diameter (1–10)	
Logs	Volume: LI	Volume ≥ 20cm diameter (20–40)	Volume ≥ 20 cm diameter (20–100)	Volume all diameters (16–140)	Volume all diameters (16–90)	Volume ≥ 40cm diameter (20–110)	LI
	No. ha ≥ 20 cm diameter (1–5)	No.ha^{-1} (3–10)	No.ha^{-1} (3–10)			No.ha^{-1} all diameters (>40)	
Ancient trees	No of trees ha^{-1} ≥ 100 years old by species (1–100) ≥ 250 years old (1–100)	No of trees ha^{-1} ≥ 100 yrs old by species (1–10)	No of trees ha^{-1} ≥150 yrs old by species (1–10)	No of trees ha^{-1} ≥150 yrs old by species (1–5)	No of trees ha^{-1} ≥150 yrs old by species (1–5)	No of trees ha^{-1} ≥ 150 yrs old by species (1–10)	No of trees ha^{-1} ≥150 yrs old by species (1–10)
Key tree species	Oak, beech, ash, field maple, hornbeam, sweet chestnut	Birch, ash, beech, oak	Pine, birch , aspen	Spruce, aspen, birch, goat willow	Pine, birch	European silver fir, *Pinus* spp.	Oak

While it is relatively easy to envisage a range of harmonised measures of deadwood for use by managers and policy-makers across Europe, the way in which these measures are interpreted is not so straightforward. It is not necessarily the case that saproxylic diversity will increase with increasing amount of deadwood as there are a range of other factors which potentially affect deadwood quality, such as composition of the surrounding landscape (e.g. Humphrey et al. 2004) and long-term habitat continuity (Verheyen et al. 2003). For example, in wood pastures, the maintenance of the pasture element is crucial to sustaining saproxylic diversity through provision of nectar sources and allowing sunny conditions around ancient trees (Alexander, 1999).

To be meaningful, any analysis of trends in deadwood indicator values needs to take account of context, and the influence of this and other ecological factors. Finally, it is recommended that once a minimum set of common protocols for the assessment of deadwood in European forests are agreed, guidance material should be produced describing the protocols and assessment methodologies in a clear and accessible format, backed up by training and advisory workshops for end-users.

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Development of Dead Wood Indicators for Biodiversity Monitoring: Experiences from Scandinavia

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Abstract

Dead wood is a key ecological factor in forest ecosystems. In Scandinavia, it has been estimated that 6000–7000 species depend upon dead wood. The paper presents a summary of different dead wood qualities that are important to wood-inhabiting species.

Complete dead wood inventories of dead wood were initiated in the National Forest Inventories (NFI) in Finland, Norway and Sweden during the 1990s. The volume of all dead wood is measured and classified according to qualitative parameters: type of dead wood (standing/lying), tree species and degree of decomposition. In this paper we also suggest that mortality cause and maximum diameter should be sampled in a standardised manner. The plot size in NFI systems is sufficient for statistics on average volumes at national and regional scales. In order to establish statistics on local abundance of dead wood, a stratified subset of the inventory plots should be enlarged to minimum 0.2 ha or supplemented by a line transect of minimum 200 m.

Biological validation analyses indicate that different dead wood parameters predict species richness and presence of threatened species with a high degree of confidence. In addition to national and regional indicators, there is a need to develop an indicator for the local abundance and qualitative composition of dead wood.

Keywords: dead wood; monitoring; forest inventory; saproxylic species; biodiversity indicator.

1. Introduction

Dead wood is a key ecological factor of the forest ecosystem. Functionally, it represents an important component of the forest carbon pool, and studies from unmanaged boreal forests

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indicate that dead wood makes up 20–30% of the total timber biomass (Linder 1986; Syrjanen et al. 1994; Krankina and Harmond 1995; Sippola et al. 1998), and up to 70% in recently disturbed areas (Krankina and Harmond 1995). Nearly all forest organism groups have species adapted to utilise this historically abundant resource base. In Finland, it has been estimated that nearly 5000 species live in dead wood during parts of, or their whole life cycle (Siitonen 2001).

Dead wood has been identified as an important component in several forest monitoring schemes in Europe. The Ministerial Conference on the Protection of Forests in Europe (MCPFE) has recently formulated a set of improved Pan-European indicators for sustainable forest management (MCPFE 2003). Under the section of biological diversity, dead wood is defined as one out of 9 indicators. In parallel, and in cooperation with the MCPFE process, the Ministerial process "Environment for Europe" has through the work programme WP-CEBLDF and the BEAR project made an effort to identify indicators for assessing forest biodiversity. The BEAR project has proposed a set of forest biodiversity indicators that includes various dead wood qualities (Larsson 2001). Also the upcoming EU-programme "Forest Focus" identifies dead wood as one of several factors to be monitored (European Commission 2002).

The high number of wood-associated species is one reason to establish dead wood as a biodiversity indicator. Another reason, related to sustainable forest management, is the link between timber extraction and the reduced amount of dead wood in managed compared to unmanaged forests. In Scandinavia, several studies have established that the amount of dead wood in unmanaged boreal forests generally is 60–90 m^3/ha (Siitonen 2001), whereas the average amount of dead wood in landscapes dominated by forest management is 3–10 m^3/ha (Stokland et al. 2003), i.e. a reduction below 10% of the natural levels.

The purpose of this paper is to focus on dead wood properties, indicators and monitoring in a biodiversity context. The first part of the paper outlines some key forest statistics from Scandinavia and presents a summary of the knowledge on species diversity related to dead wood in this region. In the main part of the paper, we present Scandinavian experiences on dead wood sampling and approaches to establish quantitative indicators. In particular, the paper highlights experiences and results from National Forest Inventories (NFI) in Finland, Norway and Sweden, as well as a Norwegian project on dead wood and biodiversity linked to the Norwegian NFI.

2. Scandinavian forests and species diversity in dead wood

2.1 Scandinavian forests

Finland, Norway and Sweden are the main forest countries in Scandinavia, and the forest covers about 65 mill. ha in these countries (Stokland et al. 2003). About 85% of the forest area is in the Boreal region, 14% is in the hemi-boreal region with a mixture of boreal and temperate forest, and 1% is in the Nemoral region that extends south to the Alps.

The far most common tree species are *Pinus sylvestris*, *Picea abies*, *Betula pubescens* and *Betula pendula* (syn. *B. verruculosa*) that together account for 95% of the standing volume in the region (Stokland et al. 2003). Other common tree species are *Populus tremula, Alnus incana, A. glutinosa* and *Salix caprea* in the whole region, and *Quercus robur*, *Q. petraea*, *Fraxinus excelsior*, *Tilia cordata* and *Fagus sylvatica* in the Hemi-boreal and Nemoral regions. Altogether there are 43 indigenous tree species in Scandinavia (Stokland et al. 2003).

Economic use of the Scandinavian forests has developed from simple timber extraction of selected tree species and dimensions since some 500 years ago. The timber extraction expanded during the 1600s and 1700s along the coastline and major river systems. Angelstam and Majewski (1995) have indicated that a timber frontier moved through Finland, Norway, and Sweden during the 1800s leaving only a few areas untouched in Northern Sweden and

Table 1. Number of wood-associated species in different organism groups in Scandinavia[1]. The numbers are based on counting individual species or expert assessments.

Fungi	
Ascomycetes	751
Basidiomycetes	1270
Lichens	220
Insects	
Coleoptera (beetles)	1257
Diptera[2]	500–1200
Hymenoptera[2]	900–1400
Other insects[2]	> 160
Other species groups	
Acarina (Ticks)[2]	300
Nematodes[2]	> 100
Myxomycetes[2]	150
Mosses	97
Vertebrates	54
Total	5800–7000

1) The numbers are based on data from Finland, Norway and Sweden
2) The species numbers in these groups are expert assessments.

Finland. In the beginning of the 1900s there was a serious concern for the timber resources across the three countries. This concern led to the start of National Forest Inventories as well as nationally organised forestry based on modern silvicultural principles. Today, nearly all forest is managed and typically 6–8% of the economically profitable area is over-mature from an optimal stand rotation point of view (Stokland et al. 2003). An exception is Northern Sweden and Finland where there is about 20% over-mature forest.

2.2 Species diversity in dead wood

Scandinavia has a strong tradition of dead wood biodiversity studies rooted in classic works on the ecology of wood-inhabiting beetles (Saalas 1917; Palm 1951; Palm 1959), and fungi (Eriksson 1958; Eriksson and Strid 1969). Recently, these data sources, subsequent publications and unpublished data have been structured in a database on the ecology of more than 3600 species living in dead wood (Dahlberg and Stokland 2004). From this database and expert judgements, we have good data on the total species richness of wood-inhabiting species in Scandinavia that is 6000–7000 species (Table 1). This corresponds to about 25% of all forest species in the region.

In order to develop proper dead wood biodiversity indicators it is crucial to understand which qualities of dead wood that are important to the wood-inhabiting species. We will therefore highlight some dead wood properties for which the database gives reliable data: type of dead wood, tree species, decay class and dimension.

Type of dead wood

Standing versus lying dead trees represent quite different habitats for many species. Some organism groups, like birds and lichens are almost exclusively associated to standing dead

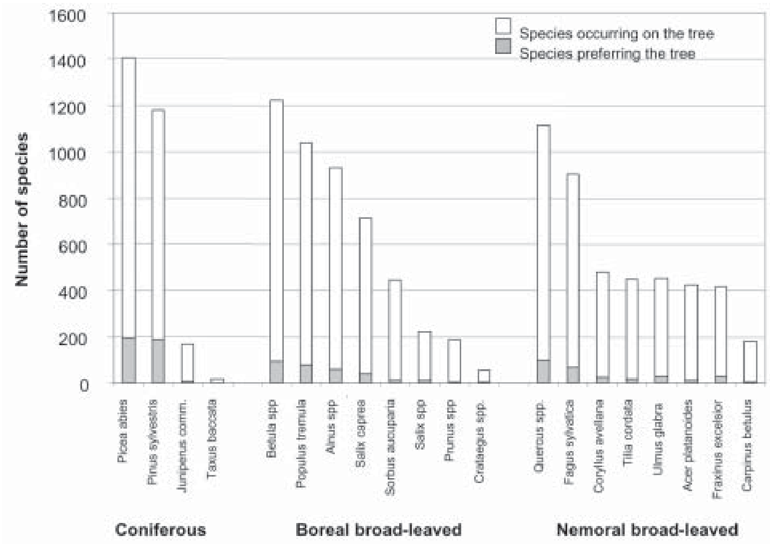

Figure 1. Number of wood-inhabiting species occurring on different host tree species in Scandinavia. The grey part of the columns represents species with clear preference for the host tree, and the white part represents species that also occur on other trees.

trees, whereas others, like fungi and mosses primarily utilise lying dead wood. In the database of 3600 species, about 40% of the species are found on downed logs and 15% on standing dead trees, whereas the preferences is unknown for a large proportion of the species.

Tree species associations

There are quite different species assemblages in dead wood of coniferous and broad-leaved trees. Of the 3600 species, about 50% are associated to broad-leaved trees, about 25% are associated to coniferous trees, 10% are generalists that utilize both broad-leaved and coniferous trees, and the remaining 15% has unknown host tree association. About 20% of all species shows clear preference for a single host tree (Figure 1).

Decay preferences

There is a distinct species diversity pattern that occurs during the process of wood decomposition. Several species are able to utilize dead wood in still living trees (about 10% of the 3600 species). The species number increases rapidly after the tree dies (Figure 2). In the middle of the process the species richness peaks, and thereafter it decreases gradually as the log becomes completely decomposed (Figure 2). There is also a distinct species turnover during this process. About 10% of the species can utilise wood of still living trees, 25% are early decomposers, 30% are active in middle of the decomposition process, 15% are late decomposers, and hardly any are generalists utilizing wood over a broader spectrum of the decomposition process. About 20% of the species have unknown decay preferences.

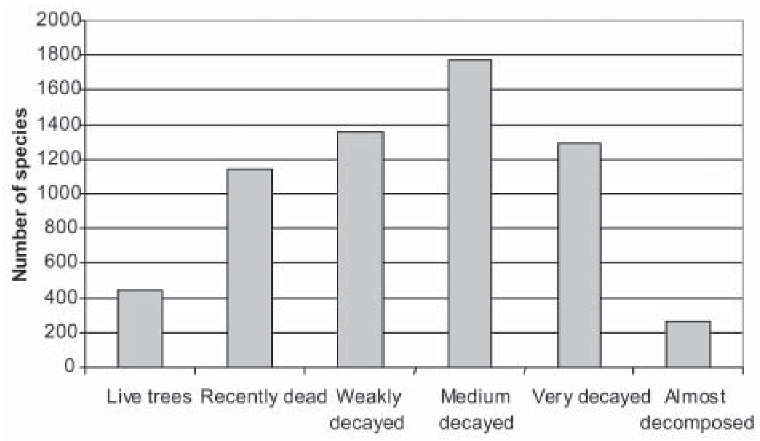

Figure 2. Total number of species that have been recorded on different decay stages of dead wood in Scandinavia.

Dimension preferences

The diameter of dead trees is another quality of dead wood that most species respond to. Of 3600 species, near 15% prefer small or small to medium-sized diameters (up to 20 cm). Somewhat more than 20% of the species prefer diameters larger than 20 cm, and additional 10% of the species prefer large diameters (> 40 cm). About 20% of the species are generalists without strong diameter preferences. In reality, all these proportions are larger since the diameter preference is unknown for about 35% of the species.

Landscape patterns

Different dead wood qualities are not evenly distributed in the forest landscape. The variation is caused by landscape properties such as topography, soil conditions and productivity. The input rate and volume of dead wood varies both temporally and spatially as a consequence of natural disturbance regimes in unmanaged forests, but it is reduced substantially by forestry in managed forests. Today, most of the Scandinavian forests are managed, and large areas with low abundance of dead wood and small areas with high abundance of dead wood characterize the landscape (see Figure 8).

This landscape pattern is important, since scattered presence of suitable dead wood qualities may be insufficient for the presence of common as well as rare wood-associated species. From several empirical studies, we know that many species occur frequently in areas with high abundance of suitable dead wood. When suitable wood substrate is sparse, the frequency of a species may be 15–25% of that found on equal substrates in localities where suitable wood is abundant (Nilsson 1997; Rukke and Midtgaard 1998; Sverdrup-Thygeson and Lindenmayer 2002; Stokland and Kauserud 2003). Other studies have identified 20–30 m^3/ha of dead wood as a threshold level below which many wood-associated species seem unable to survive (Økland et al. 1996; Martikainen et al. 2000; Angelstam et al. 2002), but the threshold level may be as high as 70 m^3/ha for very demanding species (Siitonen and Saaristo 2000).

An important message from these studies is that there is no linear relationship between the amount of specific dead wood qualities and the population level of wood-associated species. Thus, one needs to quantify the local abundance of different dead wood qualities in order to predict the presence of wood-associated species.

Other preferences

The purpose of this paper is not to make a comprehensive review of all substrate preferences of wood-associated species. Thus, it should only be mentioned that the majority of species show additional preferences, like for different parts of the tree (both vertically from branches to roots, and radially from the bark to the heart wood), hollow trees, microclimate around the dead tree (sun-exposed to shady) and burned wood.

3. Methodology

3.1 Dead wood inventory

A comprehensive dead wood inventory should include all forms of woody debris including lying dead trunks (logs) and large branches, standing dead trees (snags), and ideally also dead parts of still living trees. It is customary to subdivide dead wood into fine (FWD) and coarse woody debris (CWD) using 10 cm diameter as a threshold value. Harmon and Sexton (1996) identified the 10 cm limit as crucial for wood decomposition rate. Below this limit, the decomposition rate increased exponentially with decreasing diameter, and above the limit it decreased slightly with increasing diameter.

Plot size

In Scandinavia, inventory of coarse woody debris is a part of the National Forest Inventories using plots of 0.025–0.05 ha. Even if the individual plots are small, they produce reliable regional and national estimates of various dead wood qualities. A 0.05 ha plot does not properly measure the local abundance of dead wood, however. In a separate Norwegian case study a subset of the NFI plots were enlarged to 0.5 ha (Figure 3) for estimating local abundance of dead wood and making a biodiversity inventory. A simulation study based on these plots revealed that plots should have at least a size of 0.2 ha to document the local abundance and composition of different CWD qualities (Stokland 2001, see 3.3 for further details).

Inclusion and positioning of CWD units

Two rules may be applied for including CWD units within the plot: a) *strict inclusion* of the part of a CWD unit that is actually within the plot area, or b) *basal point inclusion* including whole CWD units with the basal point inside the plot area. In the first case one needs to find the exact intersection point(s) between the log and the plot border, and then measure the volume within the plot. Notice that one misses the important parameter *basal diameter* when the basal point is outside the plot area. In the other case, one only need to decide whether the basal point is inside the plot border and then measure the whole unit including the part lying outside the plot area (see Figure 1). The included part outside the plot is balanced by omitted

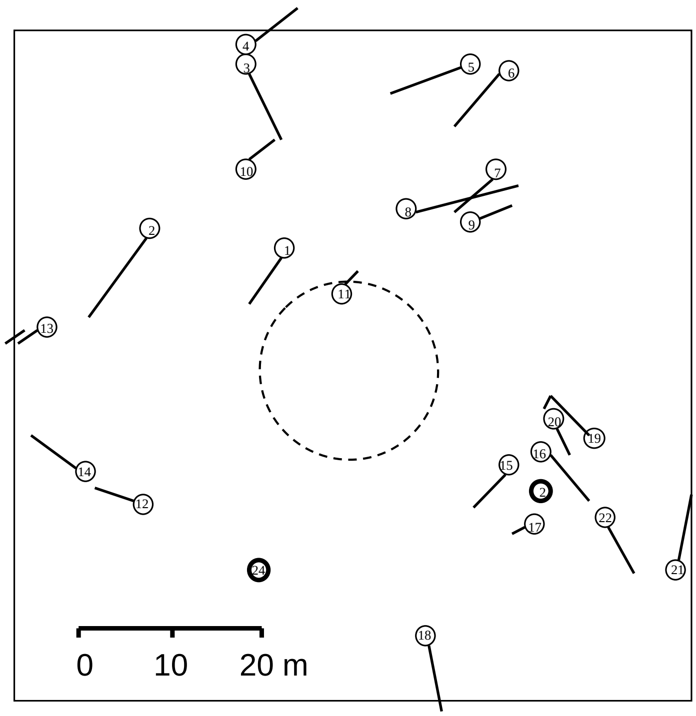

Figure 3. A NFI plot of 0.025 ha enlarged to 0.5 ha for estimating local abundance of dead wood. The circular symbols with numbers represent the position of the basis of individual CWD units. The attached lines indicate the direction and length of downed logs. Symbols without lines represent snags.

parts of logs within the plot area with the basal point outside the plot area. Line intersect sampling represents another method for quantifying volumes of dead wood. This method can be used in combination with ordinary sample plots and it has been successfully adopted in the American Forest Inventory and Analysis program (Waddell 2002).

It is useful to position the CWD units within the plot (Figure 3). For this purpose two parameters are necessary and a third is very useful: x and y coordinates of the basis of each CWD unit, and the direction of lying logs. This is particularly the case if one later wants to repeat the inventory of the plot. One then only needs to re-classify the type of dead wood (e.g. from standing snag to downed log) and the decay class of previously present CWD units, whereas full measurements only are needed for new dead wood units. Another advantage of positioning the dead wood units arise if one makes an inventory of wood-inhabiting species in the plot. Then one can concentrate on species sampling as a separate activity and subsequently get dead wood (habitat) parameters from the dead wood inventory.

3.2 Dead wood parameters

Metric attributes

In order to calculate the approximate volume of dead wood, one need to measure a) *length* and b) *mid diameter* (logs). Furthermore, it is highly recommended to measure c) *basal (maximum) diameter* and d) *top diameter* to produce more precise volume calculations (Harmon and Sexton

1996, see also Waddell 2002 for measuring branching trunks). The basal diameter is also necessary for grouping dead wood into size classes (see 3.3). For lying dead wood, one should measure the whole CWD unit, i.e. also the top part with diameter < 10 cm. In one Scandinavian country, the 10 cm minimum diameter was used as a cut-off criterion near the tree top and the top was excluded. This procedure is not recommended, however. In the same manner as standing volume measurements include the whole trunk, volume measurements of CWD units should also include the top. This part of a dead wood unit is used by wood-inhabiting species.

For standing dead wood, it is often impossible to measure mid and top diameter. One can instead measure the basal diameter and estimate the volume from tabulated relationships between basal area and volume. Often standing dead wood is broken and one should therefore judge the *proportion of the snag being intact* for estimating the volume of standing dead trees.

Type of dead wood

This is a simple classification of dead wood into the following alternative categories: snag, log, and fallen branch (Table 2). High stumps (> 0.5 m) can be classified as snags with a small proportion of the snag intact.

Tree species

Ideally, all dead wood should be identified according to tree species. One should always use positive identification criteria and not assume the probable species from the live trees in the stand. For recently dead trees it is quite easy to identify the species, but the identification problem increases for well-decayed trunks. The best identification criterion is the bark surface and structure. With a little experience one can often identify tree species in decay classes 3 to 5 (see below) based on bark remains. If no bark is found along the trunk one should try to locate the stump as one often can find pieces of bark here. The angle between the trunk and the branches is another criterion for tree species identification. In Scandinavia, this criterion is sometimes useful for separating between *Pinus sylvestris* and *Picea abies*, and for separating between coniferous and broad-leaved trees. It is useful to build up experience from observing live trees. A third criterion for identifying wood is by means of rot type. For example, fungi of the genus *Chlorociboria* stain the wood with a distinct green colour, and in Scandinavia these fungi only decompose broad-leaved trees.

Even when it is impossible to identify the tree species, it is still possible to decide whether the dead wood is either *coniferous* or *broad-leaved* wood. This distinction is strongly recommended as it is relevant to pool different tree species into these categories later when reporting indicator values. However, the category *unidentified wood* is also needed.

Decay class

The stage of decay is a very important quality for predicting the associated species composition. In Norway and Finland, a 5-class system of classifying dead wood has been adopted (Figure 4), and the relationship between the classes and the degree of decomposition in terms of dry density loss has been quantified (Næsset 1999). This makes the system suitable also for quantifying carbon pools. The criteria listed in table 2 are used for defining the classes. This classification system corresponds closely to the 5-class system used in the American Forest Inventory and Analysis program (Waddell 2002).

Table 2. Qualitative variables used to classify dead wood properties of importance to biological diversity in Scandinavian monitoring. The variables are listed in boldface, and alternative classification values are listed under each. For the decay classes, the proportions of initial dry density remaining are interpreted by the author based on Næsset (1999).

Type of dead wood	
snag	standing dead tree (including high stump)
log	lying dead trunk
fallen branch	fallen branch with maximum diameter > 10 cm
Decay class	
recently dead	bark normally attached to the wood. Hardly any fungus mycelium developed under patches of loose bark; 100–95% of the initial dry density.
weakly decayed	loose bark, normally well developed mycelium between bark and wood. The rot extends less than 3 cm radially into the wood (as measured by pushing a knife into the wood); approximately 95–75% of the initial dry density.
medium decayed	the rot extends more than 3 cm into the wood, but the log still has a hard core and it is supported by stones, humps, etc. on the forest floor; approximately 75–50% of the initial dry density.
very decayed	rotten throughout the log. The log is shaped by the contours of the forest floor and the cross-section is often collapsed to an ellipsoid; approximately 50–25% of the initial dry density.
almost decomposed	the log is completely decomposed in sections, and the log outline is strongly fragmented. The remaining parts are often overgrown; approximately 25–5% of the initial dry density.
Tree species	
species list	each tree species represent a classification value
coniferous	tree species unknown, but possible to exclude broad-leaved tree species
broad-leaved	tree species unknown, but possible to exclude coniferous tree species
unknown	tree species, or group of tree species uncertain or impossible to identify
Mortality type	
a) dry snag	standing dead tree; mortality cause: competition, drought, insect/fungus attacks, fire
b) mechanically broken	splintered fracture surface; mortality cause: snow load, heavy wind, lightening
c) up-rooted	trunk intact, root uplifted; mortality cause: heavy wind
d) broken by rot	irregular fracture surface; mortality cause: often a previous dry snag, broken by rot
e) broken by beaver	tooth marks from beaver; mortality cause: the animal
f) cut by man	fracture surface even or with axe marks; mortality cause: cut by means of saw or axe
g) unknown, natural	fracture surface decomposed; possible to exclude cut by man
h) unknown	fracture surface highly decomposed; impossible to assess

Mortality type

Based on experience from more than 4000 logs, the mortality types listed in Table 2 and illustrated in Figure 5 have proven to be a robust classification system. The term *mortality cause* is avoided, because several mortality factors may be involved for single trees (e.g. drought and insect attacks), or specific causes may be difficult to identify (e.g. snow load and

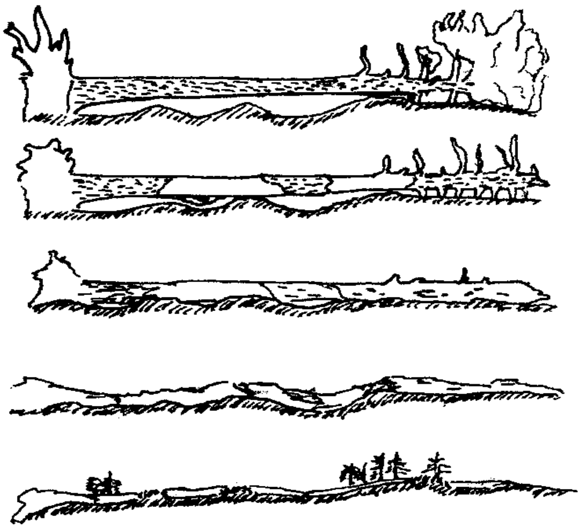

Figure 4. Visual appearance of the five decay classes used in the Norwegian NFI. See Table 2 for a description of the classes.

wind). The mortality type is useful for separating between natural mortality and dead wood left behind as logging residuals.

3.3 Local abundance and qualitative composition

By grouping the volume of dead wood into a two-dimensional matrix of decay and diameter classes, one can derive the *Coarse Woody Debris profile* of the site (Stokland 2001). The CWD profile facilitates forest history interpretation for the recent 200–500 years, and it has proven useful to predict the occurrence of wood-inhabiting fungi. We recommend to use a simple 2 × 2 matrix by splitting the decay classes into decay class 1–3 and 4–5, and the maximum diameter into 10–30 cm and > 30 cm. This sorting procedure can be carried out automatically on the data after fieldwork.

From the volume in each cell of the matrix one can classify the stand into alternative CWD profiles reflecting different forest histories (Stokland 2001). One profile, typically found in unmanaged and undisturbed forests, is characterized by large volume of dead wood in all four cells, indicating that there has been a substantial and regular input of dead wood from various diameter classes for a long time at the site. A second profile is characterized by large amount of weakly to moderately decayed wood, but small amounts of very decayed wood. This profile reflects a situation where the standing volume (or dead wood) has been removed regularly in the past, but where the input of deadwood has increased the last decades. A third profile is characterized by absence of very decayed wood, indicating a previous period of intensive wood removal (either by forestry or fire) that has produced a distinct gap in the dead wood input. A fourth profile is characterized by very small amounts or absence of dead wood in all four cells. This is a profile typical for regularly managed forests both in the past and more recently.

In a simulation study based on 128 0.5 ha plots, the volume limits for deciding between absence, small, or large volumes of dead wood were halved and doubled (Stokland 2001). It

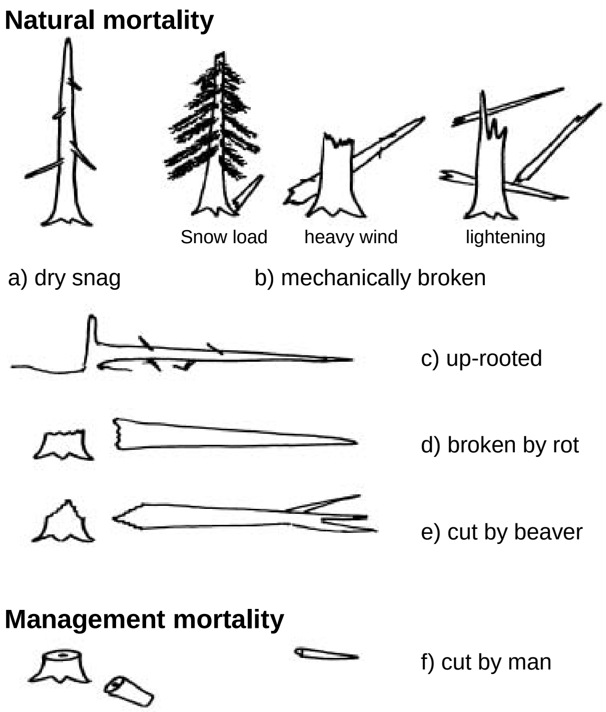

Figure 5. Visual appearance of different mortality types. See Table 2 for a description of the classes.

then turned out that 70% of the plots were classified to the same CWD profile in both cases, and the changes were internally between the two first profiles and the two last profiles mentioned above. This indicates that the CWD profiles are robust towards moderate errors in the volume measurements.

4. Results

4.1 National and regional dead wood indicators

Dead wood has been quantified in the National Forest Inventories of Finland, Norway and Sweden using type of dead wood, tree species and decay class as qualitative classification parameters. Based on these data we have produced statistics for volume of dead wood, subdivided on snags and logs as well as three decay classes for five geographical regions (figure 6). For Sweden and Finland, there was a distinct trend of increasing volume of dead wood from south to north, which corresponds to different intensity of forest management.

We grouped the dead wood volume into the following tree species classes: Pine, Spruce, unidentified coniferous wood, broad-leaved wood, and unidentified wood. For each tree species class we subdivided the volume into the same types and decay classes as in Figure 6. In this way, the volume can be presented as total volume or proportions of different tree species (Figure 7).

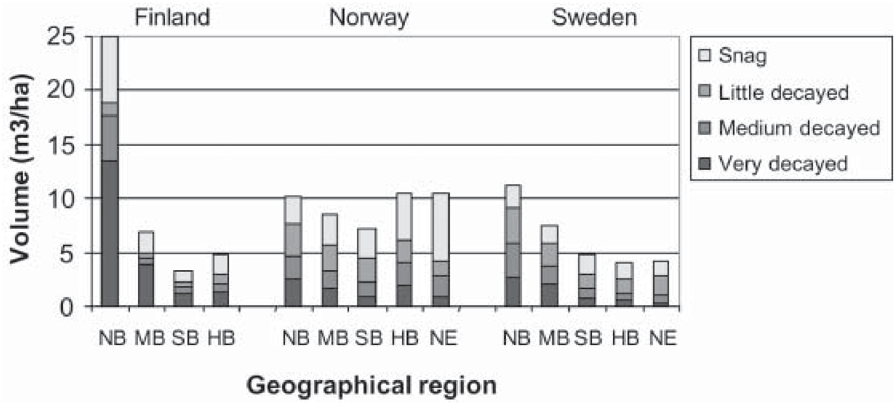

Figure 6. Volume of dead wood subdivided on type of dead wood and decay classes in different eco-geographical regions in Finland, Norway and Sweden. The regions are: NB = North boreal, MB = Mid boreal, SB = South boreal, HB = Hemi boreal, NE = Nemoral

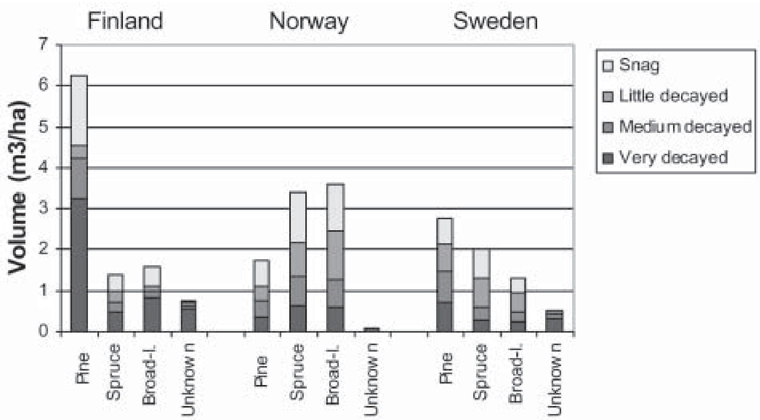

Figure 7. Volume of dead wood in subdivided on tree species and decay classes in Finland, Norway and Sweden.

4.2 Local abundance of dead wood

Volume of dead wood

Based on enlarged NFI plots one can estimate the volume of dead wood at the stand level and subdivide volume into different components according to the qualitative dead wood parameters. In a case study from SE Norway, a stratified sub-sample of NFI plots were enlarged to 0.5 ha, and it turned out that the local abundance of dead wood varied between 0.3 and 13.2 m^3/ha in recently cut and middle-aged managed forests (Figure 8). In mature forest (i.e. logging mature and over-mature stands) the volume of dead wood varied substantially. Most of the stands (n = 15) varied in the same interval as the younger stands, but five stands varied between 25 and 65 m^3/ha (Figure 8). Most of this wood was little to medium decayed, i.e. resulting from the current

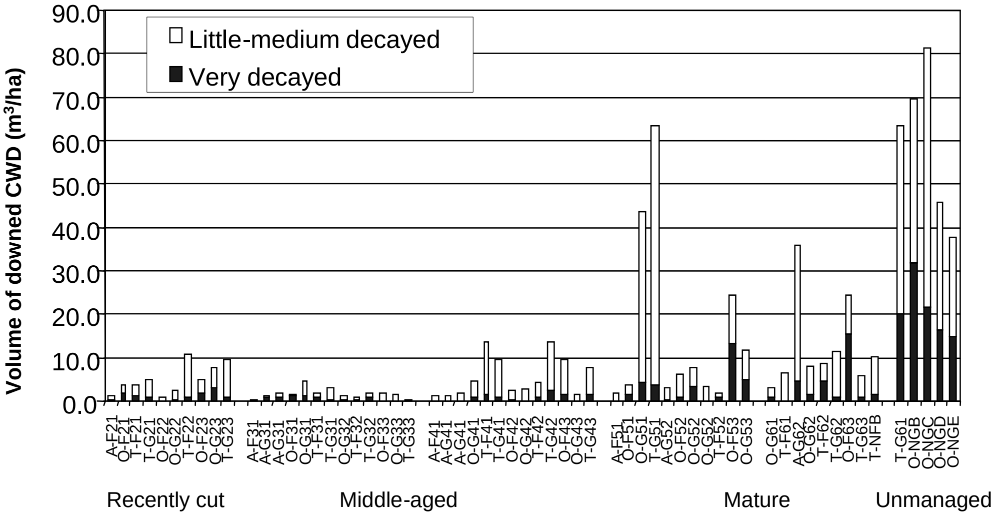

Figure 8. Local abundance of dead wood in a stratified sample of NFI plots from different development classes in coniferous stands in SE Norway.

stand through natural mortality. Common for nearly all managed stands was a low abundance of very decayed wood (decay classes 4 and 5), indicating that timber has been efficiently extracted in previous stand rotations. Only two managed stands exhibited volumes of very decayed wood above 10 m^3/ha (Figure 8). These two stands were located near the mountain range – a region characterized by low-intensive forestry. In contrast to the managed stands, five unmanaged stands exhibited volumes of downed logs between 40 and 80 m^3/ha, of which 15–32 m^3 were very decayed logs (to the right in Figure 8).

Dead wood profile

By grouping the volume of dead wood into a two-dimensional matrix of decay and diameter classes the CWD profile (see methods) was used to classify the stands into different CWD patterns indicating habitat quality for wood-inhabiting organisms. In a stratified sub-sample of 111 Norwegian NFI-plots in coniferous stands, it turned out that 8 stands (7%) had strong CWD continuity, i.e. sites with high habitat quality for wood-inhabiting species. Seven of these were found in mature stands, and only one in a middle-aged stand. 14 stands (13%) had weak CWD continuity, i.e. sites of moderate habitat quality for wood-inhabiting species. The remaining 89 sites had either small CWD volume (22%) or continuity gap in dead wood (59%), two CWD patterns that represent low habitat quality for wood-inhabiting species.

4.3 Indicator validation

It is an underlying assumption that biodiversity indicators predict the forest biodiversity. It is crucial to establish the link between different indicator states and the biodiversity component it is intended to indicate. In the Norwegian case study, a species inventory of wood-inhabiting fungi was carried out together with detailed dead wood registration in 124 coniferous stands. The

Table 3. The effects of stand parameters on species richness of all wood-inhabiting fungi and red-listed fungus species modelled by multiple regression (n = 124 stands). The asterisks indicate significance levels of the parameters: * p < 0.05, ** p < 0.01, *** p< 0.001. A) Models based on traditional stand parameters. B) Models based on traditional stand parameters plus total volume of downed wood. C. Models based on traditional stand parameters plus total volume and different qualities of downed wood.

	Altitude	Productivity	Stand type	Age class	volume ha^{-1}	logs ha^{-1}	% br-l. wood	CWD profile	r^2
A) Traditional									
All species	-	*	(*)	***					0.37
Red-listed species	*	(*)	-	***					0.35
B) Traditional+ volume									
All species	-	-	-	-	***				0.66
Red-listed species	(*)	-	-	-	***				0.69
C) Traditional + wood qualities									
All species	*	-	-	-	*	***	**	**	0.86
Red-listed species	-	-	-	*	***	***	-	**	0.79

number of fungus species varied between 4 and 100 species per 0.1 ha in the coniferous stands, and altogether more than 700 species were encountered in the full study which also included 35 broad-leaved stands. Traditional stand parameters like forest type (tree species composition), site productivity, age class and altitude accounted for 37% of the variation in species richness and 35% of the variation in red-listed fungi (Table 3a). When the local abundance of lying dead wood was included among the predictor variables, this factor increased the explained variation to 66% for all fungi and 69% for the red-listed fungi, whereas the traditional stand parameters lost their explanation power (Table 3b). When different dead wood qualities were added as predictor variables, the explanation power increased to 86% for total species richness and 79% for the red-listed fungi (Table 3c). In other words, if one measures the total volume as well as different dead wood qualities at the stand level, one can predict the species richness of wood-associated fungi with high precision.

5. Discussion

In this paper we have described operational procedures and parameters for conducting a detailed dead wood inventory. In a biodiversity context it is necessary to monitor the stock and change of various dead wood qualities since different species use different dead wood qualities. In other words, it is impossible to capture the dead wood diversity in one value summarizing the volume of dead wood (neither the diversity in dead wood qualities nor the associated species richness). Since about 25% of all forest species live in dead wood, one can argue that more than one indicator is relevant. On the other hand, the number of indicators should be kept low to avoid too much detail.

At present, two biodiversity dead wood indicators have been defined in Europe. One is the MCPFE indicator 3.5 "Volume of standing and lying dead wood by different forest types". This indicator is officially agreed upon from 41 European countries (MCPFE 2003), and prior to 2008 all countries shall report dead wood statistics according to the MCPFE process. The second indicator is the BEAR indicator "Volume of dead wood, subdivided by standing/lying, tree species and decay class" (Larsson 2001). This indicator has status as a recommended indicator from an expert panel, but no officially accepted status. However, the ministerial processes behind the MCPFE and BEAR indicators agree to develop the indicator frameworks in cooperation, and it is likely that the indicator on dead wood will develop in the future.

In Europe, the ICP/Forest (International Co-operative Programme on assessment and monitoring of air pollution effects on Forests) considers to widen the scope to include biodiversity assessment. Furthermore, the upcoming EU resolution Forest Focus also highlights the need for biodiversity monitoring (European Commission 2002). Both the ICP/Forest and Forest Focus specifically address the need to monitor the state and development of dead wood for biodiversity purposes, but do not define dead wood qualities in detail.

So far, no standardised field protocol exists for making detailed dead wood inventories at a European level. Since several initiatives on dead wood indicators and monitoring are developing, it is timely to assess adjustments of current indicators and monitoring protocols: can monitoring procedures be adjusted so that the prediction power of the indicator increases significantly while the sampling costs increase modestly?

5.1 Data sources and sampling

Data on dead wood must be established through field inventories. Like the Scandinavian countries, many other European countries have national forest inventories that are suitable for dead wood monitoring, and several countries have already included dead wood in their NFI systems. These NFI systems should constitute the empirical basis for future monitoring of dead wood stock and change because they represent well-functioning and stable data sources. Since there are numerous ways to specify sampling techniques, the NFI systems should develop their field protocols in close cooperation.

Relevance of different dead wood attributes

All dead wood inventories quantify the volume of dead wood by measuring the trunk length and diameters at various points. In the NFI systems of Finland, Norway and Sweden, the diameters were measured at different points on downed logs. As a result, we could not produce standardised statistics of dead wood subdivided on different dimension classes. In this paper we have described two procedures for including dead wood on sample plots: strict inclusion and basal point inclusion. In the first case, maximum diameter is not measured if the basal point is outside the plot. In this case we strongly recommend that one also measures the maximum diameter since it represents an important biodiversity attribute of dead wood. If one adopts a line intersect sampling procedure, one should include the maximum diameter as a standard measurement.

In addition to the metric attributes, at least three qualitative parameters are very important for the species diversity associated to dead wood: type of dead wood (standing/lying), tree species and decay class (assuming that maximum diameter is included among the metric attributes). As shown in the species diversity section of this paper, these properties represent important qualities on which the majority of the wood-inhabiting species are specialised. These qualitative parameters should be included in all inventories measuring volumes of dead wood. Fortunately, this qualitative information is rapidly collected and it takes only a minor fraction of the time used for measuring the metric attributes.

For several purposes, it is interesting to estimate input rates of dead wood (i.e. through natural mortality and logging residuals). If this dynamic aspect is within the scope of a monitoring programme, it is also relevant to include the mortality type as a qualitative parameter for judging the cause of death to quantify how much different factors contribute to the amount of dead wood in different forest types. Several species seem to prefer dead wood that has died through natural mortality, but it is still unclear whether this parameter is important for indicating the species diversity.

Plot size and geographical scale

Above we argued in favour of the NFI systems for dead wood monitoring. In these systems, the inventory plots are typically smaller than 0.05 ha. On the basis of many plots, this plot size is sufficient to establish reliable statistics for total and average volume of various dead wood qualities on a national and regional scale. On the other hand, the typical NFI plot size is insufficient for measuring local abundance and qualitative composition of dead wood.

National and regional figures for dead wood are interesting in a biodiversity context, but since several species appear to need local dead wood concentrations above certain threshold values (see Landscape patterns, section 2) it is important to establish statistics on local abundance of dead wood. Such statistics are also relevant when setting political goals in relation to indicator states (see below). The results on indicator validation shows that information about local abundance and qualitative composition of dead wood is a reliable predictor of total species richness as well as the occurrence of red-listed species. For the purpose of quantifying local abundance of dead wood, one needs to have larger plot sizes than those normally used in NFI. Stokland (2001) found that a minimum of 0.2 ha was necessary for quantifying local abundance and qualitative composition of lying dead wood in Scandinavian boreal forests. However, the plot size may need to be somewhat larger in areas where dead wood is scarce (e.g. up to 0.5 or 1.0 ha). If one adopts a line intersect sampling method, one should have a minimum transect length of 200 m in order to quantify the local abundance of dead wood (see Stokland and Sippola 2004).

There is a significant time cost involved when enlarging the plot size for measuring local abundance of dead wood. In 0.5 ha plots with small amounts of dead wood (< 5 m^3/ha) it takes about 2 hours for one person to position and measure all CWD units, whereas it takes 1–2 days in plots with large amounts (50–100 m^3/ha). In order to get statistics on local dead wood abundance without expanding the time budget too much, one could expand the plot size for a stratified subset of the plots. Another strategy would be to apply quick visual (subjective) judgements of dead wood volumes at each sample site using an extended reference area of 0.3–0.5 ha. The judged volume should be subdivided into five components: standing dead wood and lying dead wood for each of the four cells of the CWD profile matrix (see 3.3). In addition to the visual assessments, quantitative measurements should be conducted within the ordinary plot size to establish accurate dead wood estimates for regional and national statistics. These measurements can also serve as a calibration basis for the visual judgements. A third strategy would be to apply the line intersect sampling method. This method is faster than plot-based methods and one avoids subjective judgements. Two points are important to get reliable data on local abundances of dead wood: a) the transect length should be minimum 200 m, and b) one needs to measure the volume of each dead wood unit crossing the transect line (see Waddell 2002, Stokland and Sippola 2004 for further details).

5.2 Reference levels, baselines and targets

Data on indicator states alone does not say very much unless they are put into perspective. One way is to compare the indicator states with some kind of *natural reference levels*. For dead wood we can define this kind of reference level – the amount of dead wood that is present in forests characterised by natural dynamics, i.e. resulting from the balance between natural mortality and decomposition, and where no live or dead wood is removed through forestry or other human activity. For boreal coniferous forests in Scandinavia we know that the level of dead wood in old growth, unmanaged forest is 60–90 m^3/ha and declines to 20 m^3/ha close to the timberline (Siitonen 2001). In unmanaged mixed broad-leaved forests reserves in Central Europe, the volume of dead wood is 90–200 m^3/ha, and in mixed *Abies-Picea-Fagus* forests in Central Europe it is 110–300 m^3/ha (Hahn and Christensen 2004).

A *baseline level* is typically the start value(s) of a monitoring series. For Scandinavian forests the first reliable national statistics on total volume of dead wood was established through the NFI carried out in the 1990s. These data show that the national averages in Finland, Norway and Sweden vary between 6.6 and 10.0 m^3/ha in productive forests (Stokland et al. 2003), but with large regional variation (Figure 6).

Target levels are politically defined goals specifying certain indicator state(s) that should be reached within a defined year. Among the Scandinavian countries, Sweden has been most explicit about defining targets for the amount of dead wood. The Governmental proposition 2000/01: 130 on environmental forest objectives states "… the amount of weakly decayed wood should increase at least 40% for the whole country and substantially more in areas where the biological diversity is particularly threatened …". It is also stated that this target should be reached within 2010. Weakly decayed wood is specified because this fraction is the only one can substantially change during a 10-year interval.

It is constructive of Sweden to define explicit targets within a well-defined time frame. By taking this step, one expands the use of biodiversity indicators from monitoring state and change to a quite different one – policy making. Failing and Gregory (2003) strongly argue that decision aid should be a major use of biodiversity indicators, but simultaneously, they warn about fallacies when designing indicators for policy use. In the context of decision-making, they argue that alternative policies should be considered before decisions are made.

Although the scientific basis is incomplete, we would like to indicate another option for Sweden. Instead of increasing the total (or average) amount of dead wood with a specified percentage, it is probably more cost-efficient to aim at a certain proportion of the forest landscape with local abundance above a certain threshold value sufficient for demanding species, e.g. 20–30 m^3/ha (see Landscape patterns, above). This is perhaps the intension of the phrase "…substantially more in areas where the biological diversity is particularly threatened …". It is unclear, however, whether this phrase refers to areas where there is a low abundance of dead wood (adverse conditions for threatened wood-associated species) or where there is a high abundance dead wood (where threatened species are likely to occur), and as long as "substantially more" remains unspecified, it is probably the 40% increase that will get most attention when assessing the policy success. The intention of this comment is not to evaluate the Swedish dead wood policy, but rather to highlight that it is relevant for dead wood inventories to establish frequency data on local abundance of dead wood. On the basis of such data one can set policy goals specifying amount of area primarily intended for timber harvesting (where the amount of dead wood can remain low) and other areas where biodiversity enhancement has priority (where increased amount of dead wood should be one sub-goal).

5.3 Further indicator development

Currently, there exists one dead wood indicator that has been developed and agreed upon: the MCPFE indicator that shall be reported within 2008. In the longer run, it is probable that dead wood biodiversity indicator(s) will be developed further. The MCPFE indicator "Dead wood by forest type" is relevant, but probably more important is the quality of the dead wood as such. This is acknowledged by the BEAR indicator that recommends the volume of dead wood to be classified by standing/lying, tree species and decay class (Larsson 2001). In figures 6 and 7 we have focused on the dead wood qualities as such, i.e. in a manner that is close to the BEAR indicator. However, in the Scandinavian NFI systems forest type has also been classified on each plot. Thus, the volume can easily be recalculated as volumes by forest types according to the MCPFE indicator.

This brings up an important aspect, namely that sampling of dead wood in monitoring systems should be rather detailed from the beginning. Then the volumes can later be re-calculated in response to changed specification of indicators over time. Although we recommend that several qualitative properties should be sampled when measuring dead wood volumes, we do not want to suggest that a corresponding number of indicators should be developed. Instead we recommend development of a hierarchical indicator that allows the volume to be aggregated and disaggregated with different degrees of detail corresponding to different user needs (see Failing and Gregory 2003).

In addition to a hierarchical volume indicator, we also recommend development of an area-based indicator summarizing the proportion of stands with different levels of local abundance of dead wood (intervals of m^3/ha or number of CWD-units/ha) and different dead wood profiles (i.e. a summary of the decay and dimension qualities). As shown in the indicator validation part of the results, such stand properties had very strong prediction power for total species richness as well as the abundance of threatened species (Table 3). Again, we do not want to specify an indicator in detail, but we want draw attention to the importance of quantifying local abundance as well as qualitative composition of dead wood.

5.4 Indicator validation

The amount and quality of dead wood is hardly a biodiversity value itself, but instead a means to enhance the diversity of wood-associated species (see Failing and Gregory 2003 for a good distinction between means and ends in a biodiversity indicator context). Thus, one should not judge the success of a biodiversity policy on dead wood only on the basis of whether a well-defined indicator target is reached or not. In addition, one also needs to know whether the adopted indicator is a good predictor of the biodiversity itself.

In this paper, we give an example of indicator validation in relation to one organism group. That study was carefully designed to cover a broad range of forest types, development classes (stand ages), and a wide altitude interval. Furthermore, different dead wood qualities were quantified in great detail. This resulted in large environmental variation and large variation in species richness per plot. This is the reason why the statistical model (table 3) became so good – thee was a large variation to be explained and the relevant predictor variables had been quantified across a wide range of environments. We call for similar studies to be carried out on fungi in other forest types in Europe, as well as for other organism groups. Although, the study presented here is a single study from boreal forests, it is very promising and suggests that several dead wood qualities are needed to develop proper biodiversity indicators.

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Session 4: Stand-Level Indicators: The Heterotrophic Phase

Wood-inhabiting Fungi as Indicators of Nature Value in European Beech Forests

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Abstract

Wood-inhabiting fungi represent a highly species rich and ecologically important organism group in natural forest. Due to exploitation of forest resources most wood-inhabiting organisms have declined in Europe, and many specialised species are severely threatened on the continental scale. This paper presents a set of 21 fungal indicator species assumed to be usable in conservation priorities in European beech (*Fagus sylvatica*) forests. To support the proposed indicators information on fungal species diversity from 106 beech forest reserves in Europe is analysed. The presence of the indicator species is compared to the actual amount of dead wood per hectare in a subset of the sites, and it is shown that the presence of indicator species is only weakly related to the amounts of dead wood. Accordingly, it is suggested that the fungal indicator species can add valuable information relevant for prioritisation in forest conservation.

Keywords: biodiversity; biotic integrity; dead wood, forest reserves; habitat quality

Introduction

Beech dominated forest ecosystems are among the most important natural habitats in Central and Western Europe and among the most exploited forests of the world. Cutting of wood for timber and firewood has been intensive for several centuries and remnants of virgin forest are only found in remote mountain areas. In last decades several European countries have taken

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initiatives to designate new forest reserves in order to protect biodiversity. An evaluation of the effects of these initiatives is important.

Forests are complex ecosystems and several habitat factors influence the actual species diversity. A number of more or less distinct forest habitats of importance for biodiversity can be separated, e.g. the forest floor and its humus layers, the bark of living trees as well as decaying wood. Within each habitat type different factors influence the community structure and its species diversity. Dead wood is a special habitat in this context as its abundance differs considerably between natural and managed forests. Rich communities of wood-inhabiting organisms are expected only in sites with considerable amount of dead wood.

Habitat presence is, however, not the only factor determining the species diversity of wood-inhabiting organisms. The overall climatic regime is an important factor influencing the regional species pool, but also spatial and temporal aspects are highly important. In the European beech forest zone, the long history of forest fragmentation has led to the formation of highly complex and dynamic “forest archipelagos” in a matrix of farmland, plantations and urban areas. Island biogeography predicts large forests with unbroken continuity to host most of the original forest species pool, especially if situated close to the core area of the habitat. Forest continuity can be defined at several levels, e.g. as dead wood continuity, forest stand continuity, shade continuity or as continuity in the presence of ancient trees (e.g. Nilsson et al. 1995). Remnants of virgin forests embody continuity in all three aspects, while human exploited forests may represent various levels of continuity in each aspect. Further, the scale in time and space is important. For species with a limited dispersal potential, continuity at the very local scale may be crucial. On the other hand, species with good dispersal ability, but an association with temporal habitats may require continuity at a larger scale to survive, even though local stand continuity is of limited importance.

Due to the complexity of forest ecosystems it is broadly accepted that means to reduce complexity are necessary to make conservation biology practicable. The use of indicators is one widely accepted way of achieving this reduction of complexity. A variety of indicators has been suggested for habitat quality in European forests. These can be grossly separated in to structural indicators and indicator species. Typical structural indicators used to describe the conditions for forest biodiversity include, vertical structure, age class distribution and the amount of dead wood. Structural indicators, however, represent an indirect approach as they show, typically on a rather gross scale, how the house is built, but give no information on whether the inhabitants have moved in. Structural indicators are vague in describing important but subtle habitat qualities, e.g. forest climate and to account for the role of forest continuity on its various levels and scales. In contrast, monitoring of indicator species represent a more direct measurement of at least a selected fraction of the realised biodiversity.

With their dependency on old trees and dead wood, wood-inhabiting fungi are potentially suitable as indicators for dead wood associated biodiversity in general. In Fennoscandian boreal forest fungal indicator species have already been used in conservation prioritisation (Kotiranta and Niemelä 1996; Bredesen et al. 1997; Hansson 2000; Nitare 2000), and even for Corsican pine forests an indicator system has been suggested (Norstedt et al. 2001). In the nemoral zone attempts to use fungal species as indicators of forest habitat quality have been more sparse. Heilmann-Clausen and Christensen (2000) published a list of 42 fungal species suggested to be indicative for beech forest of high conservation value in Denmark. Since the proposal, information from more than 100 sites in Denmark has been used to evaluate the list. The system has later been evaluated and adapted in Belgium and The Netherlands (Walleyn and Vandekerkhove 2003; Veerkamp 2003) and used as a case on evaluation of selection strategies for forest reserves in Denmark (Strange et al. 2004).

Selecting indicators

The development of indicator systems has several phases. The first phase should include a normative evaluation of the conservation baseline, focussing either on species diversity, ecosystem health, biotic integrity or other concepts (e.g. Caro and O'Doherty 1999). The second phase typically involves a simple proposal of a list of indicators, e.g. species, which by experts are expected to be good indicators following the chosen baseline. The third phase involves evaluation and testing of the ability of each indicator to point to the conservation baseline, resulting in the fourth step in a revised list of suitable indicators.

Evaluation of the present set of indicator species

In the present case we have followed the biotic integrity baseline (e.g. Goldstein 1999), since we find this to be the only one adequate for conservation efforts in natural forest habitats. Biotic integrity emphasizes the composition of biotic communities and their relation to the natural history of their habitats. The focus is on protection and restoration of natural habitat conditions which are typical for a region, rather than on improving regional biodiversity, e.g. species diversity, which may actually increase in degraded landscapes, where exotics, often with a global distribution, may replace native species with a long local history. The proposal of 42 species suggested as indicative of habitat quality in Danish beech forest (Heilmann-Clausen and Christensen 2000) qualify as the second phase. The selection of indicator species in this was done ad hoc based on a combination of factual ecological knowledge with field experience and intuition. At present we are in the third, evaluating step in which we aim on testing the actual indicator value of the suggested species.

Of the species proposed on the first indicator list, several have appeared to be more common in other parts of Europe or to have less strict habitat requirements than expected. In addition some of the suggested species were too difficult to identify, had short sporulation periods or were very difficult to find. We therefore formulated more strict criteria in the evaluation of the species potential indicator value. Following these, all potential indicator species should be strictly wood-inhabiting, and showing a clear preference for old-grown living or death trees. Hence, potential indicators should either be heart-rot agents or late stage successor species confined to large diameter death wood. In addition to these ecological criteria we found it important also to take practical considerations in to account. Thus potential indicators should produce easily assessable sporocarps of large size or cover and have a long and stable fructification period. Further, sporocarps should posses clear and stable macroscopical characters allowing easy determination in the field.

Besides this soft evaluation procedure we are making progress in a statistical testing procedure in which we aim to test a) whether the suggested indicator species really point to sites showing a high degree of biotic integrity and b) if the suggested indicators are more precise than obvious structural indicators, i.e. death wood amounts and quality. We have collected species lists of wood-inhabiting fungi from 106 natural or semi-natural beech forests in Sweden, Denmark, The Netherlands, Belgium, United Kingdom, Germany, France, Poland, Czech Republic, Slovakia, Hungary and Slovenia (Figure 1), which we are using in the testing procedure. These lists are based on a combination of published data (Blaschke 2003, Dickson and Leonard 1996, Hocevar et al. 1995, Kuthan et al. 1999, Kost and Haas 1989, Nuss 1999), own observations, and information from several mycologists from

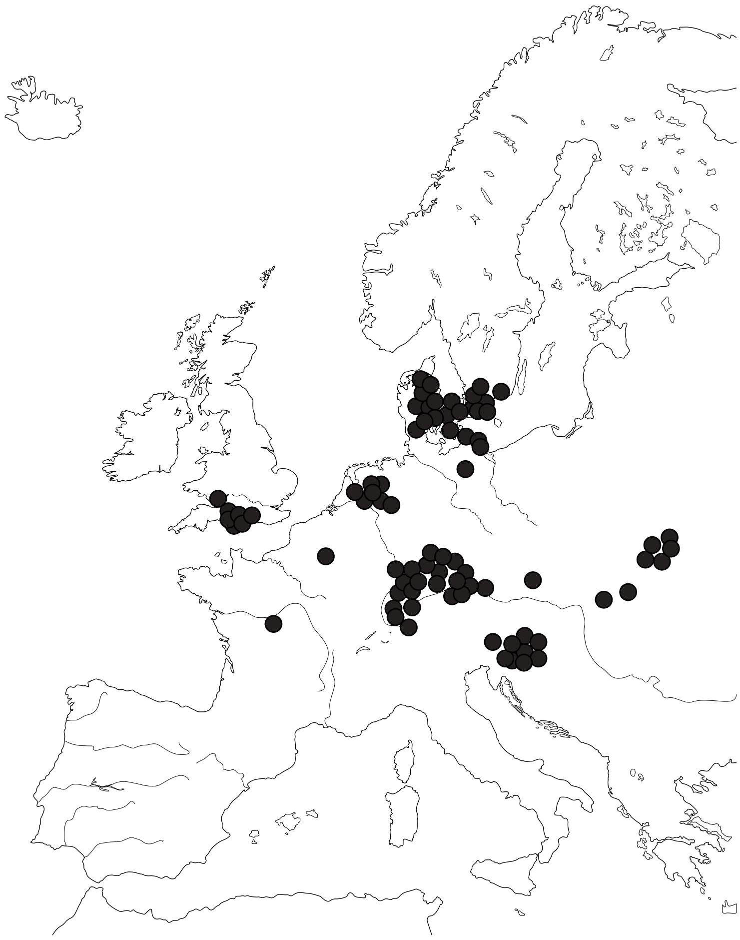

Figure 1. Map showing the sites included in the analysis

different parts of Europe (see acknowledgements). Only information on selected groups of macrofungi has been collected. Within the basidiomycetes all morphological groups, excluding fully resupinate corticoid fungi, are covered, while non-stromatic pyrenomycetes and inoperculate discomycetes with sporocarps regularly smaller than 10 mm were not taken in to consideration within the ascomycetes. The complete list contains more than 350 species recorded on beech wood in at least one of the sites.

Proposed list of indicators for European beech forest

Based on a preliminary evaluation of the collected material we here present a revised list of 21 species including 17 species from the Danish list (Heilmann-Clausen and Christensen 2000) and four new species (*Ceriporiopsis pannocincta*, *Lentinellus ursinus*, *Mycoacia nothofagi*, *Pholiota squarrosoides*). Table 1 shows the list and the ten most valuable European sites according to the suggested system.

Figure 2. The virgin forest of Rozok in Eastern Slovakia is among the most valuable sites for fungi connected to dead beech wood.

Amount of dead wood versus indicator presence

Structural indicators are often easy to monitor and understand for foresters or technical staff, which traditionally perform forest monitoring. Accordingly, they are easy to include in existing monitoring programmes.

Monitoring of indicator species typically require some level of specific knowledge and does require extra training of the monitoring team. The professional use of indicator species rather than structural indicators can be justified if they represent either a faster monitoring procedure or if they yield more relevant and precise information.

As an attempt to answer the latter question we have compared the presence of the suggested indicator species with information on the amounts of dead wood for 25 forest reserves (Christensen and Hahn 2003). This comparison clearly shows that a rich presence of fungal indicator species is related to other factors than dead wood amounts alone. (Figure 3), but it is still to early to tell whether these “other factors” relate to forest continuity or other aspects of biotic integrity.

Continuity and spatial aspects

The use of fungi as indicators of ecological continuity has recently been seriously questioned (Nordén and Appelquist 2001; Rolstad et al. 2002), based on the obvious potential of fungi

Table 1. The ten most valuable beech forest sites based on the occurrence of twenty one suggested indicator species. *) Total list of species not completed.

Species	Stuzica NR, Slovakia	Rozok NR, Slovakia	Fontainbleau, France	Zofin NR,Czech Republic	Jægersborg Dyrehave, Denmark	Havesová NR, Slovakia	Suserup Skov, Denmark	Strødam NR, Denmark	Rajhenav Rog, Slovenia	Öserdö NR, Hungary
Aurantioporus alborubescens			1	1	1		1	1		
Camarops tubulina	1	1		1			1	1	1	1
Ceriporiopsis gilvescens	1	1	1	1	1	1	1		1	1
Ceriporiopsis pannocincta	1	1	1	1		1			1	1
Climacodon septentrionalis	1	1	1	1		1		1		
Dentipellis fragilis	1	1	1	1		1			1	1
Flammulaster limulatus		1		1			1	1	1	1
Flammulaster muricatus	1	1			1	1	1	1		
Ganoderma pfeifferi					1		1	1		1
Hericium coralloides	1	1	1	1	1	1	1	1	1	1
Hericium erinaceum	1	1	1		1					
Hohenbuehelia auriscalpium	1	1	1	1		1		1		
Inonotus cuticularis	1	1	1	1	1	1	1	1	1	
Ischnoderma resinosum	1	1		1	1	1	1		1	1
Lentinellus vulpinus				1	1			1		
Lentinellus ursinus			1		1					
Mycoacia nothofagi	1	1	1							
Ossicaulis lignatilis	1	1	1	1	1	1	1	1	1	1
Pholiota squarrosoides	1	1	1	1	1	1			1	
Pluteus umbrosus	1	1	1	1	1	1	1	1	1	1
Spongipellis delectans	1		1		1	1	1			1
Lignicolous species total:	179	158	133*	69*	140	134	193	168	176	153
Indicator species	16	16	15	15	14	13	12	12	11	11

for long distance dispersal. Nordén and Appelquist (2001) thus noted that many wood-inhabiting fungi are likely to depend on a rich supply of microhabitats rather than the presence of long local forest continuity. This raises questions to the scale on which continuity should be assessed, an aspect which often is neglected in discussions on conservation of forest biodiversity.

Generally, fungi have good potentials for long range dispersal, due to their tiny windborne spores, which are often produced in waste amounts. Successful spore dispersal is however not the only factor determining the actual dispersal success. Suitable habitats for spore germination and development of mycelia on dead wood are often only available for very short periods and rare fungi may easily fail to arrive in time. This is especially the case in landscapes where human influence have changed the balance in dead wood availability e.g. by introduction of exotic tree species, or by selective utilisation of logs. Such changes tend to benefit fungal species with ruderal traits, which are normally less frequent under more natural

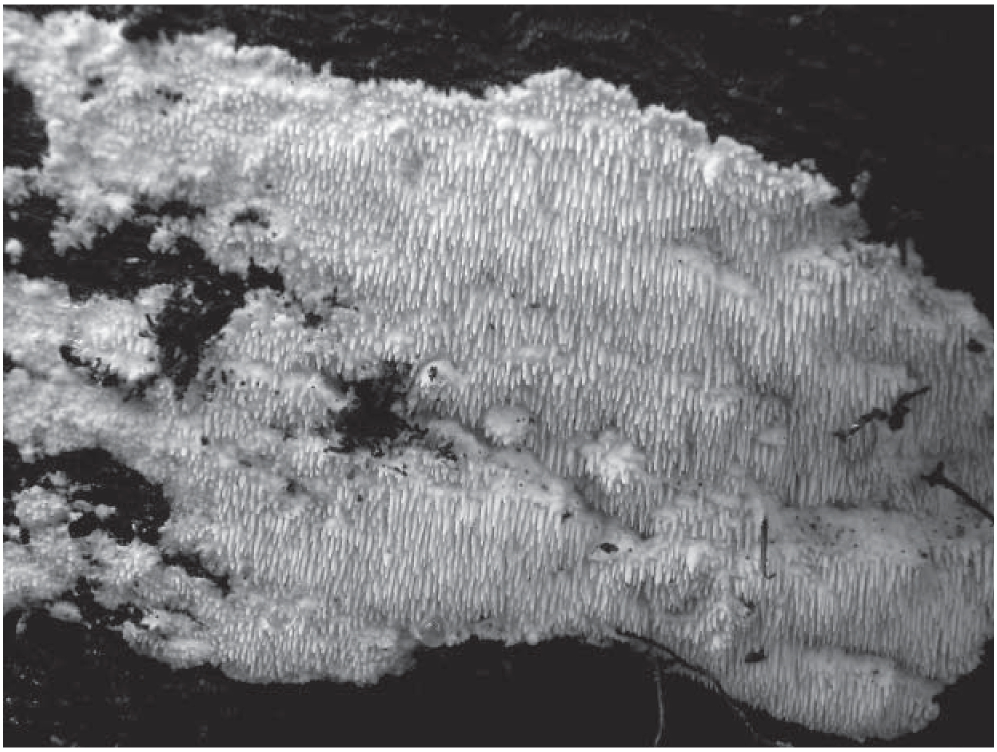

Figure 3. *Dentipellis fragilis* is suggested as one of twenty-one indicators for the value of natural beech forest.

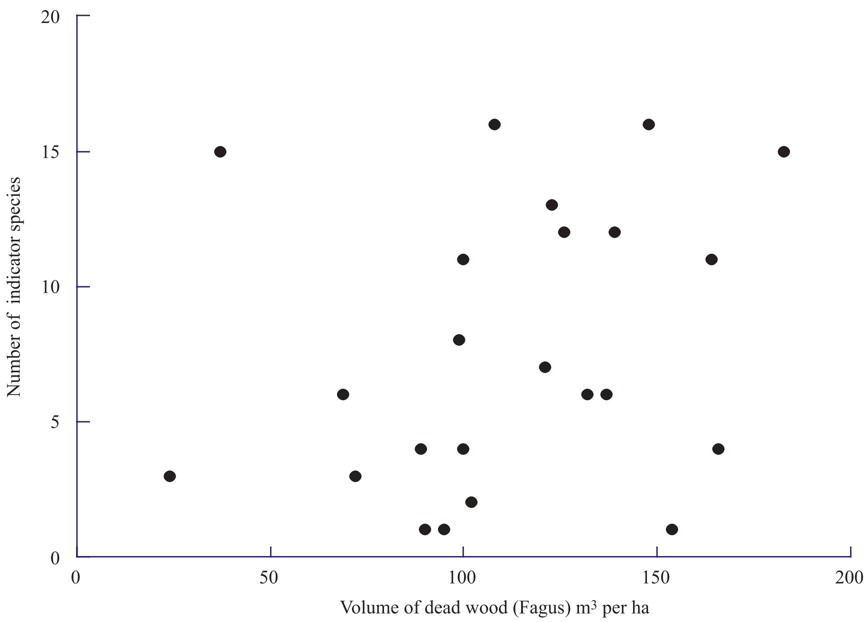

Figure 4. Relation between the volume of dead wood per hectare and number of indicator species.

conditions so that these gain a competitive advantage in the establishment situation (Sippola and Renvall 1999; Heilmann-Clausen 2003) Accordingly, wood-inhabiting fungi seem to have the biggest potential in indicating trends in forest continuity, especially dead wood and ancient-tree continuity at the landscape scale (cf. Heilmann-Clausen 2003; Sverdrup-Thygeson and Lindenmayer 2003). Our data suggest wood-inhabiting fungi to be especially suitable for identifying relatively large areas (25–10,000 ha) possessing dead wood continuity. Thus, our preliminary studies of fungal diversity in European beech forest show that the highest numbers of indicator species are found in the large areas of continuous beech forest in Central-Eastern Europe, which are still relatively rich in virgin forest remnants. On the local scale we found that the occurrence of indicator species in the Danish landscape relates to patterns in the presence of old-grown beech forests in a historical context (Heilmann-Clausen and Christensen 2003b).

Conclusion

Our evaluation process is not completed and it is too early to draw firm conclusions on the relevance of the suggested indicator species. Nevertheless, we feel convinced that wood-inhabiting fungi can provide relevant biodiversity information, which cannot be gathered directly from simple structural indicators, e.g. volume of dead wood per ha. The suggested indicator system can point to landscapes or large localities possessing dead wood and mega-tree continuity, which we regard as crucial factors determining the diversity and integrity of present forest habitats. Historical information on these more subtle aspects of forest continuity is typically difficult or impossible to obtain and the suggested list of indicators may therefore be an important shortcut in purposeful conservation initiatives aiming to secure the most valuable forests for wood-inhabiting organisms.

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Estimating the Heterotrophic Phase in Alpine Forests

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Abstract

Deadwood material is an essential element for the heterotrophic phase of the forest cycle, particularly in mountain forests where our examples are taken. Its management is now efficient in some forests and there is a need to evaluate the status of the deadwood in forest ecosystems. There is a strong relationship between the saproxylic fauna and its deadwood habitats. Saproxylic Coleoptera could provide relevant information about deadwood material in the forest landscape.

Four *Pinus cembra* managed forests (Savoie, France) were investigated, each presenting different dates of cutting. During two years, the deadwood qualities and quantities were recorded together with associated saproxylic Coleoptera. Collected data were both quantitative, qualitative and integrated the temporality of the decaying process.

It was possible to evaluate some underlying characteristics of the deadwood material present in forest and then, to evaluate the status of the forest heterotrophic phase using the presence of Saproxylic Coleoptera.

Keywords: Saproxylic Coleoptera; deadwood patterns; ecological continuity; biodiversity.

Introduction

An attempt to investigate how the forest management modified the forest heterotrophic phase was made in different forests of the northern French Alps. In the term *forest heterotrophic phase* we included the dead wood and its associated *saproxylic* fauna. The litter was not included here because of its short term decay.

Two assumptions were proposed: (1) management induced modifications of deadwood stocks, (2) these changes had an impact on the saproxylic fauna.

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Methodology

Four *Pinus cembra* forests were selected according to their last dates of management (Table 1). Theses forests were located at the timber line (2000 to 2300 m) in the Northen French Alps (Savoie, France).

Coarse Woody Debris (CWD) were assessed using the COST E4 protocol (Bock 2002; Dodelin and Lempérière, submitted) in 0.05 ha circular plots, on a 100 × 100 m grid (see Table 2).

Saproxylic Coleoptera were sampled using baited window traps and manual sampling (Dodelin 2002). This method was more efficient for xylophagous beetles living in fresh dead wood (decay class I and II).

The surface of window traps used in the four forest was comprised beetween 0.66 m^2 and 2.1 m^2. Numbers of beetles were reported to a mean surface of 0.92 m^2/forest.

Table 1. Date and intensity of management of the four study sites.

	Maturity	Time to the latest management	Volume exported
Villarodin	Mature	20 years	1949 m^3
Montonaz	Mature	40 years	2028 m^3
Tuéda	Mature	40 years	Unknow
Orgère	Over mature	60 years	2462 m^3

Table 2. Collected data according to the COST E4 protocol.

Standing dead tree	Stump	Log	Decay
$D_{130cm} \geq 7.5$ cm	$D_{130cm} \geq 7.5$ cm	D ≥ 10 cm at an extremity at least	I. Recently dead
H > 130 cm	H ≤ 130 cm		II. Decay < 1/3 D III. Decay > 1/3 D IV Totally decayed

Results

Total CWD volume

The volume of CWD in forest was lower 40 years after management and maximum after a recent management of the forest (Figure 1). 45% of the CWD was created by the management. We also noticed that some anthropic CWD remained after 60 years. This illustrated the slow decaying rate in the forests we studied where summers were dry and winters very cold.

Pools of CWD

Recent CWD (Decay class I) was scarce and the corresponding CWD recruitment (I) was estimated from 0.1 to 0.8 m^3/ha/yr (Figure 2). Latest stages of decay (Decay classes III and

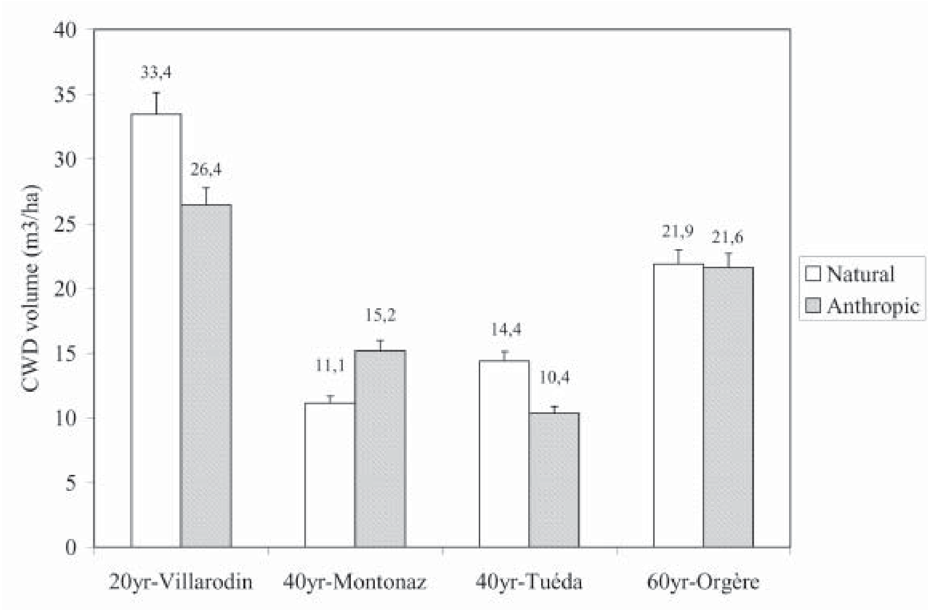

Figure 1. CWD volumes in the four *Pinus cembra* forests (m^3/ha).

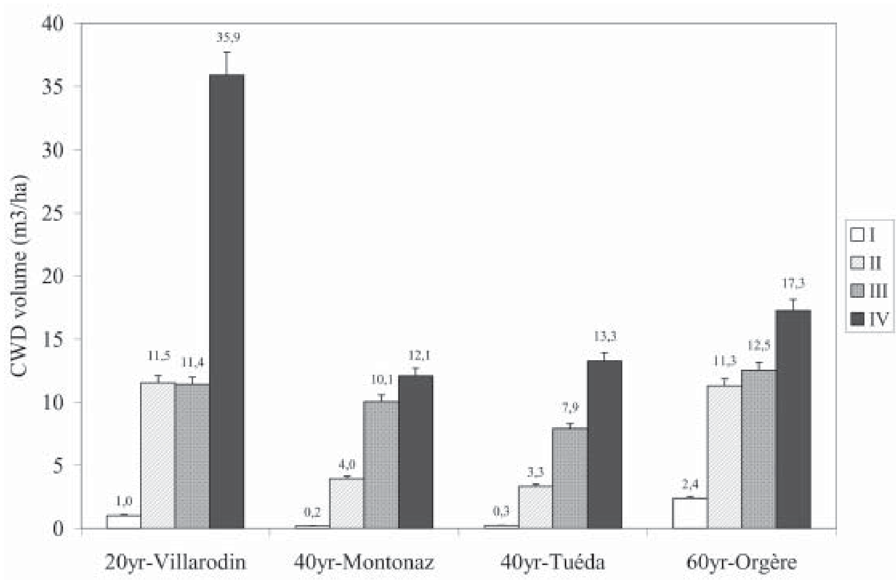

Figure 2. CWD volumes in the four *Pinus cembra* forests (m^3/ha). Details of the CWD patterns (I, II, III and IV are the decay classes).

IV) were the most common, in relation with the simple negative exponential model describing the decay rate (Harmon et al. 1986; Siitonen 2001). We could also notice CWD gaps in the decay classes I and II some in the forests managed 40 years ago. The effects of such gaps could influence the presence of xylophagous insects in the future.

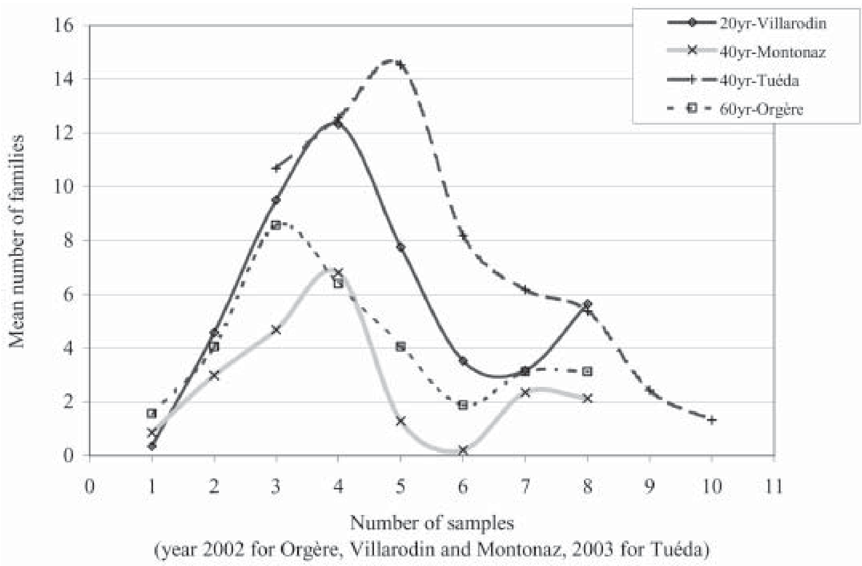

Figure 3. Mean number of families trapped (nb ind./trap).

Mean numbers of families

Mean numbers of the families of Coleoptera differed from site to site but showed similar evolutions in the time (Figure 3). The highest level of insects was found from mid-June to the beginning of July (samples 3 to 5)

Patrimonial species

We consider as ‘patrimonial’, the species that are only known from few (<15) stations in France and/or are given as ‘rare species’ in the literature. Such beetles were collected in 8 families of Coleoptera (Table 3). Rare beetles were found from the recently and the oldest managed forests. A lack of patrimonial species in the two forests managed 40 years ago could also be noticed.

Functional saproxylic groups

All the beetles were pooled in four functionnal groups: xylophagous, zoophagous, saproxylophagous and non saproxylic. Xylophagous and their zoophagous associates were more abundant in the forest managed 60 years ago (Figure 4). 40 years after the management, the number of saproxylic species was very low.

Conclusions

This preliminary study allowed to sort the four forests along a theoritical curve illustrating the deadwood quantity evolution over time (Figure 5). The forest management had an impact on the

Table 3. Patrimonial beetles found in 8 families.

Patrimonial species		20 yr-Villarodin	40 yr-Montonaz	40 yr-Tuéda	60 yr-Orgère
Buprestidae	*Chrysobothrys chrysostigma* L.	+			
Cerambycidae	*Semanotus undatus* L.	+			
Oedemeridae	*Calopus serraticornis* L.	+			
Elateridae	*Ampedus auripes* Reitter	+	+		
Buprestidae	*Anthaxia istriana* Rosenh.	+	+		+
Elateridae	*Ampedus scrofa* Germar	+	+	+	+
Melandryidae	*Hallomenus binotatus* Quensel	+	+	+	+
Pythidae	*Pytho depressus* L.	+	+	+	+
Elateridae	*Athous emaciatus* Binhagi	+		+	+
Cerambycidae	*Anastrangalia reyi* Heyden	+			+
Cerambycidae	*Judolia sexmaculata* L.	+			+
Cerambycidae	*Tetropium gabrieli* Weise	+			+
Lycidae	*Dictyoptera aurora* Herbst	+			+
Cerambycidae	*Oxymirus cursor* L.			+	+
Elateridae	*Melanotus castanipes* Paykull			+	+
Melandryidae	*Marolia leseigneuri* Nicolas				+
Melandryidae	*Zilora sericea* Sturm				+

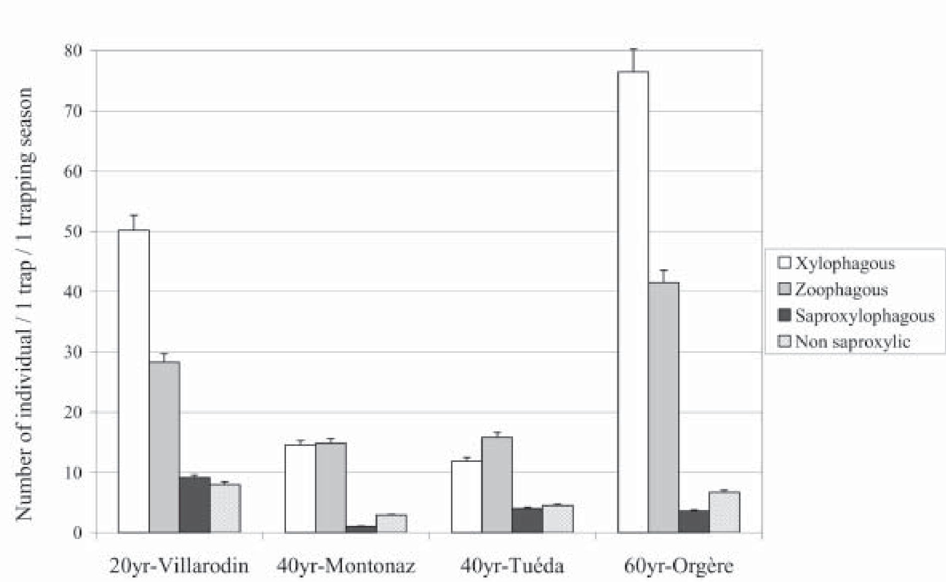

Figure 4. Numbers of individuals trapped (nb. Ind./trap/trapping period).

deadwood stocks. It created some gaps in the continuity of this material. Such gaps had an impact on the saproxylic populations. The sequence of events could be summarized as follow:

- 20 years after the management: The large amount of deadwood created by management and the diversity of trees favoured saproxylic beetles.

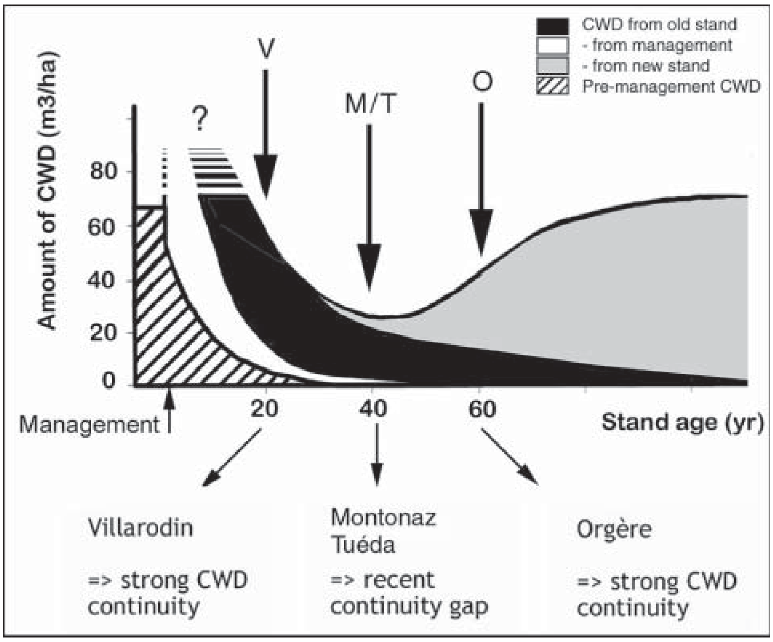

Figure 5. Positions of the forests on a theorical CWD evolution curve and estimations of the ecological continuity of CWD.

- After 40 years, the decay and the poor recruitment created a gap in the deadwood continuity and the populations of saproxylic beetles declined.
- After 60 years, the forest was older and created sufficient deadwood material in order to maintain an abundant saproxylic fauna.

Functional groups of saproxylic beetles could be good indicators of the status of deadwood stock in forest, both in quantity and quality of dead wood material. The assessment of deadwood could provide relevant information on the status of the saproxylic population in a stand. The use of some families of saproxylic insects could be proposed in order to evaluate some underlying characteristics of the deadwood material present in the forest and to assess the state of the forest heterotrophic phase.

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Annex: List of saproxylic families.

Families	Villarodin	Montonaz	Tuéda	Orgère
Alleculidae	-	-	1	1
Anobiidae	-	-	1	3
Anthicidae	-	-	-	3
Aphodiidae	7	1	4	3
Biphyllidae	1	-	-	-
Bostrychidae	-	-	1	-
Bruchidae	2	-	-	-
Buprestidae	-	1	1	-
Cantharidae	3	6	20	15
Carabidae	-	2	10	4
Catopidae	1	-	3	1
Cerambycidae	1	1	37	6
Chrysomelidae	-	1	1	1
Ciidae	3	1	5	-
Clambidae	17	-	7	3
Cleridae	1	3	5	4
Coccinelidae	-	1	-	2
Coleoptera sp.	-	2	15	5
Colydiidae	1	-	-	2
Cryptophagidae	3	-	-	2
Cucujidae	-	-	3	-
Curculionidae	16	25	8	68
Dascillidae	-	-	1	-
Elateridae	-	-	47	7
Histeridae	4	-	5	-
Hydraenidae	1	1	-	-
Hydrophilidae	-	-	1	
Lathrididae	7	-	13	7
Leiodidae	2	-	-	3
Lycidae	3	1	-	1
Melandryidae	1	-	2	1
Monotomidae	31	2	84	107
Mordellidae	1	-	-	1
Nitidulidae	16	9	62	81
Oedemeridae	2	-	-	-
Phalacridae	2	-	-	5
Pselaphidae	1	1	8	2
Ptiliidae	11	-	43	1
Ptinidae	1	3	-	8
Pythidae	1	-	-	-
Salpingidae	3	1	2	3
Scarabaeidae	1	-	-	5
Scolytidae	241	43	195	544
Scraptiidae	8	3	12	3
Silphidae	3	-	2	-
Silvanidae	-	-	-	1
Staphylinidae	80	63	152	119
Total	476	171	751	1022

Diversity of Bryophytes – Useful Indicators of Madeira Laurel Forest Conservation

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Abstract

The current fieldwork in the Madeiran Laurissilva concerns the study of the diversity of bryophytes occurring on slopes and its variation along three altitudinal levels (400–699 m, 700–999 m and 1000–1299 m). The first 41 sampling plots led to the identification of 127 different species (66 liverworts and 61 mosses), 9 of which are exclusive to Macaronesia and 3 to the Madeira archipelago. Liverworts with watersacs, ecomorphological adaptations for water storage, were frequently found at the altitudinal levels of 700–999 m and 1000–1299 m. Others, with a rill-like arrangement of leaves, adaptations for condensing water vapour from fog and mists, were particularly frequent between 1000–1299 m. The presence of *Plagiochila* species, together with the occurrence of endemic taxa, is a good indicator of the richness and importance of bryophytes diversity in each studied plot.

Keywords: Madeira: Laurissilva; Bryophytes; diversity indicators.

Introduction

Due to its insular characteristics, the Madeira archipelago shows an important floristic diversity. In fact, its laurel forest composition is similar to the one found in the Tertiary period, which covered the Southern Europe and Northern Africa millions of years ago.

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Monitoring and Indicators of Forest Biodiversity in Europe – From Ideas to Operationality
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Nowadays this subtropical forest has its largest extension located on Madeira Island, where it covers about 15 000 hectares. In 1999, Madeiran Laurissilva was declared a World Natural Heritage under the protection of UNESCO, due to the forest's richness, biological particularities and good preservation (Jardim and Fontinha 2000).

Bryophytes are a major part of ecosystems biodiversity, and as they lack a leaf cuticle they are capable of gaining and loosing water very quickly (Hallingbäck and Hodgetts 2000). In the Laurissilva, the bryophytes contribute to retain water from fogs and vertical precipitation, regulating the water flow and allowing the forest to gradually release water into water streams, preventing flash floods, erosion, and landslides (Fontinha 1997; Jardim and Fontinha 2000).

The orography of the Madeira Island, along with the strong influence of the trade winds, is the primary causes for the existence of a complex ecosystem structure. In fact, the Laurissilva forest hosts a rich and diverse bryoflora, including several endemic species to Madeira, as well as further species occurring only in Macaronesia (Sim-Sim et al. 2000; Fontinha et al. 2001).

The biodiversity study presented here is based on the sampling of the bryophytes vegetation occurring on the Laurissilva slopes. To determine the variation on bryophytes diversity with altitude, three altitudinal gradients were considered (400–699 m, 700–999 m and 1000–1299 m). It is also our goal in this study to gather adequate information on bryophyte biodiversity and point out indicator species, which may contribute in the future to the indication of the most relevant sites for the conservation of bryophyte diversity.

Material and methods

Site description

The study was performed in the Madeiran Island located on the Madeira Archipelago, in the Atlantic Ocean, between latitudes 32º 38' and 32º 52' N and longitudes 16º 39' and 17º 16' W. Aged of 5.2 million years and with a maximum altitude of 1861 meters, Madeira is an island with 736.7 km^2 and a very irregular topography, since 65% of its surface has an inclination over 25% (Sunyer 2000).

Most of Madeira Laurissilva is included in the 27.5% of the island territory that has been considered a Place of the Natura Network 2000, Laurissilva Site (PTMAD0001), under the Habitats Directive 92/43 ECC of the European Council, belonging to its chain of Biogenetic Reserve since 1992 (Sunyer 2000). This forest develops on the cloudy slopes of the island northern side, from 300 to 1300 meters in altitude (Sjögren 1975), being restricted to a few places of difficult access on the southern coast.

According to Jardim (2003), the distribution of Laurissilva is related with the mesotemperate and mesomediterranean belts on the island northern side, and with the mesomediterranean belt on the southern one.

The Laurissilva slopes being exposed to the NE predominating winds and with a rainfall greater than 3000 mm per year have a high humidity level nearly all year long (75–90%) (Fontinha et al. 2001; Sunyer 2000). The average temperature on the south coast of Madeira is 18ºC (Jardim and Francisco 2000), decreasing by an average of about 1ºC per 150 meters in altitude (Sjögren 1975).

The vascular flora of Madeira is exuberant and diverse and the present number of vascular plants endemic to Madeira is 165 (Jardim and Francisco 2000). The studied Laurissilva is essentially described by the *Clethro arboreae-Ocoteetum foetentis* association, and its most characteristic tree is *Ocotea foentens* (Ait.) Benth. & Hook. together with the endemic *Clethra arborea* Ait., *Pittosporum coriaceum* Dryand. ex Ait., *Ilex perado* Ait. ssp. *perado* and *Vaccinium padifolium* J. E. Sm. ex Rees. Associated to these other vascular species also

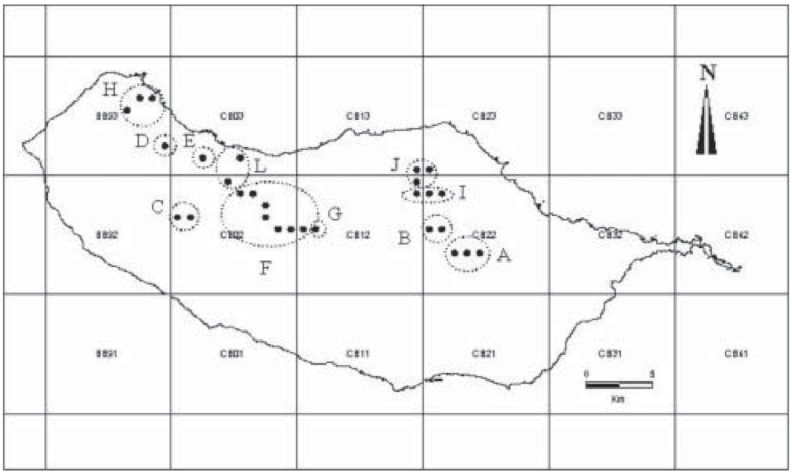

Figure 1. Map of Madeira Island exhibiting the Laurissilva sampled areas (A-L), and the UTM 1 x 1 km studied (● = the center of an UTM 1 × 1 km square).

Table 1. Location name of the studied areas.

Area	Location name
A	Santana, Balcões
B	Santana, Fajã da Nogueira, Lev. João de Deus
C	Calheta, Rabaçal, Levada do Risco
D	Porto Moniz, Fanal, Lagoa do Fanal
E	Porto Moniz, Seixal, Ribeira do Seixal
F	São Vicente, Levada do Folhadal
G	S. Vicente, Encumeada, Chão da Relva
H	Porto Moniz, Ribeira da Janela
I	Santana, Queimadas, Caldeirão Verde
J	Santana, Ribeiro Bonito
L	S. Vicente, Montado do Pessegueiros

occur, such as *Laurus azorica* (Seub.) Franco, *Picconia excelsa* (Aiton) DC. and *Erica scoparia* L. spp. *maderincola* D. C. McClint. (Capelo et al. 1999; Jardim 2003).

Vegetation sampling and data analysis

The fieldwork was performed between August 2001 and July 2003 in 11 distinct forest areas, corresponding to 28 UTM squares, 1×1 km (Figure 1, Table 1). A total of forty-one plots were selected to represent the main bryophyte vegetation occurring on the slopes. The slopes were either of natural or anthropomorphic origin, the latter located along the "levadas" (man-made water channels). The studied areas show an altitudinal range of 400 to 1299 m, and were distributed by three distinct altitudinal levels (400–699 m, 700–999 m and 1000–1299 m) in order to determine the altitudinal effect on bryophytes diversity.

One inventory was made for each sampling plot (2500 cm^2) and the quantitative occurrence of each bryophyte species was estimated in absolute value (%), using a rigid frame. For each sampled plot, several environmental factors such as position (UTM-squares 1 × 1 km), altitude, micro- and macro-aspect, topography and cover of the different vegetation layers

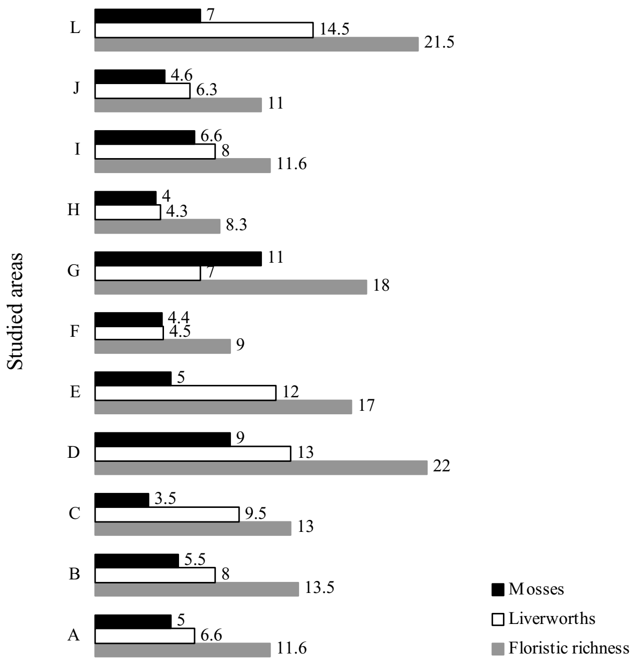

Figure 2. Representation of the floristic richness, liverworts and mosses, values per UTM 1 × 1 km in each studied area.

and substrate features were considered, in order to allow a future statistical approach of all the data, also including climatic variables.

The nomenclature for the acrocarpous mosses followed Corley et al. (1981) and Corley and Crundwell (1991); for the pleurocarpous mosses, Hedenäs (1992); and for liverworts, Grolle and Long (2000), Heinrichs et al. (1998), Heinrichs et al. (2002), Rycroft et al. (2001) and Rycroft et al. (2002).

Results

A synthesis of the floristic composition of each studied plot (a total of 41 plots) is presented in the appendices. Although the presented results correspond to specific habitats, relatively high bryophyte diversity was recorded. One hundred and twenty seven bryophytes (66 liverworts and 61 mosses) were identified. From those, 9 species are exclusive to Macaronesia and 3 to the Madeira archipelago.

The evaluation of the bryophyte diversity in the 11 studied areas reveals that those exhibiting the highest values of floristic richness are D, L, G, E and B. However, and despite the differences in the floristic richness of the different areas, it is clear that liverworts represent, in most of the cases, at least 50% of the total biodiversity sampled (Figure 2).

In Figure 3, we observe that the occurrence of endemic taxa and that of *Plagiochila* species are, in general, related to the total biodiversity surveyed in each area. The areas exhibiting greater floristic richness, D, L, G, E and B also show a considerable proportion of endemic and *Plagiochila* taxa.

In accordance to the liverworts diversity presented in the annex, it was possible to select the species, which exhibit important ecomorphological water storing and conducting adaptations

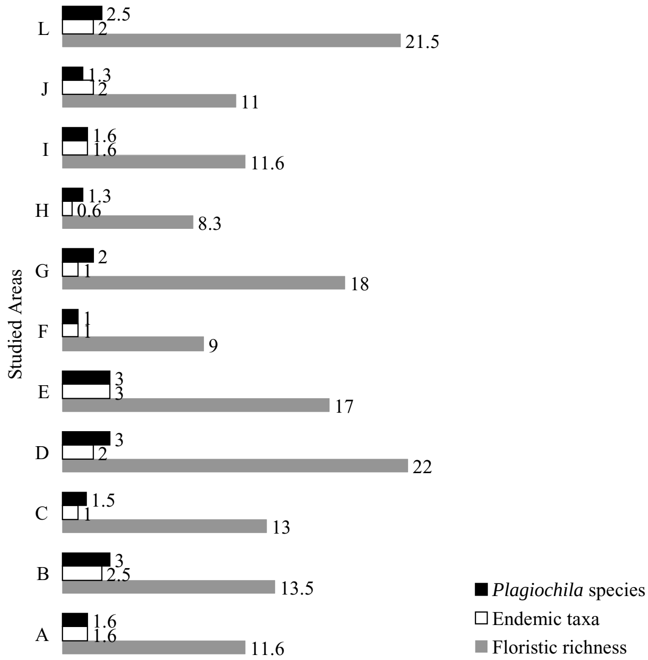

Figure 3. Bryophytes diversity in the studied areas***:*** floristic richness, endemic taxa and *Plagiochila* species, values per UTM 1 × 1 km in each studied area.

(Table 2). Watersacs and rills are essential for the maintenance of physiological processes under distinct hydric conditions.

For the 11 studied areas, all the plots were compared along an altitudinal gradient related with the different bioclimatic belts defined for the Madeiran Laurissilva. In Figure 4 it is possible to observe the distribution of liverwort biodiversity, according to the three-levels studied. The results obtained indicate that the highest biodiversity values were observed between 700–999 m and 1000–1299 m.

Discussion and conclusions

Biodiversity

As mentioned above, during this study we observed a rich bryophyte flora in the 11 sampling areas of Madeiran Laurissilva, which permitted a biodiversity analysis. In most of the cases, liverworts represent a significant proportion of the total biodiversity sampled (Figure 2). This is due to the liverworts ability to grow on substrates exposed to high humidity levels, originated by rainfall, fogs and mists, an environmental characteristic of the Laurissilva forest.

The differences in species diversity observed between areas (Figure 2), depend greatly upon the type of the sampled slopes (natural or antropomorphic). In fact, the areas exhibiting the highest floristic richness, D, L, G, E and B also comprise a considerable number of endemic and *Plagiochila* taxa, and correspond to plots on natural slopes. On the contrary, plots with lower floristic richness were recorded on the slopes of anthropomorphic origin. These are mainly located along the “levadas” (H, F, I, A, and C), and have fewer endemic and *Plagiochila* species. In general, there is a correlation between the occurrence of *Plagiochila*

Table 2. List of the liverworts species selected according to their ecomorphological water adaptations.

Liverworts	Watersacs	Rills
Aphanolejeunea microscopica (Taylor) A. Evans	+	
Diplophyllum albicans (L.) Dumort.		+
Drepanolejeunea hamatifolia (Hook.) Schiffn.	+	
Frullania polysticta Lindenb.	+	
Frullania tamarisci (L.) Dumort. var. *tamarisci*	+	
Frullania tamarisci (L.) Dumort. var. *sardoa* (De Not.) De Not	+	
Frullania tamarisci (L.) Dumort. var. *schiffneri* Nichol.	+	
Lejeunea cavifolia (Ehrh.) Lindb.	+	
Lejeunea eckloniana Lindenb.	+	
Marsupella emarginata (Ehrh.) Dumort.		+
Marsupella profunda Lindb.		+
Microlejeunea ulicina (Taylor) A. Evans	+	
Plagiochila bifaria (Sw.) Lindenb.		+
Plagiochila exigua (Taylor) Taylor		+
Plagiochila punctata (Taylor) Taylor		+
Plagiochila retrorsa Gottsche		+
Plagiochila spinulosa (Dicks.) Dumort		+
Plagiochila stricta Lindenb.		+
Porella canariensis (F. Weber) Underw.		+
Porella inaequalis (Gottsche ex Steph.) Perss.		+
Porella obtusata (Taylor) Trevis		+
Radula aquilegia (Hook. f. & Taylor) Gottsche et al.	+	
Radula carringtonii J. B. Jack	+	
Radula nudicaulis Steph.	+	
Radula wichurae Steph.	+	
Scapania compacta (A. Roth) Dumort.		+
Scapania curta (Mart.) Dumort.		+
Scapania gracilis Lindb.		+
Scapania nemorea (L.) Grolle		+
Scapania undulata (L.) Dumort.		+

species and endemic taxa, indicating that the *Plagiochila* species referred to Madeira (Heinrichs et al. 1998, Heinrichs et al. 2002, Rycroft et al. 2001, Rycroft et al. 2002, Sim-Sim et al. 2003 and Söderström et al. 2002) can be useful indicators of bryophytes diversity.

Mosses and liverworts have developed important water-control mechanisms. The formation of water lobules and water sacs in some liverworts such as *Frullaniaceae* and *Radulaceae*, are seen as important adaptations to delay water-loss, ensuring permanent water supply in relatively “dry”, periodically xeric habitats. Effective water-storing structures are the upward folded leaves (rill-like arrangement) forming a channel system found in various liverworts (Table 2). Typical taxa with these adaptations are some *Plagiochila* species. Such a leaf arrangement functions as an effective capillarity system between the leaves and stems, or between different leaves, to conduct water (Kürschner et al. 1999). This was also observed in the laurel-forests of the Canary Islands, where taxa with these structures are concentrated in communities strongly exposed to daily mists and fogs (Zippel 1998).

The altitudinal gradient of 400–699 m refers mainly to the Madeiran mesomediterranean bioclimatic belt, where the liverworts biodiversity sampled was the lowest, and the occurrence of liverworts with watersacs and rill-arrangement of leaves was similar (Figure 4).

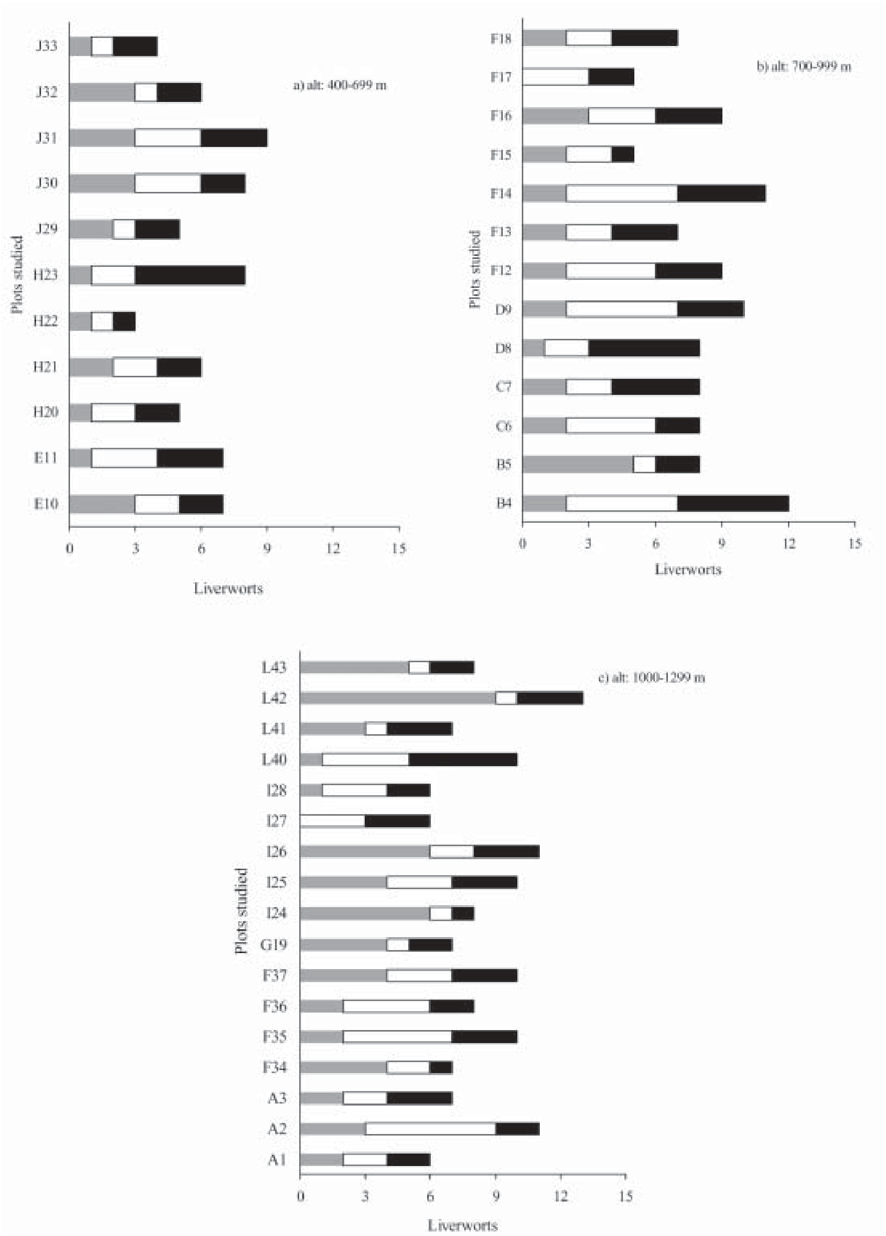

Figure 4. Number of liverworts species exhibiting ecomorphological water adaptations, according to the altitudinal gradient studied: a) 400–699 m, b) 700–999 m, c) 1000–1299 m; watersacs, rills and other. (Each studied site is represented by a letter followed by the plot's number).

In the mesotemperate bioclimatic belt, represented by the altitudinal gradients of 700–999 m and 1000–1299 m, the liverworts biodiversity is higher. Although liverworts with watersacs are also well represented, it is between 1000 and 1299 m that those with rills, like *Plagiochila*, are more frequent. The high environmental humidity levels at these altitudes explain the presence of liverworts with smaller water-sacs, such as *Harpalejeunea* sp., some *Lejeunea* sp., and *Microlejeunea* sp..

Useful indicators for biodiversity conservation

The frequent recording of *Plagiochila* species during this study confirms the abundance of this genus in Madeira, where 50% of the total species referred to Europe occur (Sim-Sim et al. 2003). *Plagiochila* is one of the largest liverwort genus, with about 450 species (Heinrichs 2002). It often exhibits an intercontinental range of distribution, so that species from Atlantic Europe are usually closely related with the neotropical taxa, whereas those widespread in Europe are related to the Asiatic ones, and not to those in the neotropics (Groth et al. 2003). According to Dierßen (2001), the *Plagiochila* species occurring in Madeira exhibit distinct phytogeographic affinities, namely with America, Europe and Asia. The results of this study indicate an affinity between the occurrence of *Plagiochila* species and that of endemic taxa, in the areas with important floristic richness (Figure 2).

In the future, the presence of *Plagiochila* in Madeiran Laurissilva, as a potential indicator of floristic richness, endemic taxa, and species with similar geographic patterns, shall be a useful tool for the recognition of the most relevant Laurissilva sites for the conservation of bryophyte diversity, and also a good tool to infer the total biodiversity in a forest.

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List of the bryophyte species collected during this study, * and ** indicate Macaronesia and Madeira endemics, respectively.

Liverworts and hornworts

Acanthocoleus aberrans
Acrobolbus wilsonii
Aneura pinguis (L.)
Aphanolejeunea microscopica
Asterella africana
Calypogeia arguta.
Calypogeia fissa
Cephalozia lunulifolia
Cephaloziella turneri
Chiloscyphus polyanthos
Conocephalum conicum
Diplophyllum albicans.
Drepanolejeunea hamatifolia.
Dumortiera hirsuta
Fossombronia angulosa
Frullania polysticta *
Frullania tamarisci var. *sardoa*
Frullania tamarisci var. *schiffneri*
Frullania tamarisci.
Frullania teneriffae
Harpalejeunea molleri
Jubula hutchinsiae var. *hutchinsiae*
Jubula hutchinsiae var. *integrifolia* *
Jungermannia aff. *leiantha*
Jungermannia gracillima
Lejeunea cavifolia
Lejeunea eckloniana
Lejeunea flava subsp. *moorei*.
Lejeunea hibernica
Lejeunea lamacerina
Lejeunea patens.
Lophocolea bidentata
Lophocolea fragrans
Lophocolea heterophylla
Marchantia polymorpha subsp. *ruderalis*
Marchesinia mackaii
Marsupella emarginata
Marsupella profunda
Metzgeria conjugata
Metzgeria furcata.
Microlejeunea ulicina
Nardia scalaris
Phaeoceros laevis.
Plagiochila bifaria
Plagiochila exigua
Plagiochila punctata
Plagiochila retrorsa
Plagiochila spinulosa.
Plagiochila stricta
Porella canariensis.
Porella inaequalis **
Porella obtusata
Radula aquilegia
Radula carringtonii
Radula lindenbergiana
Radula nudicaulis
Radula wichurae *
Riccardia multifida
Saccogyna viticulosa.
Scapania compacta
Scapania curta
Scapania gracilis
Scapania nemorea
Scapania undulata
Telaranea nematodes
Tylimanthus madeirensis **

Mosses

Amphidium curvipes *
Andoa berthelotiana *
Anoectangium aestivum
Atrichum angustatum var. *muelleri*
Atrichum undulatum
Brachythecium percurrens **
Brachythecium plumosum
Brachythecium rutabulum var.*atlanticum*
Brachythecium velutinum
Bryum caespiticium
Bryum capillare
Bryum rufifolium
Campylopus flexuosus
Campylopus fragilis
Campylopus pyriformis
Cyclodictyon laetevirens
Dicranella heteromalla
Dicranum canariense
Diphyscium foliosum
Echinodium prolixum *
Echinodium spinosum *
Epipterygium tozeri
Eurhynchium hians
Eurhynchium praelongum.
Fissidens asplenioides
Fissidens luisierii *
Fissidens polyphyllus
Fissidens rivularis
Fissidens taxifolius
Homalia lusitanica
Hookeria lucens
Hypnum cupressiforme
Hypnum uncinulatum
Lepidopilum virens
Leucobryum juniperoideum
Leucodon sciuroides
Neckera intermedia
Philonotis marchica
Philonotis rigida
Plagiomnium affine

Mosses (continued)

Plagiomnium medium
Plagiomnium undulatum var. *madeirense* *
Plagiothecium nemorale
Pogonatum nanum
Pohlia elongata
Polytrichum formosum
Polytrichum juniperinum
Polytrichum piliferum
Ptychomitrium nigrescens
Ptychomitrium polyphyllum.
Rhizomnium punctatum
Rhynchostegium murale
Rhytidiadelphus loreus
Scleropodium purum
Scleropodium touretii
Thamnobryum maderense
Thuidium tamariscinum
Tortella flavovirens
Trichostomum brachydontium
Trichostomum brachydontium cuspidatum
Ulota crispa

On the Interest of Litter-Dwelling Invertebrates to Assess Silvicultural Impact on Forest Biodiversity

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Abstract

In the actual context of urgent need of biodiversity indicators to assess forest ecosystems, the use of insects and more generally invertebrates remains infrequent whilst they constitute the major part of biodiversity.

Our paper, focusing on litter-dwelling invertebrates (*Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*) at the stand level, come within the framework of the validation of indicators of forest biodiversity. So our research is attempting to partly fill in the gaps in our knowledge about correlations between indicators (e.g. coarse woody debris, main tree composition, stand structure and stand age) and litter-dwelling invertebrates species richness. The correlations between the groups under investigation and with plant richness are then considered. These correlations appear to be low so the reliability of one taxonomic group as indicator should always be cautiously tested.

Taking into account bioindicator criteria, the indicator ability of the four group studied is discussed. *Staphylinidae* seem to constitute the best indicators of forest management and naturalness in landscapes with long human disturbance history.

Keywords: Carabidae; Staphylinidae; Diplopoda; Chilopoda; silviculture; biodiversity indicator.

1. Introduction

In the context of the current biodiversity crisis and the subsequent development of political, legal and management instruments in favour of biodiversity, we urgently need reliable

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indicators of biodiversity. Such indicators play a major role in monitoring biodiversity, identifying, targeting and helping prioritise the types of action to be taken in forest ecosystems and in measuring and assessing the results of the implementation of such actions (Noss 1990; Larsson and Esteban 2000; Watt 2003).

In a long-term perspective, it is necessary for a standardised system of measurable attributes or indicators of forest biodiversity to be implemented in a global assessment of state and trends in biological diversity (Noss 1990; Larsson 2001).

But conceptual problems arise, biodiversity have various meanings for different people. The ways in which biodiversity is considered are numerous and depend frequently on the speciality of each scientist (Noss 1990; Rainio and Niemelä 2003).

A contradiction occurs concerning bioindicators. While insects and more generally invertebrates constitute the major part of the biodiversity and particularly in forest ecosystems (Figure 1), they are quite rarely used as indicators. This statement is easily explained by the concern for saving money, simplicity and rapidity. This is why forest structure and plants (taxonomically well described and easily identified in the field) are frequently used to estimate forest biodiversity. However, little is known about the correlations between plant richness and invertebrate richness, moreover there is some evidence that flowering plant diversity does not constitute an accurate indicator of the species richness of invertebrates (Oliver and Beattie 1993; Larsson and Esteban 2000).

The quality and reliability of the chosen indicators should be scientifically tested by firstly defining without fog the objectives and the corresponding biodiversity aspects or entities. In a time-consuming second step, a number of habitats have to be sampled as thoroughly as possible with regard to the aims defined. The third step consists in testing the correlations between our choice of easily measurable indicators and the habitat samplings done in the second step. Most studies claiming to measure or indicate biodiversity assume that the group of organisms they investigate is somehow representative of biodiversity. However, in only very few cases has the correlation between a group or several groups of species with a more or less representative sample of all organisms been measured and published (Duelli and Obrist 2003). So although there is a wealth of indicators to choose from, most have been poorly tested and require rigorous validation in order to be used and interpreted with confidence (Noss 1999; Larsson 2001). Our research is trying to partly fill in these gaps. Besides bioindicator criteria, sensitivity to forest management and correlations between the groups studied and with vegetation species richness brought valuable information about the bioindicator power of litter-dwelling invertebrates (*Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*).

2. Methods

2.1 Study region, sampling design and selection of plots

The study (du Bus de Warnaffe 2002) took place in the natural region of the Belgian Ardennes (about 5000 km²), situated between the cities of Namur and Luxembourg. The Ardennes are mostly composed of pastures and woodlands, partially transformed into commercial conifer stands during the last 150 years (Devillez and Delhaise 1991). This region is characterised by a humid sub-mountane climate, a hilly relief and loamy acidic soils.

The plots were chosen in order to minimise the variation of climate and soil, and according to three categorical management variables: the *structure* of the forest, the *composition* of the canopy and the *stage* reached in the silvicultural cycle. The altitude ranged from 320 to 600 m, the mean annual rainfall from 830 to 1170 mm yr^{-1} and the mean annual temperatures from 6.7

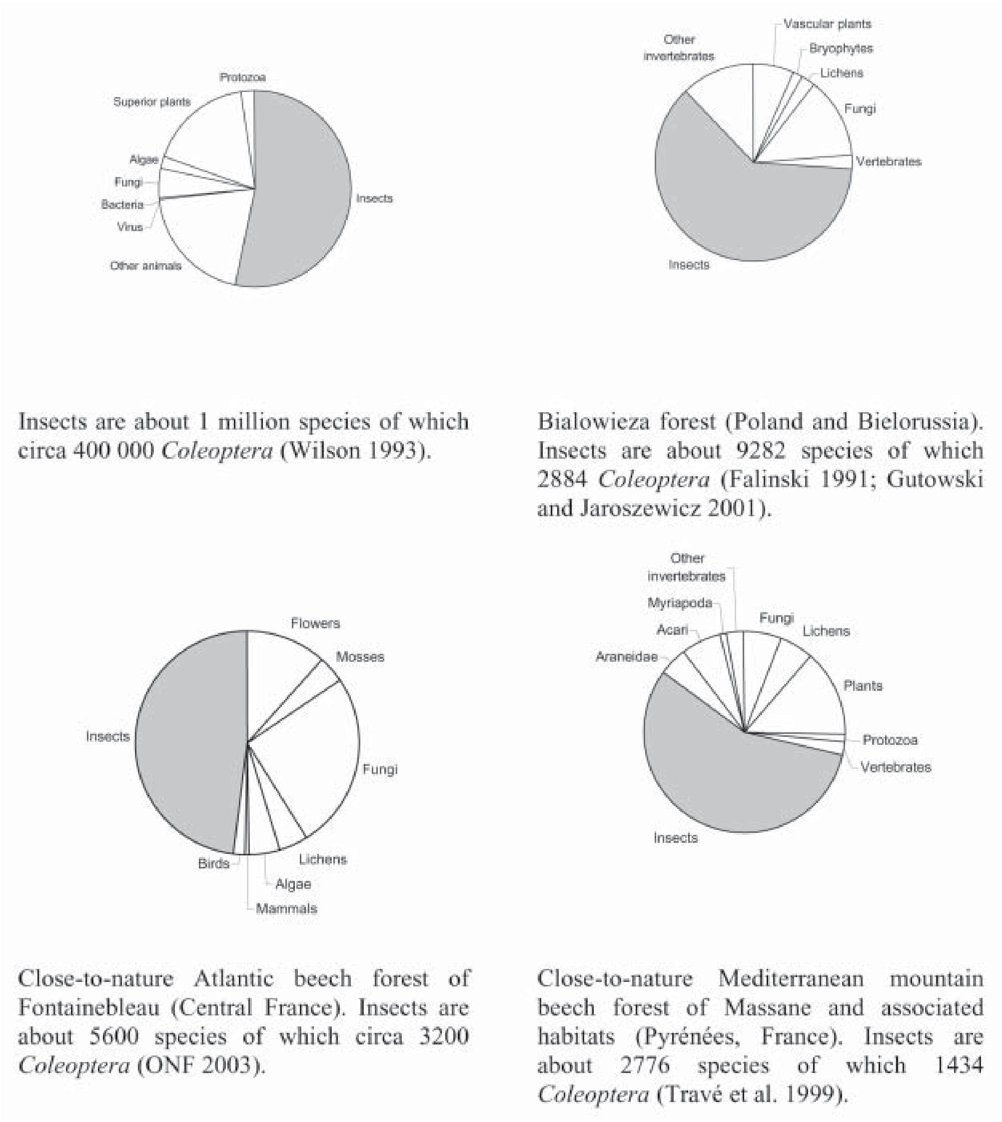

Figure 1. Proportion of known species richness in the main living groups.

to 9.2 °C (Weissen et al. 1994). All the study plots were on flat or very slightly sloping ground of acidic brown and moderately dry soils (*Dystric cambisol*: FAO 1990), which were very similar in terms of water and nutrient availability. All stands were forests for at least 150 years. Plots were all situated in large forests and at least 100 m from the nearest field or meadow.

The *structure*, as determined by the mean size of patches (homogeneous stand) in the forest, was evaluated by Geographical Information System (G.I.S. Star-Carto) on aerial photographs. Three classes were distinguished: even-aged (E): final tree exploitation by large clear-cuttings (> 2 ha), group (G): final tree exploitation by medium-sized clear-cuttings (0.2–0.5 ha), and uneven-aged (U): final tree exploitation by small clear-cuttings (< 0.2 ha).

The *composition* classes were defined by the local cover of the tree species (0.04 ha): more than 90% of the area covered by beech (B; *Fagus sylvatica* L.), by oaks (O; *Quercus petraea*

Table 1. Plot number per habitat type which are combinations of the structure (first initial), composition (second initial) and stage reached in the silvicultural cycle (see 2. Methods). Except the close-to-nature stands in brackets (r), all the plots are intensively managed.

Structure - composition	EC	EB	EO	GM	UC	UB	Total
Stage 1	8	6	6	6	5	10	41
Stage 2	8	6	22 (r =16)	6	7	10	59
Stage 3	8	6	8	6	6	24 (r =14)	56
Stage 4	-	4 (r =4)	-	-	-	-	4
Total number of plots	24	22	36	18	18	44	162

Table 2. Total number of species and individuals trapped in the 162 plots.

	Carabidae	Staphylinidae	Diplopoda	Chilopoda
Number of species	80	136	19	25
Number of individuals	43 658	13 608	4334	3293

(Mattme.) Liebl. and *Quercus robur* L.), by spruce and Douglas fir (C; *Picea abies* (L.) Karst and *Pseudotsuga menziesii* (Mirb.) Franco) or by a mixing of the previous species (M).

Three *stages* were defined for each combination of structure and composition: regeneration (stage 1: trees aged 3–10 years); medium-aged stand (stage 2: trees aged 20–40 years for conifer stands and 30–60 years for beech and oak stands), mature stand (stage 3: trees aged 50–80 years for conifer stands and 80–140 years for beech and oak stands) and overmature stand (stage 4: trees aged > 150 years for beech stands).

We also distinguished between intensively managed stands (m) and close-to-nature stands (r; reference stands: indigenous tree species, native provenance, natural regeneration, dead wood abundance and forest continuity in time).

The 21 classes defined by combining structure, composition and stage were called *habitat types*. In all, 162 plots were selected. The number of plots by class is given in Table 1.

2.2 Species data

We used 8.5 cm diameter × 17 cm deep pitfall traps with 5% formaldehyde (over-saturated salty solution was used in close-to-nature stands) to collect ground-dwelling arthropods (Dufrêne 1988). In each plot, 3 pitfalls were placed in a triangle of 3 m base (Desender et al. 1999), and emptied monthly (Heliölä et al. 2001). We sampled the traps for seven months (Benest and Cancela da Fonseca 1980; Düldge 1994), from 10 April to 5 November 1999 (384 traps in intensively managed stands) and from 1 April to 31 October 2002 (84 traps in close-to-nature stands). Pitfall traps are known to gather valuable information on activity and relative abundance of various groups of ground-dwelling arthropods (e.g. Dufrêne 1988; Branquart et al. 1995; Rainio and Niemelä 2003). Among all the groups captured, only the most representative and relevant taxa were sorted, identified and analysed: *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda* representing an important part of the litter-dwelling invertebrates. The classification of the 162 sample plots in the 21 habitat types allowed us to compute mean species richness and cumulative number of species trapped for each habitat type.

Table 3. Main results for the *Carabidae*, *Staphylinidae* and *Diplopoda* for each habitat type (n = plot number; N = mean individual number; S = mean species richness and Stot = total number of species trapped per habitat type).

			Carabidae			*Staphylinidae*			*Diplopoda*		
Habitat type		n	N	S	Stot	N	S	Stot	N	S	Stot
Beech forest	EB-1m	6	303.0	20.5	43	137.8	11.0	32	30.0	5.8	11
	EB-2m	6	250.5	8.5	19	36.0	7.3	19	7.7	2.2	7
	EB-3m	6	225.0	8.3	18	30.8	6.7	20	14.5	2.3	6
	EB-4r	4	87.8	9.0	16	70.0	19.0	42	4.0	1.0	1
	UB-1m	10	272.0	13.8	36	33.8	10.7	35	24.6	4.2	9
	UB-2m	10	354.4	8.0	23	65.7	9.0	25	21.9	3.6	8
	UB-3m	10	355.3	8.6	22	50.1	8.2	26	11.0	3.0	7
	UB-3r	14	240.9	11.2	32	56.5	12.0	64	74.1	5.5	13
Mixed forest	GM-1m	6	104.0	15.3	36	52.5	12.2	33	25.8	4.5	9
	GM-2m	6	148.5	9.3	23	65.0	10.3	29	28.2	5.2	11
	GM-3m	6	223.0	8.7	19	53.7	9.3	23	17.2	3.7	8
Coniferous forest	EC-1m	8	220.0	17.4	40	109.6	14.6	42	57.9	4.5	12
	EC-2m	8	278.5	9.5	23	72.3	9.3	29	14.3	4.0	10
	EC-3m	8	333.6	9.5	18	60.3	8.1	23	5.9	3.1	8
	UC-1m	5	63.4	8.6	20	40.6	11.0	27	44.2	5.4	10
	UC-2m	7	227.6	7.9	16	207.7	9.0	27	25.0	3.9	10
	UC-3m	6	276.5	7.7	17	49.8	9.5	28	10.7	3.8	7
Oak forest	EO-1m	6	147.8	14.2	33	49.5	8.8	32	29.8	3.5	7
	EO-2m	6	289.3	11.3	23	202.5	6.3	14	49.2	5.2	8
	EO-2r	16	393.3	14.4	47	77.8	15.7	68	26.1	4.4	13
	EO-3m	8	431.4	12.9	29	267.0	4.9	21	41.9	4.6	11

EB = even-aged beech stand
UB = uneven-aged beech stand
EO = even-aged oak stand
GM = group mixed stand
EC = even-aged coniferous stand
UC = uneven-aged coniferous stand

1 = regeneration stand
2 = growing stand
3 = mature stand
4 = overmature stand
r = close-to-nature stand
m = intensively managed stand

Vascular plant, bush and tree species richness was obtained for the 162 plots by sampling vegetation on 0.04 ha according to Braun-Blanquet method (Braun-Blanquet 1951).

Finally using the SAS package, we computed the Spearman rank correlations between species richness of vascular plants, bushes, trees, *Carabidae, Staphylinidae, Chilopoda* and *Diplopoda* (SAS 2000).

3. Results

The paired comparisons (Table 3) between the same habitat type show that close-to-nature forests shelter a higher species richness for the *Coleoptera Carabidae* (except for Stot in EB) and *Staphylinidae* in comparison with intensively managed forests. The results are less clear for *Diplopoda* whose species richness is quite poor. The *Chilopoda* results not presented here gave similar statistics to *Diplopoda*.

Whilst the total number of species trapped are quite similar between *Carabidae* and *Staphylinidae*, the difference between close-to-nature and intensively managed forests is

Table 4. Spearman rank correlations between species richness of vascular plants, bushes, trees, *Carabidae* (Car), *Staphylinidae* (Staph), *Chilopoda* (Chilo) and *Diplopoda* (Diplo) (*: 0.05<p<0.01; **: 0.01<p<0.001; ***: 0.001<p<0.0001 ; ****: p<0.0001).

	Plant	Bush	Tree	Car	Staph	Chilo	Diplo
Plant	-	0.50****	0.04	0.26***	0.28***	-0.14	0.06
Bush		-	-0.07	0.05	0.003	0.06	0.21***
Tree			-	-0.25**	0.02	0.05	0.12
Car				-	0.26***	-0.19*	0.17*
Staph					-	-0.37****	-0.06
Chilo						-	0.15
Diplo							-

much higher for *Staphylinidae* than for *Carabidae*. This statement is supported by the fact that the trappings in close-to-nature stands brought nine *Staphylinidae* species new for Belgium.

Examining now the correlations between the species richness of the groups under investigation (Table 4), we observe that these relations are in general low and can be positive, absent or even negative. This is also true concerning the correlations between the vegetation richness (vascular plants, bushes or trees) and the invertebrate richness. Yet, the species richness of *Staphylinidae* and *Carabidae* are slightly correlated (0.26, p=0.0008) and the species richness of *Staphylinidae* and *Chilopoda* are negatively correlated (-0.37, p<0.0001). The correlations between the groups under investigation and vegetation are low. *Carabidae* (0.26, p=0.0006) and *Staphylinidae* (0.28, p=0.0003) present significant correlation with vascular plants, *Diplopoda* with bushes (0.21, p=0.0062) and *Carabidae* are negatively correlated with trees (-0.25, p=0.0014). These results are more or less similar to the figures obtained by Saetersdal et al. (2003).

4. Discussion

According to our results comparing close-to-nature stands to intensively managed stands for the same habitat types, it appears that forest management has a negative impact on the species richness of litter-dwelling invertebrates.

The poor *Diplopoda* and *Chilopoda* species richness observed is mainly due to acidic soil conditions, altitude and harsh climate, at least above 500 m (Kime pers. communication). This statement limits the value of our results for analysing their indicative power.

Carabidae communities have been seriously altered in Belgian forests (du Bus de Warnaffe and Lebrun in press). Large brachypteran forest stenotopic species have disappeared or have seen their population decreasing. Strict forest species such as *Carabus intricatus* or *C. glabratus* have disappeared in Belgian forests and *Abax carinatus* has seen its population diminishing drastically. In fact, only the species well adapted to silvicultural perturbations could still be present in Ardennes forests, intensively managed for centuries. Due to this situation, we have to use cautiously naturalness differences between our Belgian forests with this group. The *Carabidae* species richness, higher in regeneration stage (Table 3), is due to the colonisation of large clear-cuttings by open-field species (du Bus de Warnaffe and Lebrun in press). So it appears clearly that the indication given by the species richness is insufficient.

It is always important to examine the list of species (red-listed, rare and endangered species, stenotopic species, etc.).

Of the four groups, *Staphylinidae* constitutes the best indicator of the main forest management types but also of forest naturalness. This family is very common in natural, semi-natural and managed forest ecosystems (Bohac 1999).

While the majority of species are known as non-specific predators, a significant proportion of species possess narrow ecological requirements of prime importance to assess the naturalness of forest ecosystems (i.e. *mycetophagous*, *fungicolous*, *myrmecophilous* or *saproxylophagous*, etc.). Unfortunately this family still remains insufficiently known (Koch 1989; Bohac 1999). In any case, *Staphylinidae* constitute a very promising group not only as bioindicators but also for future ecological studies.

As stated by Larsson (2001), there is a need for reference values and critical thresholds for forest biodiversity. A reconstruction of reference situations of forest biodiversity (e.g. near-native state, pre-industrial state) expressed through indicators values would be of practical use for setting biodiversity targets (Larsson 2001). We are trying to approach this reference i.e. the state of *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda* in the natural undisturbed community in order to possibly place our stands on a naturalness scale and to distinguish the actual “anthropic biodiversity” versus “natural biodiversity” under low human influence.

We intend to realise this approach in a near future by combining data:

- from natural or near-natural European forest ecosystems in, as far as possible, similar ecological conditions;
- on “strict forest species” and more precisely on “forest naturalness indicators”.

Although the indicator species approach is opened to criticism (questionable assumptions, methodological difficulties, biased applications, etc.), species are often more tangible and easy to study than communities, landscapes or genes (Noss 1990). Noss (1999) proposed seven kinds of species that might make good targets for monitoring supposing that these most sensitive focal species (or group of species) would presumably constitute umbrella species (area-limited, dispersal-limited, resource-limited, process-limited, keystone, narrow endemic and finally special cases which gather species important in the forest ecoregion that do not fall within one of the previous categories).

The bioindicators, rigorously tested, must be measurable and repeatable surrogates for biodiversity which should ideally be:

1. well-known in its taxonomy and biology;
2. easy and cost-effective to measure, collect, identify and/or calculate;
3. widely applicable and then distributed over a broad geographical area
4. distributed over a large range of habitats with species that are pledged to specialised habitat in order to be sufficiently sensitive to provide an early warning of change;
5. capable of providing a continuous assessment over a wide range of stress and also able to differentiate between natural cycles or trends and those induced by anthropogenic stress (e.g. Noss 1990; Dajoz 1996; Rainio and Niemelä 2003).

Because no single indicator will possess all of these desirable properties, a set of complementary indicators is required (Noss 1990; Larsson and Esteban 2000). Table 5 examines the characteristics of the litter-dwelling groups under investigation in regard to the indicator desirable properties given above.

Whilst, according to Table 5, *Carabidae* seem to gather the best results concerning the bioindicator criteria, *Staphylinidae* appear to be more sensitive and then to bring more valuable information in landscapes with long human disturbance history.

Table 5. Indicator properties of the groups under investigation (Car: *Carabidae*; Staph: *Staphylinidae*; Diplo: *Diplopoda*; Chilo: *Chilopoda*).

Desirable properties	Bioindicator criteria	Car	Staph	Diplo	Chilo
1	Identification key	++	+/-	++	++
1	Checklists	++	+/-	++	++
1	Experts	++	+	+	+
1	Ecological information	+++	+/-	+	+
1	Chorological data	++	+/-	+	+
2	Easy to identify	+	-	+	+/-
2	Easy to sample (standardised)	+	-	+	+
2	Cost-effective	+/-	-	+/-	+/-
3	Distribution	broad	broad	broad	broad
3	Widely applicable	+++	+++	+++	+++
4	Habitat range	++	+++	++	++
4	Specialised species	++	+++	+	+
4	Sensitivity	++	+++	++	++
4	Forest naturalness indicators	+	++	-	-
4	Indicators of forest continuity	+/-	+	-	-
5	Continuous assessment	+++	+++	+++	+++
5	Natural >< anthropogenic	+	+++	+/-	+/-

Besides species or species group, biodiversity indicators can be habitat variables (structural and/or functional variables such as diameter and age class distributions, landscape pattern using remote sensing, etc.) (Noss 1999; Saetersdal et al. 2003). In many landscapes however, human land-use indicators (both structural and functional: e.g. deforestation rate, road density, fragmentation or edge index, etc.) may be the most critical variables for tracking the status of biodiversity (Noss 1990). Such habitat variables assumed or ideally proved to be important to the species (habitat suitability indicators) can usefully complete species richness and sometimes even replace the daunting samplings necessary to estimate this richness. Dead wood is a structural element critical to forest biodiversity and in particular to saproxylophagous species which represent one of the most important compartment of biodiversity in forests (e.g. Speight 1989; Noss 1990; Samuelsson 1994; Martikainen 2000). So the assessment of dead wood cannot be ignored in any forest biodiversity evaluation. Unfortunately, the presence of suitable habitat is no guarantee that the species of interest are present. Thus monitoring both habitat variables and species seem to be essential in most cases (Noss 1990; Saetersdal et al. 2003).

5. Conclusions and prospects

Two major results emerge from our study. First, of the four litter-dwelling groups investigated, *Staphylinidae* seem to constitute the best indicators of forest management and naturalness in landscapes with long human disturbance history and a fortiori in the Belgian conditions. Yet this group has the drawbacks of remaining insufficiently known and difficult to identify. And second, our analysis show that the frequently assumed correlations between the richness of different groups is not so evident. This is the case notably for plant richness which appears to be poorly correlated to the *Carabidae*, *Staphylinidae*, *Diplopoda* and *Chilopoda*.

In biodiversity assessment matters, there is a consensus for an interdisciplinary approach including scientifically tested biological and habitat indicators. However, we should avoid the situation where the number of indicators required becomes an obstacle for an operational indicator system (Larsson and Esteban 2000). So in order to be applicable on the field, the set of indicators required should remain easy to use by field technicians.

Although the wealth of forest biodiversity indicators, the choice of reliable ones remains critical and unresolved. To select them, the major task remaining is to test and validate our ostensible indicators to check what they are really telling us (Noss 1999; Nilsson et al. 2001). Still of little use, the most species rich organism group, the invertebrates and in particular insects, should be included in any serious biodiversity assessment method (Dajoz 1996; Nilsson et al. 2001). To reduce the money, time and efforts required to collect and identify such complex group, two solutions exist. The first one, the "rapid biodiversity assessment" using "recognisable taxonomic units" was successfully tested for some groups. This method consists in collecting within a few selected weeks in a standardised trap combination and just considering the level of morphospecies, i.e. taxa that are readily separable by morphological differences obvious to non-specialists (Oliver and Beattie 1993, 1996). And the second one consists in finding suitable indicators (habitat variables, other easiest groups) providing a good assessment of the invertebrates (e.g. dead wood characteristics and abundance for xylophagous insects).

For forests biodiversity assessments, the interdisciplinary approach could follow the steps listed hereafter for which the indicators must be specified depending on the objectives and the scale under investigation:

1. general description of the area (topography, soil, geology, climate, landscape matrix, phytosociology and habitat types);
2. structure of the forest (dendrometric approach at the stand-level and remote sensing approach at the landscape level);
3. complementary and relevant bioindicators (depending on the scale, the habitat type and its situation) respecting the bioindicator criteria (at least trees, plants, lichens, lignicolous fungi, some insect groups, other invertebrates and possibly mosses and some mammal or bird species respecting the criteria developed by Noss 1999);
4. complementary and reliable habitat variables (a broad example of these variables is given by Noss 1999).

Finally, we have to bear in mind that monitoring, research and land planning should be in continual interplay, with information from one always informing the others in order to reach an adaptive management. But the time lags in the response of populations to habitat degradation suggest that when the decline is detected, it may be too late to make the necessary changes in management (Noss 1999). So the best way remains to reproduce as much as possible the characteristics of close-to-nature ecosystems (e.g. disturbance regime, etc.). But the lack of reference is sorely felt, hence the prime importance of natural woodlands and strict forest reserves which can be studied as references to settle the indicators of forest biodiversity and their thresholds.

To complete this paper, our future prospects will be:

1. to identify the silvicultural reasons explaining the different communities of the groups studied prevailing under different forest habitat types;
2. to identify the "forest naturalness indicators" allowing us to classify forests on a naturalness scale;
3. to bring to the fore simple and reliable silvicultural indicators of the identified management influences in order to avoid, if possible, the tedious assessment of the invertebrates;
4. and finally to draw biodiversity-friendly silvicultural recommendations.

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Epiphytic Bryophytes and Lichens in *Quercus rotundifolia* Lam. Woodlands of Portugal and their Value as Ecological Indicators

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Abstract

The epiphytic bryophyte and lichen vegetation in *Quercus rotundifolia* woodlands in Portugal and its dependence on a number of environmental variables were investigated in six sites of Portugal. 36 bryophyte (32 mosses, 4 liverworts) and 105 lichens were found.

Multivariate analysis was used to examine the epiphytic bryophyte and lichen composition of these woodlands. TWINSPAN and Canonical Correspondence Analysis (CCA) recognized four major groups of bryophytes and lichens species, one related to pastoral woodlands, another to altitudinal woodlands, a third one to the woodlands under Atlantic influence and a fourth with generalist species.

Differences in bryophyte and lichen species composition and cover in the studied areas were attributed mainly to distinct humidity, precipitation, evapotranspiration and insolation levels. The bryophytes *Leucodon sciuroides*, *Frullania dilatata*, and the lichen *Evernia prunastri* presented the highest cover values.

A very rare bryophyte species *Zygodon forsteri*, included in the European Red Data Book, was found. Obtained results are a good basis to improve methods based on bryophytes and lichens as bio-indicators of forest biodiversity.

Keywords: bryophytes; lichens; ecological indicators; woodlands; TWINSPAN; Portugal.

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Introduction and aims

Forests cover approximately one-third (3.2 million km^2) of the European land area (Schuck et al. 2003). Mediterranean type vegetation is characterized by sclerophyllous species, namely the distribution of holm oak woodlands which has been severely affected by human activity. Today there are 20 000–30 000 km^2 of holm oak woodlands on the Iberian Peninsula, whereas potentially holm oaks would cover 300 000 km^2 (Plieninger et al. 2003).

Ancient woodlands may be essential to the continued existence of a rich and diversified flora, including bryophytes (mosses and liverworts) and lichens (Rose 1976, 1992; Tibell 1992; Humphrey 2002; Zedda 2002). The epiphytic bryophytes and lichens species represent a significant ecological component of woodland biodiversity (Kenkel and Bradfield 1986; Humphrey et al. 2002). These habitats are the refuge of many threatened species, being these organisms vulnerable groups.

Information on the distribution and rarity of some epiphytic bryophytes and some lichens are relativity well known in Portugal (Sim-Sim and Sérgio 1998; Jones 1999). However, data on the evaluation of environmental quality, are restricted mainly to olive trees (*Olea europeaea*) (Sérgio and Sim-Sim 1985; Sérgio et al. 1989; Sim-Sim et al. 1995; Sim-Sim et al. 2000). In Portuguese native woodlands, investigations on bryophyte and lichen species diversity and rarity, combined with information on the phorophyte preference, environmental variables and natural or anthropogenic causes, is not available. These data are also poorly known in the Mediterranean region in relation to the epiphytes.

The objective of this study is to examine the composition, and abundance, of the epiphytic communities on *Quercus rotundifolia* woodlands in 6 distinct sites of Portugal, included in Natura 2000 Network (Natura 2000; European Comission 2003).

Correlations between epiphytes versus phorophytes in respect to different factors, such as height of tree, aspect, microhabitat, geomorphology, management type, etc., were also considered in this work. As a result efficient instruments to minimize biodiversity loss in Portuguese woodlands can be expected. A final aim of this study is to select bryophyte and lichen species that can be used as indicators of habitat quality, and for monitoring the biodiversity in Mediterranean forests.

Material and methods

Site description

The study was carried out in mainland Portugal, and is part of a project that comprises a national survey of the oak woodlands mentioned in the Natura 2000 Network. The Iberian woodlands communities of *Q. rotundifolia*, exhibits the Habitat code: 9340, in the Habitats Directive 92/43/EEC.

Mainland Portugal comprises a surface of 92000 km^2; the climate is generally of a Mediterranean type combined with a strong Atlantic influence. Maximum elevations are in the northern central part with elevations up to 2000 m (Figure 1).

In the studied areas the elevations range from 290 to 1076 m, the rock type are schist, greywacke (Montezinho, Estrela, Malcata, and Monfurado) and calcareous (Sicó and Barrocal – Figure 1). The soils are of the Litossol Barrocal and Malcata), Cambissol (Montezinho, Estrela and Sicó), and Luvissol type (Monfurado) (Atlas do Ambiente, 2002).

In the study area with mild humid winters and hot dry summers the temperature averages are between 4.1ºC (Montezinho) to 11.6ºC (Barrocal Algarvio) in December and in July between

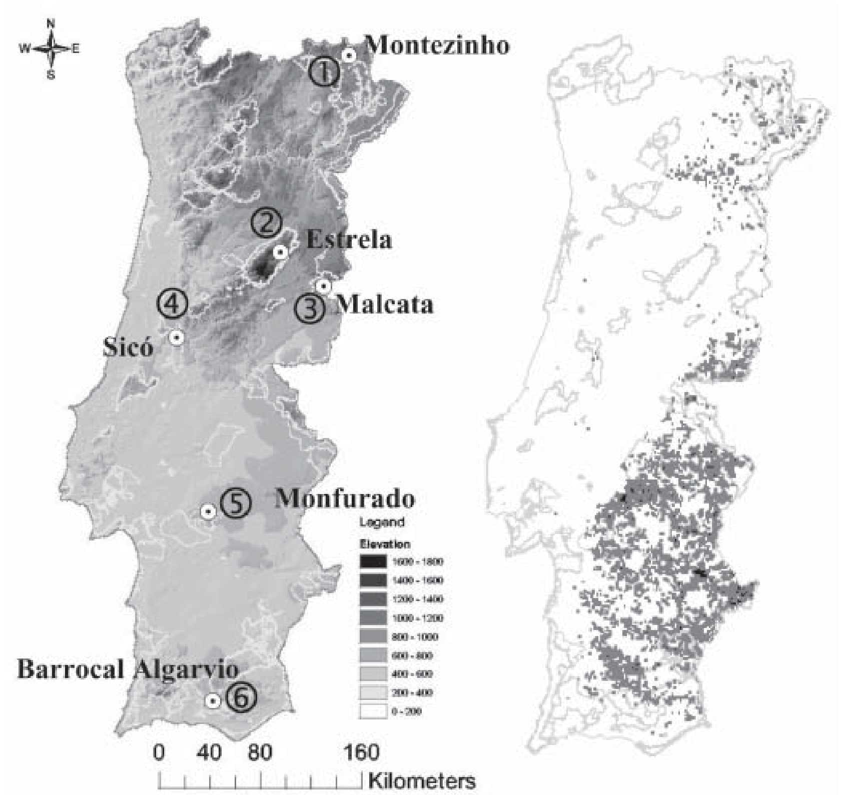

Figure 1. A) Distribution map showing the Natura 2000 areas in Portugal, altitude, and the studied sites. B) Natura 2000 areas and the distribution of *Quercus rotundifolia* according to the Direcção-Geral das Florestas. In the south and east of Portugal the woodlands areas are generally named "montados" (see text).

21.5 ºC (Montezinho) and 28.3 ºC (Monfurado). Mean annual rainfall ranges from 665 mm (Monfurado) to 1463 mm (Estrela) with a monthly maximum of 278 mm in January (Estrela) and a monthly minimum of 3.2 mm in July (Barrocal). The number of precipitation days is about 50 days in the xeric areas and above 100 days in the more humid areas (Sicó) (Figure 1).

Vegetation sampling and data analysis

The 180 surveys were performed according to the following methodology: at each site a representative *Quercus rotundifolia* seminatural woodland (at least 1 ha area and 50% of cover of the dominant trees) was studied, 15 tree trunks were examined, randomly selected (10 inside the woodland and 5 on its margin). On each tree, two relevés were performed with north and south aspect. The epiphytic bark cover for each taxon and the total epiphytic bryophyte and lichen vegetation were estimated using a 500 cm^2 grid, which as adequate to the different phorophyte diameters, and was placed on the north and south sides of each tree, at a vertical height between 150 and 180 cm. Fieldwork was carried out from 2001 to 2003.

In each site the environmental variables was obtained as represented in Figure 5. Data were collected in the field, and also the Atlas do Ambiente was consulted for information concerning some variables.

Table 1. Bryophytes and lichens recorded in the sampling quadrats on each studied site and codes used in statistical analysis. The nomenclature generally follows Corley et al. (1981) and Corley and Crundwell (1991) for mosses, and Grolle (1983) and Grolle and Long (2000) for liverworts. For lichens, nomenclature is generally from Nimis (1993), and from Clauzade and Roux (1985) for species not considered in the aforementioned work. For some taxa, species names are not given as the material collected was too scanty and/or sterile for definitive identification.

Mosses

Antca	*Antitrichia californica*
Antcur	*Antitrichia curtipendula*
Auland	*Aulacomnium androgynum*
Brycap	*Bryum capillare*
Cynbru	*Cynodontium bruntonii*
Diamuc	*Dialytrichia mucronata*
Didvinf	*Didymodon vinealis* var. flaccida
Fabpus	*Fabronia pusilla*
Habper	*Habrodon perpusillus*
Hedsp.	*Hedwigia* sp.
Hedste	*Hedwigia stellata*
Homser	*Homalothecium sericeum*
Hypcup	*Hypnum cupressiforme*
Lepsmi	*Leptodon smithii*
Leusci	*Leucodon sciuroides*
Ortacu	*Orthotrichum acuminatum*
Ortaff	*Orthotrichum affine*
Ortdia	*Orthotrichum diaphanum*
Ortibe	*Orthotrichum ibericum*
Ortlye	*Orthotrichum lyellii*
Ortrup	*Orthotrichum rupestre*
Ortsp.	*Orthotrichum* sp.
Ortstr	*Orthotrichum striatum*
Ortten	*Orthotrichum tenellum*
Ptegra	*Pterogonium gracile*
Rachet	*Racomitrium heterostichum*
Torfla	*Tortella flavovirens*
Torlae	*Tortula laevipila*
Zygrup	*Zygodon rupestris*
Zygrupm	*Zygodon rupestris* var. *mediterraneum*
Zygsp.	*Zygodon* sp.
Zygvir	*Zygodon viridissimus*

Liverworts:

Frudil	*Frullania dilatata*
Frutam	*Frullania tamarisci*
Radlin	*Radula lindenbergiana*
Radsp.	*Radula* sp.

Lichens:

Agonsp.	*Agonimia* sp.
Amanpunc	*Amandinea punctata*
Anapcili	*Anaptychia ciliaris*
Bacirube	*Bacidia rubella*
Caloferr	*Caloplaca ferruginea*
Caloholo	*Caloplaca holocarpa*
Candconc	*Candelaria concolor*
Candrefl	*Candelariella reflexa*
Candxant	*Candelariella xanthostigma*
Chrycand	*Chrysothrix candelaris*
Cladpyxi	*Cladonia pyxidata*
Cladsp	*Cladonia* sp.
Cladauri	*Collema auriforme*
Collfurf	*Collema furfuraceum*
Collnigr	*Collema nigrescens*
Collsubfu	*Collema subflaccidum*
Collsubn	*Collema subnigrescens*
Degeplum	*Degelia plumbea*
Dendumha	*Dendriscaulon umhausense*
Diplcane	*Diploicia canescens*
Everprun	*Evernia prunastri*
Heteleuc	*Heterodermia leucomelos*
Hypeadg	*Hyperphyscia adglutinata*
Hypostoe	*Waynea stoechadiana*
Hypophys	*Hypogymnia physodes*
Hypotubu	*Hypogymnia tubulosa*
Koerbifo	*Koerberia biformis*
Lecaallo	*Lecanora allophana*
Lecachla	*Lecanora chlarotera*
Lecaconi	*Lecanora conizaeoides*
Lecapall	*Lecanora pallida*
Lecapull	*Lecanora pulicaris*
Lecasp.	*Lecanora* sp.
Lecielae	*Lecidella elaeochroma*
Leprsp.	*Lepraria* sp.
Leptbreb	*Leptogium brebissonii*
Leptlich	*Leptogium lichenoides*
Leptsatu	*Leptogium saturninum*
Leptsch	*Leptogium schraderi*
Lobapulm	*Lobaria pulmonaria*
Lobascro	*Lobaria scrobiculata*
Nephlaev	*Nephroma laevigatum*
Nephtang	*Nephroma tangeriense*
Normpulc	*Normandina pulchella*
Ochrsubv	*Ochrolechia subviridis*
Parmtar	*Ochrolechia tartarea*
Pannmedi	*Pannaria mediterranea*
Parmarno	*Parmelia arnoldii*
Parmcape	*Parmelia caperata*
Parmcarp	*Parmelia carporrhizans*
Parmexas	*Parmelia glabra*
Parmglab	*Parmelia laevigata*
Parmquer	*Parmelia quercina*
Parmsuba	*Parmelia subaurifera*
Parmsulc	*Parmelia sulcata*
Parmtili	*Parmelia tiliacea*
Parmreti	*Parmotrema reticulatum*
Peltcoll	*Peltigera collina*
Pertalbe	*Pertusaria albescens*

Table 1. continued.

Pertamar	*Pertusaria amara*	Physomph	*Physma omphalarioides*
Pertcaes	*Pertusaria caesioalba*	Placnigr	*Placynthium nigrum*
Pertcocc	*Pertusaria coccodes*	Pseufurf	*Pseudoevernia furfuracea*
Pertfico	*Pertusaria ficorum*	Pyrrquer	*Pyrrhospora quernea*
Pertflav	*Pertusaria flavida*	Ramacali	*Ramalina calicaris*
Perthemi	*Pertusaria hemisphaerica*	Ramacana	*Ramalina canariensis*
Perthete	*Pertusaria heterochroa*	Ramafari	*Ramalina farinacea*
Perthyme	*Pertusaria hymenea*	Ramafast	*Ramalina fastigiata*
Pertpert	*Pertusaria pertusa*	Ramafrax	*Ramalina fraxinea*
Pertsp.	*Pertusaria* sp.	Ramasubg	*Ramalina subgeniculata*
Phaeorbi	*Phaeophyscia orbicularis*	Rinocape	*Rinodina capensis*
Phlyarge	*Phlyctis argena*	Rinocort	*Rinodina corticola*
Physadsc	*Physcia adscendens*	Rinoisid	*Rinodina isidioides*
Physaipo	*Physcia aipolia*	Rinosp.	*Rinodina* sp.
Physcaes	*Physcia caesia*	Sphiturbin	*Sphinctrina turbinata*
Phystene	*Physcia tenella*	Sticlimb	*Sticta limbata*
Physdete	*Physconia detersa*	Tephatra	*Tephromela atra*
Physdist	*Physconia distorta*	Usnecera	*Usnea ceratina*
Physente	*Physconia enteroxantha*	Usneespe	*Usnea esperantiana*
Physgris	*Physconia grisea*	Usneflor	*Usnea florida*
Physperi	*Physconia perisidiosa*	Usnerubi	*Usnea rubicunda*
Physsubp	*Physconia subpulverulenta*	Usnesp.	*Usnea* sp.
Physvenu	*Physconia venusta*		

Location of sites and details of the studied sites:

Local, UTM, altitude, dominant vascular plant species, and sampling date.

1. Trás-os-Montes e Alto Douro: Bragança, Rabal, 29TPG8737, 644 m, *Quercus rotundifolia*, *Cistus ladanifer*; *Daphne gnidium*; *Lavandula stoechas*, 12-07-2002.

2. Beira Alta: Serra da Estrela, próximo do Cerro do Gato, vertente do Mondego, 29TPE3281, 1076 m, *Quercus rotundifolia*, *Genista florida*; *Chamaespartium tridentatum*; *Erica* spp.; *Pinus pinaster*, 04-06-2003.

3. Beira Baixa: Serra da Malcata, Quinta do Major, 29TPE6754, 611 m, *Quercus rotundifolia*; *Q. pyrenaica*; *Arbutus unedo*; *Erica* sp.; *Pteridium aquilinum*; *Vaccinium myrtillus*; *Rubus* sp.; *Rubia peregrina*; *Fraxinus angustifolia*; *Olea europaea*; *Osmunda regalis*; *Phillyrea angustifolia*; *Alnus glutinosa*, 03-06-2003.

4. Beira Litoral: Serra de Sicó, Alvaiazere, Venda do Negro., 29SNE5013, 425 m, *Quercus rotundifolia*; *Q. faginea*; *Crataegus monogyna*; *Laurus nobilis*; *Cistus monspeliensis*; *Arbutus unedo*; *Rhamnus alaternus*; *Smilax aspera*; *Peonea broteroi*; *Cistus salvifolius*; *Cistus crispus*; *Rubia peregrina*.; *Tamus communis*, 11-07-2001.

5. Alto Alentejo: Serra de Monfurado, Herdade dos Padres., 29SNC7575, 330 m, *Quercus rotundifolia*; *Q. suber*; *Crataegus monogyna*; *Cistus salvifolius*; *Asparagus* sp.; *Prunus spinosa*; *Arum italicum*; *Umbilicus rupestris*; *Arisarum vulgare*, 10-12-2001.

6. Algarve: Barrocal, Rocha da Pena, Quinta do Freixo, 29SNB7924, 290 m, *Quercus rotundifolia*, *Pistacia lentiscus*; *Juniperus phoenicea*; *Smilax aspera*.; *Viburnum tinus*; *Arbutus unedo*, *Myrtus communis*; *Cistus albidus*; *Cistus crispus*; *Osyris alba*; *Rosmarinus officinalis*; *Lavandula stoechas*; *Lonicera* sp.; *Rhamnus oleoides*; *Phlomis* sp.; *Olea europaea* var. *sylvestris*; *Daphne gnidium*; *Ruscus aculeatus*; *Juniperus turbinata*; *Ceterach officinarum*; *Peonea broteroi*, 12-03-2002.

Statistical analysis

Species covers of each taxon (mosse, liverwort and lichen species) were subjected to a TWINSPAN (Hill 1979) analysis, a divisive hierarchical method that produces a two-way classification of species and stands. Species cover and environmental variables were ordinated simultaneously using CCA carried out by means of CANOCO (Ter Braak 1991) in order to assess the relationship between species cover percentage and environmental variables.

Results and discussion

In the 6 sampling sites, 141 taxa were recorded, 36 bryophytes (32 mosses, 4 liverworts) and 105 lichens.

As a result of the statistical analysis, differences in community composition and floristic richness were observed (see Figure 2).

In general, lichens are more abundant in terms of species numbers and cover percentage, apart from no 4 (Sicó in Figure 1), There, mosses presents an higher percentage of cover, as is it a shady area with a strong Atlantic influence.

The climate in the studied region whole of the Mediterranean type with mild humid winters and hot dry summers is more appropriate in terms of epiphytic flora to the development of the lichens (Table 2).

We found a corresponding reduction in the number of bryophytes from north towards the south of Portugal, and an increase of the lichen biodiversity. Differences were also noted between trees in the woodland and isolated trees. In some areas like no 3 (Malcata) (Figure 1), the isolated trees showed a very high cover of *Frullania dilatata* and *Habrodon perpusillus*, with strong Mediterranean influence which is unusual. However in some areas this was not observed, because the environmental conditions both for isolated trees and woodlands trees were similar (Barrocal and Monfurado) apart from the closed forests in the north of the country.

On woodlands trees and north versus south aspect, some differences were observed in the studied area (Table 2). The number of species and their cover percentage was higher on the northern aspect of the tree trunk.

The TWINSPAN analysis (Figure 2) shows that the first and second division separates the 141 taxa into four major groups. The eigenvalues of the two CCA axes are 0.68 and 0.56. The species environmental correlations are 0.99 and 0.98 respectively. Figure 3 shows the TWINSPAN division to the 3rd level, along with the indicator species for each group.

The CCA biplots of the first two axis show that the distribution of individual species is influenced by environmental factors (Figures 4 and 5). The first axis is mainly correlated with the number of frost days and temperature, demonstrating a gradient from the relevés with higher temperatures (left) to those with lower temperatures (right).

The second axis shows a positive correlation with insolation. In addition, there is also a negative correlation, with the annual precipitation mean and the number of rainy days. These results are according to a gradient observed for the relevés in closed forest, with higher numbers of rain days than relevés carried out in pastoral woodland open sites (Figure 4 and 5). Most evergreen oak forests growing in some areas of the southwestern Iberian Peninsula have been gradually transformed into a unique kind of pastoral woodland, the portuguese montados and the spanish dehesas. By means of an agro-forestry use, human induced transformations of the original forests by tree clearing and land ploughing (Pulido et al. 2001). Several taxa associated to the portuguese pastoral woodlands (▲ Figure 4) were found, more correlated with maximal temperature, enthalpy of air and land use (agriculture) (Figures 4 and 5) include bryophytes such

Table 2. Species number and percentage cover in each studied site, forests / isolated trees; and north (N)/ south (S) aspect.

		Montesinho		Estrela		Malcata		Sicó		Monfurado		Barrocal Algarvio	
		Taxa nº	Cover %	Taxa nº	Cover %	Taxa nº	Cover %	Taxa nº	Cover %	Taxa nº	Cover %	Taxa nº	Cover %
FLOREST N	Liverworts	1.0	3.5	1.0	5.9	1.0	24.4	3.0	14.0	1.0	17.2	1.0	5.6
	Mosses	12.0	21.6	12.0	37.9	11.0	31.6	5.0	44.4	5.0	37.7	2.0	17.9
	Lichens	21.0	57.7	13.0	41.8	13.0	58.2	19.0	35.8	24.0	52.3	28.0	29.8
FLOREST S	Liverworts	1.0	0.3	1.0	0.5	1.0	12.2	3.0	1.0	1.0	3.6	1.0	0.5
	Mosses	11.0	18.9	10.0	8.0	8.0	19.1	5.0	45.0	7.0	21.0	2.0	1.9
	Lichens	20.0	58.5	9.0	85.9	9.0	33.2	14.0	31.4	28.0	49.9	29.0	29.7
ISOLATED N	Liverworts	1.0	1.0	1.0	0.5	1.0	13.3	2.0	13.5	1.0	4.5	0.0	0.0
	Mosses	3.0	3.8	11.0	33.8	11.0	61.8	6.0	36.0	5.0	17.5	3.0	6.7
	Lichens	17.0	72.7	11.0	60.2	10.0	28.5	2.0	1.0	27.0	68.5	20.0	26.7
ISOLATED S	Liverworts	1.0	0.9	1.0	0.5	0.0	0.0	0.0	0.0	1.0	2.0	0.0	0.0
	Mosses	2.0	0.9	9.0	19.9	11.0	44.6	3.0	25.5	4.0	7.5	4.0	8.6
	Lichens	20.0	79.3	10.0	55.8	7.0	27.5	2.0	1.5	22.0	74.5	24.0	37.0

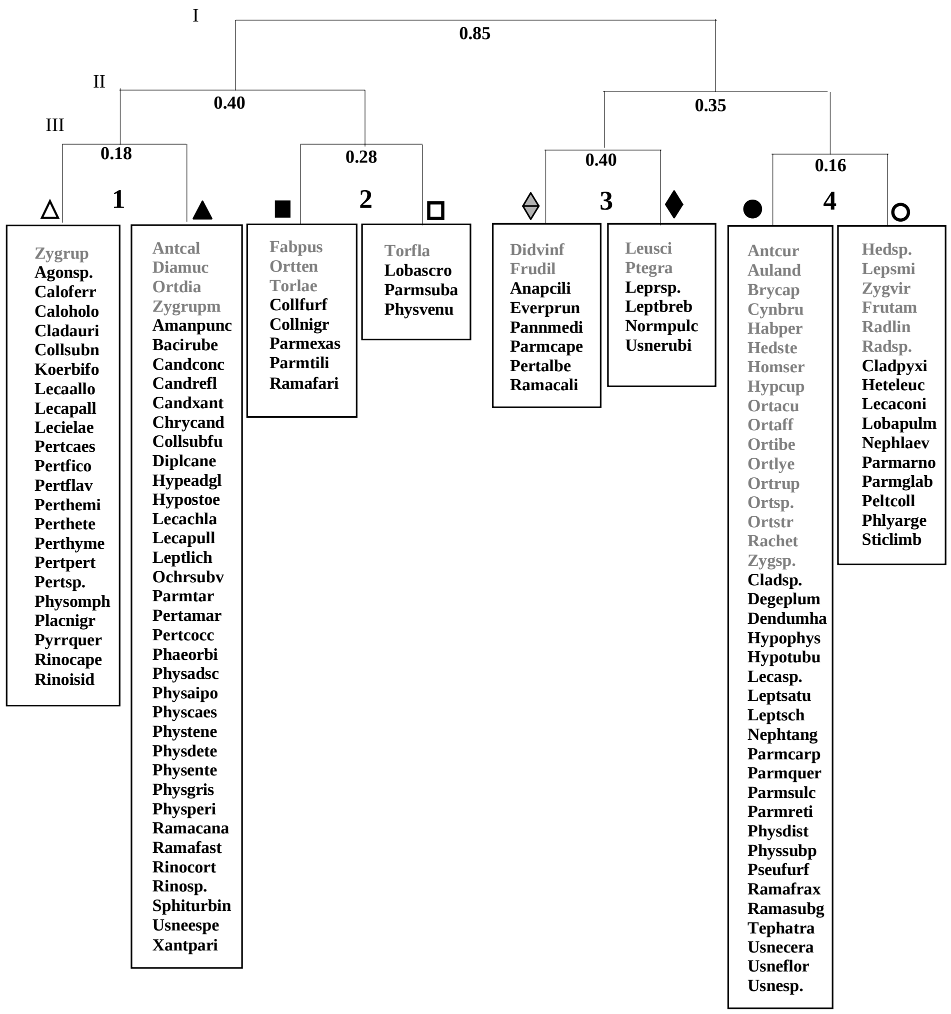

Figure 2. Dendrogram representing the major divisions of TWINSPAN (3 rd level) classification of the epiphytic species groups from *Quercus rotundifolia* in Portugal, based on percentage cover of each species. Gray color the bryophytes, and the black color the lichens

as *O. diaphanum*, *Zygodon* sp., and several species of lichens genus: *Pertusaria*, *Lecanora* and *Physcia*. The high species number and cover values of lichens are explained by the eutrophicated and light conditions, due to the open structure (and dry microclimate), especially in the south of Portugal (Monfurado and Barrocal Algarvio no 5 and 6) (Figure 1). The light strongly affects temperature and evaporation, rendering it one of the decisive factors of epiphyte environment in an indirect way (Barkman 1958).

The intermediate positions (2 and 3 Figure 2 and 4) on the axis are represented by species with a wide distribution in the studied area, such as *Fabronia pusilla*, *Orthotrichum tenellum*, *Tortula laevipila*, *Frullania dilatata* and lichens such as *Collema* spp., *Evernia prunastri*, *Parmelia caperata*, *Physconia venusta* and *Leptogium brebissonii*.

The CCA ordination also show the presence of a group found at highs altitudes (●-Figure 4) (Montezinho, Estrela and Malcata numbers 1, 2, 3) (Figure 1). In the Centre-North of

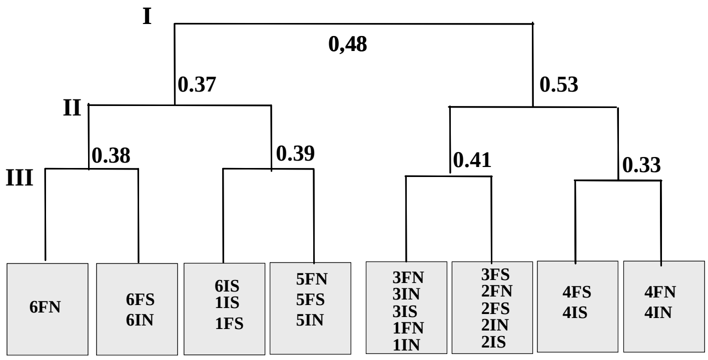

Figure 3. Dendrogram representing the major divisions of TWINSPAN (3 rd level) classification of the sites groups from *Quercus. rotundifolia* in Portugal, based on species cover. Abbreviations: F – relevés carried out on forest trees, I – relevés carried out on isolated trees, N – relevés carried out on northern aspect, S – relevés carried out on southern aspect.

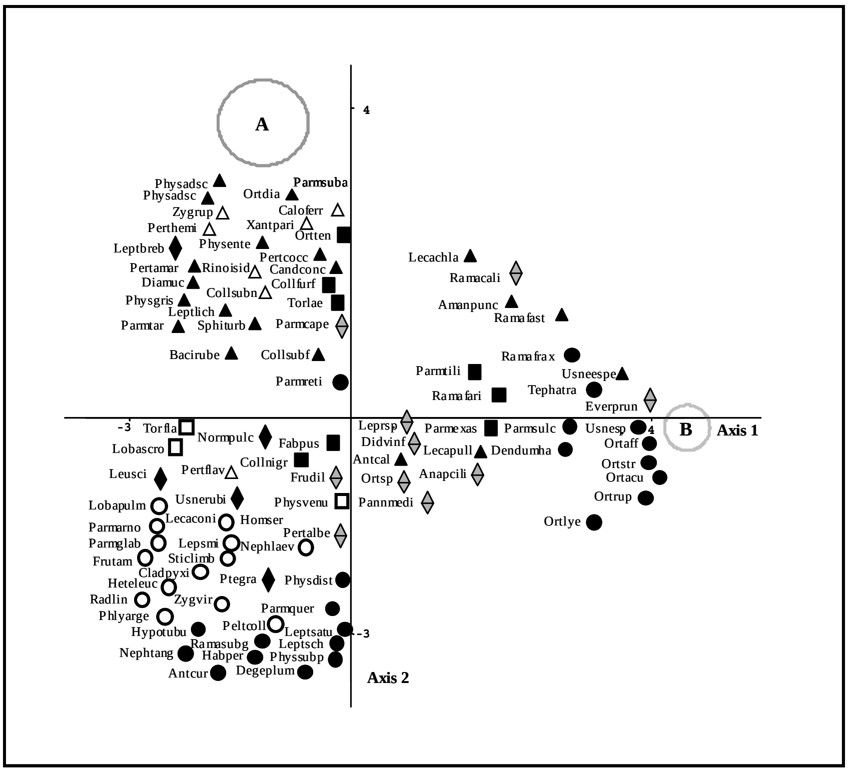

Figure 4. Ordination diagram of species on axes 1 and 2 (abbreviations as in Table 1), based on CCA of epiphyte assemblages on *Quercus rotundifolia* in Portugal. The symbols are according to the 8 species groups isolated by TWINSPAN (Figure 2). The group A included: Caloholo, Lecaallo, Koerbifo, Pyrrquer, Agonsp., Cladauri, Rinocape, Pertpert, Perthete, Lecielae, Placnigr, Physomph; Pertcaes, Hypeadgl, Pertfico, Ochrsubv, Rinosp., Phystene, Lecapall, Hypostoe, Rinocort, Zygrupm, Physcaes, Diplcane, Phaeorbi, Physperi, Physaipo, Physdete, Ramacana, Candrefl, Perthyme, Candxant, Chrycand. The group B included: Parmcarp, Ortibe, Hypophys, Usneflor, Usnecera, Brycap, Pseufurf, Rachet, Auland, Hedste, Cynbru, Hypcup.

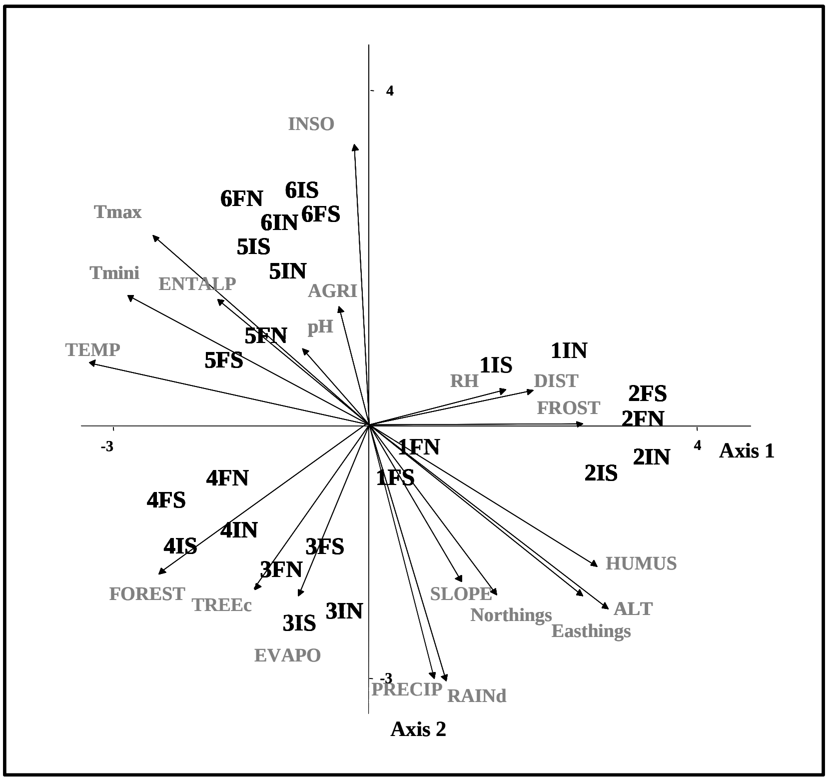

Figure 5. Ordination diagram of samples and environmental variables on axes 1 and 2 based on CCA of epiphytes species on *Quercus rotundifolia* in Portugal. Abbreviations: AGRI – Land uses (agriculture), ALT – Altitude (m), DISTANCE – Distance to roads and ways, EASTHINGS – Longitude, ENTALP – Enthalpy of air (kcal.kg^{-1}), EVAPO – Annual Evapotranspiration Mean (mm), FOREST – Forest (mixed / pure), FROST – Frost (monthly mean) (days), HUMUS – Humus (%), INSO – Insolation (monthly mean) (hours), NORTHINGS – Latitude, pH – pH (Bark), PRECIP – Annual Precipitation Mean (mm), RAINd – Number of days with rain, RH – Annual Air Humidity Mean (%), SLOPE – Slope (º), TEMP – Daily Temperature Mean (ºC), Tmax – Maximal temperature – (monthly mean) (ºC), Tmini – Minimal temperature (monthly mean) (ºC), TREEc – Tree cover (%). Samples abbreviations: 1–5 number of site, F – relevés carried out in forest trees, I- relevés carried out on isolated trees, N – relevés carried out with North aspect, S – relevés carried out with South aspect.

Portugal most of the remaining *Q. rotundifolia* woodlands are fragmented into small ungrazed islands and mosaics (1–3 ha), without pasturing. In these “forests islands” we found several species of phytogeographical and ecological value namely *Antitrichia curtipendula*, *Orthotrichum acuminatum*, *Orthotrichum lyellii*, *Orthotrichum ibericum*, *Degelia plumbea*, *Usnea florida* as well as other bryophyte and lichen species.

Another group (○ -Figure 4) can be related to the atlantic woodlands (site no 4, Sicó). These rich woods contain rare atlantic species, have supported some continuous tree canopy over the last 1200 years and contain an abundance of moist, shaded habitats (Edwards 1986). Species as *Leptodon smithii*, *Frullania tamarisci*, *Radula lindenbergiana*, *Lobaria pulmonaria*, *Nephroma laevigatum*, *Sticta limbata*, and many others species were found in this type of woodland. This group is correlated with pure forests, higher tree cover and higher annual evapotranspiration (Figures 4 and 5).

A very rare bryophyte species, *Zygodon forsteri,* found during the course of this work but not in the relevés in the Montezinho area. This species is probably limited in this distribution by its precise habitat requirements. It is listed as Vulnerable in the 1995 Red Data Book of European Bryophytes (ECCB 1995), and Endangered in the Red List of the Iberian Peninsula (Sérgio et al. 1994).Several *Lobarion* species were also found in some places, indicating a very good environmental quality. The *Lobarion* still occurs in some places left undisturbed and it seems to have little or no ability to colonise new sites (Humphrey et al. 2002). The *Lobarion* is a lichen-dominated, species-rich epiphytic plant community, declining in most parts of Europe (Gauslaa et al. 1995; Jones 1999).

There are important criteria to consider when selecting groups of species to be used as indicators of biodiversity (Pearson and Cassola 1992; Lawler et al. 2003), the distributions of species in the groups should be well-known or easily determined (Lawler et al. 2003). In Europe many countries have checklists and Red Lists for bryophytes and lichens.

Portugal has a good database concerning the ecological requirements for bryophytes. An important issue in conservation biology is the extent to which one group of organisms can function as a surrogate for less well-known groups (Pharo et al. 1999). The relationships between potential indicator species and the total biodiversity are not well established. Carefully designed studies are required to test relationships between the presence and abundance of potential indicator species and other taxa and the maintenance of critical ecosystem processes in forests (Lindenmayer et al. 2000). Moreover, applying “Gap analysis” in Portuguese data, Sérgio et al. (2000) suggested that bryophyte species reflect the biodiversity richness in a given region, as they are the best elements that can be used as bioindicators.

Conclusions

The present study has demonstrated that the bryophytes and lichens are good indicators of the management type, and climate and can be good instruments for indicating the biodiversity in woodlands habitats. The results of multivariate analyses are essentially a sum of the differences between northern and southern, oest-west, altitude and management type. However, in Portugal this type of forest has a great importance as they include high diversity, so conservation planning and appropriate management practices are recommended.

In our opinion is not possible to infer the potential of the total biodiversity in a area without having in account other parameters like pollution, climate, geographic position and altitude. Distribution areas may depend also on latitudinal, longitudinal, or altitudinal gradients. The great advantage of the cryptogamic flora is that some species can be used as pollution indicators, as well as indicators of environmental conditions and management type, as we intend to demonstrate in this work for the *Quercus rotundifolia* woodlands in Portugal. As final remark we propose the establishment of close links between taxonomists, biogeographers and climatologists and the training of specialists in cryptogramie in each country.

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Session 5: Forest Indicator Development at Landscape Level

Cultural Landscapes and Biodiversity: A Case Study Suggesting a Different Approach In Conservation Strategies in Italy

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Abstract

The conservation of biodiversity may require a different approach especially when referred to the diversity connected to cultural landscape. The way in which biodiversity is often interpreted in protected areas shows an emphasis upon biodiversity in species, but not in the preservation of the diversity represented by the complex landscape mosaics featuring many territories. In several countries the end of traditional agricultural and forestry practices and the abandonment of rural areas are endangering landscape diversity, especially when forest successions are occurring on former cultivated fields and pastures. This process is somehow favoured also by many EU directives in agriculture an environmental conservation. This paper shows that landscape diversity had a dramatic decrease in the last 200 years (–86%) in a study area inside the Apuane Regional Park (Tuscany, Italy), but this problem is not addressed nor by the park or the regional authorities. This paper argues that the preservation of landscape diversity is sometime more important than the preservation of large dense forest cover reducing diversity, and it also suggests that a revision of international standards in order to suite different concepts of sustainability is needed. In countries where the environment is essentially a “cultural landscape”, the conservation of landscape diversity has often a better chance to ensure sustainability rather than natural evolution. The case study presented is taken from an international research project aimed to analyse and evaluate landscape resources in Tuscany, the research analysed almost 25 000 ha of territory from 1832 to 2000, with a multidisciplinary approach using historical, ecological and economical investigations. The results of this study are now applied in several projects as the creation of a landscape park, the Environmental Impact Assessment, the new guidelines for the management of protected areas.

Keywords: biodiversity; landscape; GIS; history; conservation.

Marco Marchetti (ed.)
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EFI Proceedings No. 51, 2004

1. Introduction: the problem of landscape diversity

The application of European directives concerning nature conservation as well as the structure of forest certification standards, requires a profound reflection on the concepts of sustainability they are proposing, as well as on the effect of their application in relation to the cultural and historical values of landscape. As a matter of fact, the hierarchy proposed by the ministerial conference for forest protection in Europe shows that CO_2 storage, sustainable timber production and biodiversity of species play a major role in defining sustainability. However, in countries like Italy, where the small extension of forest land allows only a minor role of the forests concerning CO_2 at world scale, and forest utilizations are not surely overexploiting woodlands, these concepts can hardly be considered representatives of the real values involved in sustainability. Even more contradictory seems the interpretation of biodiversity especially in the management of protected areas (parks, natural reserves, etc.) where the emphasis is often placed upon biodiversity in species, but not in the preservation of the diversity represented by the complex landscape mosaics featuring the Italian territory. The interruption of traditional agricultural and forestry practices and the abandonment of rural areas are endangering landscape diversity, especially when forest successions are occurring on former cultivated land and pastures. The reduction of landscape diversity is indirectly supported also by European policies, as the incentives in agriculture encourage the reduction of traditional cultivations and forest plantations in former pastures or cultivated land, while certification standards in forestry are also not addressing the conservation of the historical, cultural and landscape values. In countries where the forest environment is essentially a "cultural landscape", as it happens in most of Mediterranean regions (Grove and Rackham 2001), representing the integration of social, economic and environmental values through time, the conservation of landscape diversity according to historical land uses has often a better chance to ensure sustainability rather than favouring natural evolution. As a matter of fact, if nature conservation and the network of Nature 2000 has actually established a legislative framework allowing an active conservation in designated areas, it is important to look at the way these areas are managed and the effect of management strategies at landscape level.

2. Methods

The project made for the Tuscan Regional selected 12 study areas representing the main landscape unit featuring the Tuscan territory: the mountain, the hills and the coast. These units are also resulting from a first attempt to classify landscape in Tuscany (Rossi 1996), although this work only referred to a general morphological classification, rather than to the features of land uses. The study areas have an average extension of 900 ha, for a total extension of 25 000 ha, in each one of them an historical investigation from 1832 to 2000 has been carried out. Three basic sources have been considered: the cadastre of 1832, the aerial photographs of 1954, the aerial photographs of 2000, with an approach already adopted in other studies (Foster 1992), but applied systematically in all selected areas (Agnoletti 2002). The cadastre of 1832 is survey on land uses at the scale 1:5000 carried out for almost all the Tuscan territory, with very detailed information on crops, trees and woodlands existing at the time. For each area archival and printed documents have been collected to help a better understanding of landscape changes according to socio economic developments, including farming activities and demographic changes. Field work helped for a better understanding of the present situation, describing forest vegetation and the features of the soils. The information collected where transferred into a GIS database, using the technical maps and

Figure 1. Distribution of study areas in Tuscany. The study area presented is marked with number 10.

DTM provided by the regional geographical services, producing thematic maps with a minimum cartographic unit detected of 1500 mq. In the case of the Apuane study area the year 1954 was not used, but instead aerial photographs of the 1981 flight were selected, since the regional office for nature conservation wanted to evaluate the effects of the management made by the Apuane park, established in 1985, on landscape diversity. The cadastre of 1832 reflects a historical moment characterized by what can probably considered the highest diversity of landscape, in terms of land uses, although this is often increasing in the following years and then reducing from the early decades of the 20th century, especially in mountain areas. The cadastre shows a very complex landscape mosaic, together with the morphological variety of Tuscany this complexity represents the most important feature of the landscape in the region. The cadastre of 1832 is still today a very detailed documents, compared to present cadastre and to remote sensing surveys, at least for what concern the extension and structure of landscape patches. One of the problems of the methodology adopted is the reduced ability to detect "slash and burn" techniques well know in the area and studied in the near by Ligurian Appennine (Bortolotto and Cevasco 2000), for this reason the survey is simply drawing a picture of the areas in three different years, but not showing changes due to rotation systems of crops.

3. The study area

The study area presented is placed inside the Regional Park of the Apuane Alps, extending for approximately 21 000 ha in the provinces of Lucca and Massa, and including 16 villages. The study area has an extension of 1053.5 ha, it goes from the top of mount Pania della Croce (1858 m above the sea level) down to the river Cardoso, including one Special Area of Conservation and one Special Protected Area. The area chosen allowed us to analyse the whole altitudinal gradient including the bottom of the valley, the village of Cardoso and the mountain top. The climate of the area is sub-atlantic dominated by the wind blowing from the sea, average rain fall in the last thirty years vary between 2000 and 3000 mm, while medium temperature range between 15 °C at sea level and 7.5 °C close to mountain tops. The natural vegetation, between 300 and 1000 meters, is dominated by *Ostrya carpinifolia* and *Quercus pubescens*, with an understory mostly made of *Erica carnea*, *Amelanchier ovalis* and *Viburnum lontana.* This vegetation belt, once extending over most of the Apuane Alps, it has been affected by dramatic changes induced by the influence of the man. The rise of an economy based on pastures, woodlands and fields, as well as the development of the first villages can be traced back to the 8th–5th century B.C. (Decandia 1994). Before the Romans the populations living there, mostly Liguri, already cultivated the land using "slash and burn" techniques and rotation, to alternate crops. The coming of the Romans about the 2nd century B.C. extended cultivated areas and reduced forests, but one of the most important fact is probably the increasing cultivation of chestnut orchards (Pitte 1986), more often named in the written documents of the 13th century. This species has dominated the local economy for centuries, as the growth of population did not allowed surviving in mountain areas without the help of chestnut flour, used by the farmers to integrate their diet, since cereal production at this altitude was quite limited. At Italian level the cultivation of chestnut has been extended from the seal level up to 1500 m a.s.l., often independently from soil conditions, far beyond the climatic limits assigned to this species. The chestnut can perhaps be considered the most important cultural tree in Italy, as well as in Tuscany and in this study area, where it often presents the features of a monumental tree several centuries old, but it is now strongly reducing its extension. The abandonment of chestnut cultivation favours species like *Ostrya carpinifolia,* more easily growing on calcar soils, while *Fagus sylvatica* is colonizing the higher side of mountain slopes. At higher altitude pastures were regularly burned until a few decades ago, this has deeply affected the species composition of meadows, not allowing to ascribe their structure to a specific type. In one way this is also the situation of woodlands, that cannot be described according to potential vegetation pattern because the action of the man has totally changed their composition. The cultivation of chestnut, as well as vines and cereals, was often made terracing the slopes, a process already started in medieval times but probably increased in the 19th century when there is the strongest population growth ever registered in the Italian mountain, allowing the population to grow by 50% from 1861 to 1920.

4. The analysis of landscape changes

In 1832 we can see the presence of a complex landscape mosaic organized in more than 65 land uses and 618 patches, dominated by pastures and meadows (38%), cultivated land (32%), while the woods are covering only 20% of the territory. The reduced amount of forest does not mean a reduced amount of tree species in the area, they were an important element of 27 ha of fields, pastures, meadows, all described as wooded. An estimate of the number of trees per hectare in 1832 could not be made, but it must be remembered that farmers used to plant many trees in the field to feed cows with leaves, but also to produce timber, fuel wood and other products like bark and fruits. The number of species per ha estimated in other rural areas shows even 165 trees/ha, showing that the density of tree species in the fields could even exceed the number of trees in many Italian forests at that time (Cazzola 1996). The relative reduced density of Italian forests was often due to grazing carried out in many woodlands called "pastured woods" (in opposition to "wood pasture"), requiring a reduced number of trees to allow a large development of the crowns needed to produce nuts. In the

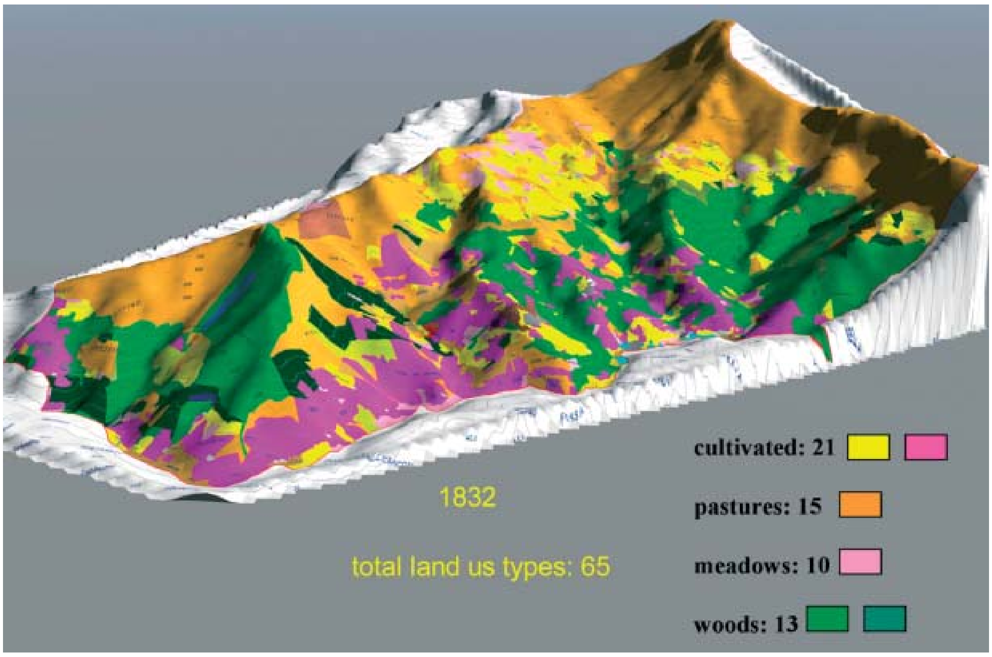

Figure 2. The landscape mosaic of the area in 1832 was characterized by 65 different land uses, the number next to each land use class indicates a different type of wood, field, meadow, pasture described by the cadastre.

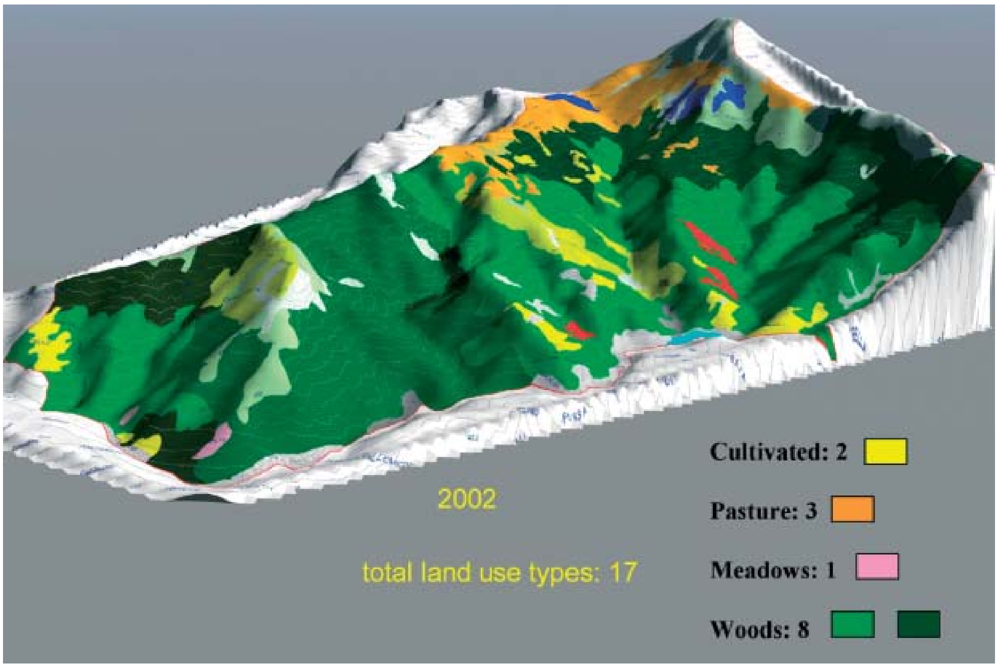

Figure 3. The landscape mosaic in 2002 is characterized by only 17 land use types.

area usually vineyards were cultivated with vines raised bound to maple trees (*Acer campestris*), a technique used since Etruscan times, but mulberry trees, walnuts, chestnut, heather and many fruit trees are described in the fields. The land use classes described in Figure 1 means that different types for each class existed, cultivated fields had not only different crops (e.g. rye, wheat, corn) but also different tree species growing in each type. In the same way the land use class "pastures" can be categorized into "bare pastures" and "wood pastures", the latter again divided into 11 types according to the different trees growing.

Intensive farming in this area continued up until 1920–30 although there where already clear signs of a crisis in food production, partly helped by the import of wheat, a process already described at European level (Sylla and Toniolo 1991), and not solved even by the fascism that placed a lot of emphasis on self-sufficiency of the country. In 1921 there are the first signs of an inversion of demographic trend, but a dramatic fall in population living occurred quite rapidly only after the second world war, with the turn of Italy into and industrialized country. In the decades between 1950 and 1980 the resident population goes back to the situation of the early 19th century, inducing a dramatic change in all the features of this mountain environment due to the suspension of farming activities.

The changes occurred in the area between 1832 and 2002 can be analysed at three different levels: considering the evolution of the main process affecting landscape, the main land use types and the evolution of every land use class. In Figure 4 it is quite evident the role played by the forest in the changes affecting the area, as forest extension concern 49% of the changes occurred, but it must be observed that even the 42% related to "unchanged" does not actually mean that there are no changes at all, but only that the main land use categories considered (e.g. "woods", pastures, fields) have not been transformed into another land use. As the cross tabulation in Table 1 shows, inside these categories there can be changes due the evolution of the internal structure of every land use type considered.

Taking into account the changes in the three main land use types in Figure 5 is worth noting the dramatic fall of fields going from 234 to 26 ha, followed by pastures decreasing from 483 ha to 107 ha. The loss of fields and pastures and the secondary successions occurring explains the extension of woodlands passing from 315 ha in 1832, to 818 ha in 2002. These landscape changes represent pretty well the effects of the dramatic socio-economic transformations occurring in this mountain areas. It is worth noticing that between 1981 and 2002 this trend is reducing its speed, but woodlands continues to expand, a process reported in similar situations in Europe and US (Foster 1998).

In Table 1 it is possible to analyse more closely the changes occurred in all the land use types defined in 1832. It has been necessary to reclassify the 65 land use classes of 1832 to allow a comparison with 2002, therefore a certain amount of information are lost in this matrix, but is possible to follow the evolution for the most important types. It is important to note how most of the new woods in 2002 are growing on former pastures, wood pastures, and fields with vines once covering a large portion of the area, but also the absence of the class "chestnut" in the land use of 2002. In other words, all the former pure stands of chestnut are now turned into a mixed forest with hornbeam and oak, because they are not cultivated anymore, as described for other areas in Tuscany especially at lower altitude (Agnoletti and Paci 1998). This natural trend is helped by the management policies carried out by the park of the Apuane, favouring mixed woods and the extension of forest land. In this respect it is interesting to note that the landslides occurred during a severe climatic event in 1996, all took place on abandoned chestnut groves and terraced fields, suggesting that the interruption of rural practices has probably favoured degradation and erosion.

Considering the main focus of this research it is very important to note how the number of patches of the landscape mosaic, and the number of land use types dramatically decreased (see Table 2). It is also very significant the change in the average extension of each patch,

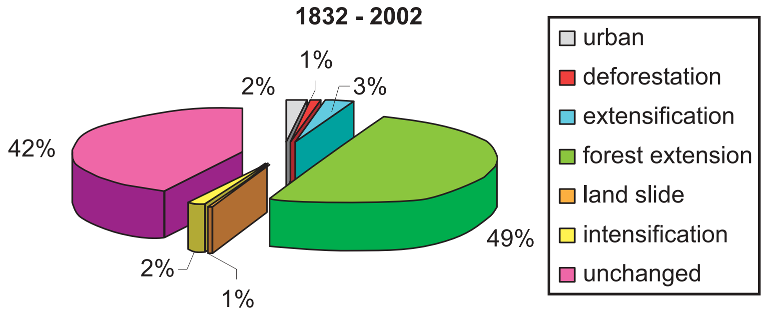

Figure 4. Main processes affecting landscape between 1832–2002.

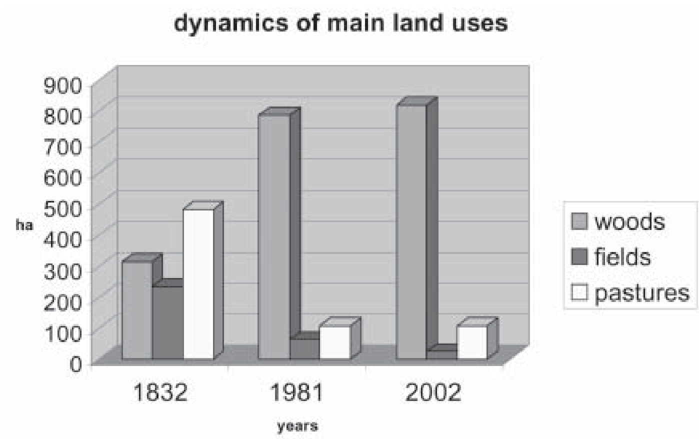

Figure 5. Evolution of the three main land use types between 1832 and 2002.

while the dominance index it is not showing clearly that today we have the presence of larger patches, while in the past there were many small patch as cultivated fields, together with very large patches of forest. Looking at the cadastre of 1832 we can actually see land uses classes with a total extension of only 0.09 ha, this can give a rough idea how complex was this landscape mosaic and how the texture of the landscape and its diversity has been turned into a rather simple pattern dominated by woodlands.

5. Evaluating landscape diversity: the historical index

The use of the indices commonly used in landscape ecology to describe an evolution like the one presented in this case study is not sufficient to analyse a process in which historical factors interrelate with natural dynamics, or to assess strategies suited to consider all the values involved in the conservation of protected areas. For this reason the reflections made

Table 1. Cross tabulation matrix showing land use changes in each class between 1832 and 2002. The matrix allows to follow the evolution of each class, reading from left to right one can see how the original class in 1832 has been transformed in 2002, looking at the cross between columns (year 2002) and lines (year 1832). The total extension of each land use class in the period is indicated at the bottom of columns and lines. The original long list of land use in 1832 has been reclassified into a shorter list to allow a comparison with the very short number of land use in 2002. Colours refer to Figure 4.

		2002													
	L.U. 1832_RECLASS	rocky areas	urban	shrubland	low density*	woods	mixed woods + chestnut	quarry	terraced fields	abandoned terr. fields	landslide	pasture	wood pasture	meadow	**Total**
1832	urban		2,1177			0,6434	0,4213		0,4175	0,0638	0,0439			0,0067	3,7143
	shrubland					1,4775	0,2168								1,6943
	wood	0,7119	0,053		0,5478	28,8729	15,7816	0,2127						0,125	46,3049
	hemp		0,6572			0,3771	0,1063	0,1558	0,0222						1,3186
	chestnut	6,1746	2,0998	0,1578	9,0175	46,2589	193,8046	0,6037	2,3975	0,004	6,5049		0,0014		267,0247
	quarry					0,4612	0,0243								0,4855
	uncultivated		0,1236			6,2681	1,5052			0,311			0,0623		8,2702
	fallow + trees		0,3799				1,5189		0,5836		0,0169				2,4993
	fields	0,0498	3,8561	1,0505	0,5009	54,771	40,8528	0,3375	2,4293	5,0131	0,1546	8,9283	2,2504	0,3132	120,5075
	fields+trees	0,2145	0,8496			3,2443	3,9566	0,0004	0,7887		0,3335				9,3876
	fields + vines	0,2161	4,3364			54,7951	31,0648	1,9057	12,6657		0,1152			0,7664	105,8654
	pasture	63,0465	0,3592	1,494	26,8145	181,9919	27,5117	0,0591	0,0047	0,3289		85,333	0,884		387,8275
	wood pasture	0,9771	1,8939		0,2345	38,6083	15,4618		0,7571	0,1774	0,0299	0,202	0,0591	0,0211	58,4222
	meadow	0,1019	0,1638	0,6375		15,5963	8,211	1,2172		0,3647		6,8088	1,0621	0,1714	34,3347
	wooded meadow		0,2248			2,7957	1,3158	0,1148	0,0205		0,0008				4,4724
	water				0,903	0,476									1,379
	total	71,4924	17,115	3,3398	38,0182	436,6377	341,7535	4,6069	20,0868	6,2629	7,1997	101,2721	4,3193	1,4038	**1053,5081**

* This definition refers to areas with scattered trees and shrubs

Table 2. Structural changes in the landscape mosaic between 1832 and 2002

	1832	2002
Number of patches	618.00	84.00
Average extension (ha)	1.70	12.54
Standard deviation	7.33	41.24
Dominance index (Shannon and Weaver)	2.02	0.84
Hill diversity number	9.00	8.21

inside the working group involved in the research suggested the need to develop an index to evaluate the dynamics of cultural landscape, in order to understand where the most important values to preserve lies in the territory analysed, and to evaluate the risk that some landscape types can disappear (Agnoletti 2002). This index, now applied in the creation of a landscape park and in the Environmental Impact Assesment rules of the Tuscan Regional Government is still a "work in progress", but the application of the first stage of its development to the study area presents interesting results.

The structure of the index is made to evaluate the role of the each land use type still existing in the territory, considering their presence over time and changes in its extension. For this reason there is no evaluation given for land uses not existing anymore, or for those cases as chestnut, today present in the form of mixed woods with chestnut.

According to the Figure 6 most of the "emergencies" of the area lies in the land use class "meadows", whose high index value correctly represents the reduction of this class passed from 54 to 4 ha, followed by wood pastures, fields and pastures. Pastures are still present in the higher side of the mountain, where forest vegetation is extending at a slower rate. The map showing the distribution of the historical index in the study area indicates that the highest values can be found in small areas where meadows, fields and wood pasture are still cultivated today. The area close to the mountain top, dominated by pastures, represents the second zone where an active conservation should be made, while all the woodlands present to day do not represent any kind of emergency, since they are not endangered and their expansion continues.

6. Conclusions

Landscape changes occurred in the last two centuries are mostly due to socio-economic development. In addition to macro-structural modifications due to urban and industrial development which while easily identifiable are not easily controllable, there are changes that go much deeper. Specifically, we refer to changes in the economic fabric, manufacturing trends and working techniques in farming and forestry. These changes have led to the abandonment of traditional working techniques that were linked to local cultures and have triggered processes that substantially alter the structure of the land. Some of the most important effects concern the modification of forest systems that have enormous landscape value (e.g. chestnut orchards) and required continuous work on the part of man to maintain them as well as for the disappearance of fundamental elements (e.g. mixed crops, hedges, tree rows, terraces) that characterized farming. The interruption of rural practices had also impact on the hydrogeological balance because the abandonment of land arrangements that required continuous human presence caused erosions and landslides affecting especially mountain and

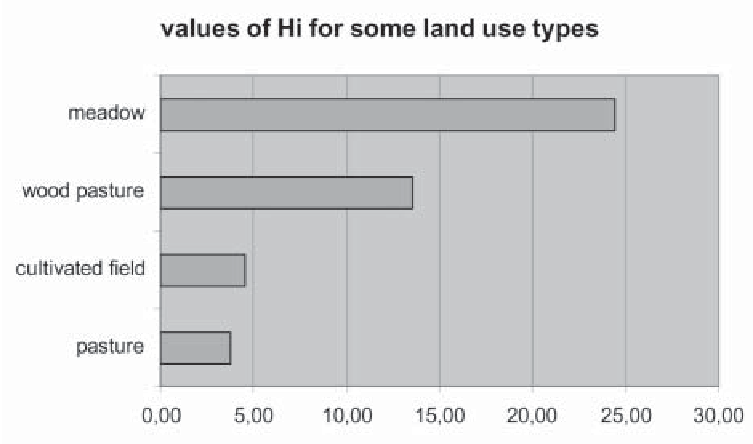

Figure 6. Values of historical index for the most endangered categories of land uses.

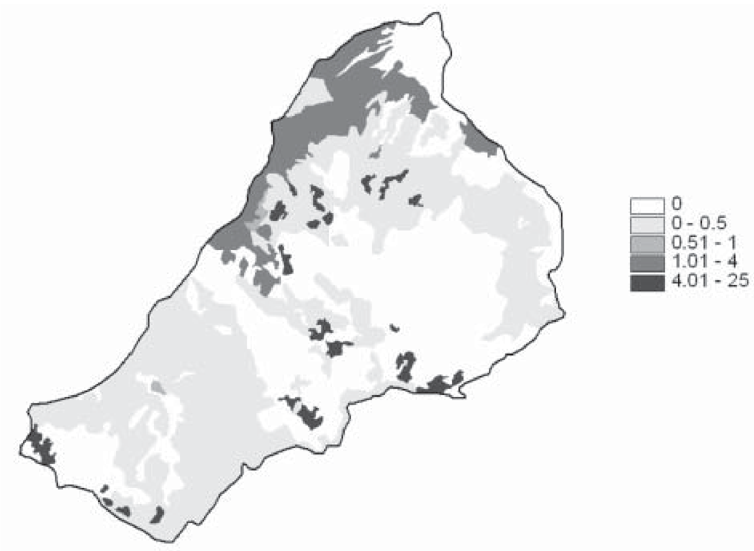

Figure 7. Map of the distribution of historical index in the whole study area.

hilly areas. In this respect the landscape existing in the area until the early Fifties can probably be compared with the ones still surviving in places like eastern European countries where industrialization will soon induce the same process (Angelstam et al. 2003). The protection of the diversity of land uses today is a fundamental point in a serious conservation program that cannot but be tied to a different model for understanding the concept of sustainable development.

The study area presented can be considered a typical case study to evaluate the effect of long term human influence on the environment, but also a test to evaluate the effectiveness of the European directives for nature conservation applied in Italy and in Tuscany. Despite the reports on the state of the environment (Calistri 2001) showing very positive evaluation of biodiversity, it can be questioned that these analysis are not representing the whole issue in biodiversity conservation and the management of protected areas. As a matter of fact the study shows that we might perhaps agree on the positive role of increased forest extension for species diversity, although we don't know how many species introduced by the man in this area since roman times for farming are not existing anymore. Nevertheless, we can surely note that since 1832 there has been a loss of 86% of diversity in landscape patches, with the severe decrease of many land uses linked to pastures, fields and meadows and a consequent reduction of the heterogeneity of the landscape, now substituted by a continuous forest layer. This also means the reduction of many valuable habitats represented by each landscape patch, not only contributing to the overall biodiversity but also of utmost importance for the fauna. This process is comparable to the general trend reported for Italy in the last 80 years, where the forest land has increased almost by the double, depending on the different statistical method adopted (Agnoletti 2003). This loss of diversity is particularly important for the Italian environment because is not the diversity of species the most important feature of Italy, quite relevant however in the European context, but rather the diversity of spaces created by the man in centuries of rural practice, also introducing many species originally not present in the Italian territory. While in some area the increase of forest cover can play a positive role in many cases this trend is not sustainable, not only for biodiversity but also for the conservation of landscape resources, in Tuscany forest have increased by almost 70% and now cover approximately the half of the territory.

The reasons of the failure in evaluating diversity at landscape level can be found in the structures of European directives, in the way they are applied in Italy and in the management policies carried out in protected areas. The "habitat directive", acknowledged by Tuscany with the law n. 56-2000, is enclosing a list of protected habitats, in which many habitats referring to traditional land use types are not considered (as chestnut orchards), while mixed forest with the dominance of chestnut are included. This means that a secondary forest, representing an ecological transition intended to evolve into a mixed stand, is be preserved by law, while a habitat with a very high ecological and historical value is not protected. Another reason for the reduced attention in landscape diversity is due to the philosophical background of management in parks and protected areas, where the extension of forest land and renaturalization is often seen as the best policy to carry out. The results of this study suggest that a more objective evaluation of methods and goals in nature conservation should be adopted not only in Italy, but also in other countries where cultural landscapes are endangered.

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Maintaining Forest Biodiversity in Actual Landscapes – European Gradients in History and Governance Systems as a "Landscape Lab"

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Abstract

Maintaining forest biodiversity by combining protection, management and restoration in forest and woodland landscapes is one of the components of ecological sustainability. Succeeding with this can even be viewed as an acid test of sustainability as a whole. The principle of sustainable forest management has stimulated a proliferation of a number of criteria and indicators. However, to achieve ecological sustainability, it is vital that the monitoring of a suite of relevant indicators are compared with performance targets to assess both status and, if repeated, the trends in actual landscapes. We first describe how traditional measurement tools for describing wood resources need to be complemented by monitoring of the elements of biodiversity including species, habitats and functions at multiple spatial scales. Second, we provide examples of empirical non-linear relationships between presence and fitness of species' populations and different levels of anthropogenic change in their respective habitats at different spatial scales, and argue that this can be used to formulate science-based performance targets for indicators. Finally, using the results from monitoring and with relevant performance targets, it is possible to make assessments of the status of a certain criterion, such as biodiversity. Examples of practical assessment tools such as regional gap analysis and habitat suitability models are presented for strategic and tactic planning, respectively, of operational management for protection, management and re-creation of different elements of biodiversity. Additionally, the need to assure communication with iterated feedback of the results of assessments to managers and policy-makers is discussed. Aiming at bridging the gaps among the fields of policy, science and management, we finally stress the need for international co-operation. We suggest the establishment of a network of case studies in the form of "landscape laboratories" in gradients of forest alteration and different governance systems together with managers of forests and woodland representing

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different trajectories towards SFM. Watersheds, Model Forests, Biosphere Reserves and National Park are four relevant concepts to be employed.

Introduction

Sustainable forest management (SFM) represents a vision for the use of forests based on satisfying ecological, economic and social values (Rametsteiner and Mayer 2004). In response to unsustainable use of products and services several phases in the ongoing development of SFM can be identified (Davis et al. 2001). The first has a very long history and is based on sustained yield of wood products where humans dominate nature (Schuler 1998). Second, there is at present in Europe a phase based on multiple use sustaining primarily wood production, but also with attempts to accommodate new issues such as biodiversity and multiple use (Farrell et al. 2000). Third, a future phase is envisioned with sustainable social-ecological systems inspired by and maintaining valued natural or cultural processes in which humans and nature co-exist (Berkes et al. 2003).

The current starting point for trajectories towards the SFM vision, however, varies considerably among countries and regions with different socio-economic settings and ecosystems (Angelstam et al. 2004a). Sweden, Switzerland and Russia provide three contrasting European examples where the focus has been and is on quite different SFM criteria. In Sweden, where forests have used for intensive management for sustained yield of wood for a long time, biodiversity interpreted as the maintenance of viable populations of naturally occurring species has been a major driver of changes towards SFM during the past decades (Angelstam 2003). However, currently social forestry issues are also appearing. In the revision of the current forest policy that will be completed by the end of 2005 ecological, economic and socio-cultural criteria are considered. Switzerland's steep terrain has for long promoted management for protective functions of the conifer-dominated mountain forests, and only recently with an additional focus on biodiversity (Kräuchi et al. 2000; Neet and Bolliger 2004). Finally, in remote parts of Russia such as in the Komi Republic in northeasternmost Europe, large-scale logging of forests started only recently, and large intact forest areas still remain (Yaroshenko et al. 2001). Here a major current challenge is to use not previously managed forest landscapes for the development of human welfare in a sustainable manner (Pollard 2003), but also to maintain the functionality of the last remaining large intact forest areas.

The international and national policy arenas, the forest sector, non-governmental organisations and scientists are the major actors trying to develop and interpret international and national policies on sustainable development in forests. In Europe the Ministerial Conference for the Protection of Forests in Europe (MCPFE) has derived a reasonably complete set of indicators defining different SFM criteria at the policy level, including biodiversity and forest health (Rametsteiner and Mayer 2004). Additionally, in analogy with the discussion on critical load of anthropogenic pollution (Nilsson and Grennfelt 1988), different actors presently also ask for performance targets for biodiversity indicators, and tools allowing assessment of the degree to which ecological sustainability is actually achieved in actual landscapes (e.g. Djurberg et al. 2004).

The MCPFE's criteria regarding ecological dimensions of SFM can ultimately be viewed as proxies of the natural capital (Costanza et al. 1991). Consequently, the maintenance of biodiversity and resulting products and services is a prerequisite for satisfying the economic and social benefits of forests and woodland including both terrestrial and aquatic systems (Wiens 2002). To support this notion we give an overview of some recent empirical studies of non-linear responses of specialised species to habitat loss that can be described using MCPFE indicators. We then present tools for making strategic and tactical assessments of the functionality of habitat networks. Finally, we discuss the need to overcome barriers for integrated iterative assessment in the spirit of active adaptive management (Lee 1993), and suggest the introduction of active adaptive management teams using landscapes as laboratories for transdisciplinary research and development (Boutin et al. 2002; Jakobsen et al. 2004). In this context Europe is viewed as a grand arena for replicated landscape scale studies of social-ecological systems with different history and governance systems.

Hierarchical assessment of biodiversity

Defining forest biodiversity

The natural potential vegetation of a large part of the terrestrial ecosystems in Europe is forest (Mayer 1984). Forest biodiversity is made up by species, habitat structures and processes found in ecosystems with trees, and can therefore be maintained in both natural forests and in remnants of pre-industrial cultural woodland landscapes.

Policies related to biodiversity of European forests and woodland make explicit reference to the concept of naturalness (e.g. Rametsteiner and Mayer 2004). In spite of the ambiguity of this concept (Egan and Howell 2001), it is obvious that forest biodiversity indicators should represent elements found in naturally dynamic forests (Peterken 1996; Angelstam and Dönz-Breuss 2004). Additionally, the maintenance of ecological values found in pre-industrial cultural landscapes is encouraged (Kirby and Watkins 1998). Reference areas for both visions are characterised by the presence of habitat elements such as dead wood, large old trees, a diversity of tree species, old-growth stands and the ecological integrity of aquatic systems (e.g. Rackham and Moody 1996; Karr and Chu 1999; Karr 2000; Shorohova and Tetioukhin 2004). While ‘laissez-faire’ management usually can enhance the naturalness vision, the maintenance of cultural landscapes requires a certain amount of social and cultural capital (Pierce Colfer and Byron 2001). In other words the maintenance of forest biodiversity encompasses two sets of broad visions depending on the history of forests and woodland in the actual landscape (e.g. Peterken 1996; Agnoletti 2000). The development of SFM should reflect both these visions.

The cover and types of forests and woodland are dynamic, including both degradation and restoration related to socio-economic changes (Nilsson et al. 1992). Consequently, monitoring and assessment of SFM should not only encompass the area covered by forest and woodland at present, but rather a geographically contiguous units representing actual landscapes or ideally natural units such as watersheds. Such areas, hereafter called landscapes, have a wide range of implementing actors representing different institutions ranging from the forest and wood industry, small private landowners and commons to different public interests. However, even if the forest cover is constant, the relative proportion of different governance systems varies considerably among landscapes, regions and countries (e.g. Angelstam and Pettersson 1997). This should be expected to have strong effects on the ways of and extent to which different policies can be implemented using different management and planning tools (Fries et al. 1998; Angelstam et al. 2003a).

Monitoring indicators of biodiversity

Monitoring the elements of forest biodiversity need to be made at multiple spatial scales. At the international and national policy levels, indicators aim at communicating the status and trends of biodiversity to policy-makers and the general public. However, to allow effective operations, indicators should also be developed and applied at the level of forest management units. Such practical indicators need to be adapted to the local conditions and resources available to different end users ranging from corporate companies to the owners of small non-industrial private forests.

Because it is impossible to measure all aspects of biodiversity, there is a need for cost-efficient monitoring tools. At the policy level, international reporting is based on individual countries providing data like those collected in national forest inventories. In general, however, such programmes provide neither sufficient data on compositional (e.g. occurrence of specialised species), structural (e.g. quality of habitat networks), nor functional elements of biodiversity (e.g. ecosystem processes) to allow comprehensive monitoring (Angelstam et al. 2004b). Conversely, at the management unit level, the development of comprehensive biodiversity monitoring systems is still in its infancy. Based on the MCPFE indicators, attempts to build a system for biodiversity monitoring based on the composition, structure and function of natural forest ecosystems have been proposed (e.g. Larsson et al. 2001). Evaluations of this system in European land use history gradients showed promising results at the scale of both stands and landscapes (Puumalainen 2001; Angelstam and Dönz-Breuss 2004; Angelstam et al. 2004c). Additionally, there is a need for continuous evaluation both of the scientific validity of indicators and of the degree to which the results from indicator systems can be interpreted and communicated to stakeholders at all relevant levels (Uliczka et al. 2004). These factors probably explain why real-life applications of such monitoring systems are still rare.

Performance targets

Habitat loss is a major factor affecting directly or indirectly the global decline of biodiversity (Heywood 1995). With a biodiversity conservation perspective, the evaluation of hypotheses claiming species-specific ‘extinction thresholds’, defined as the minimum amount of habitat required for the persistence of species in the landscape, is an urgent task (Mönkkönen and Reunanen 1999; Muradian 2001; Fahrig 2002). Appearing empirical evidence show that human-driven landscape changes have resulted in the trespassing of such critical levels of habitat loss. This applies to structural elements such as dead wood (Bütler et al. 2004a, b), large habitat patches (Mykrä et al. 2000) and reduced amount of certain tree species (Jansson and Angelstam 1999). In Europe an obvious consequence of this is that countries and regions with a lower intensity and shorter history of forest use still host populations of species specialising on natural forest structures, while other do not (Mikusinski and Angelstam 1998; Puumalainen 2001; Angelstam et al. 2004d).

Even though research on thresholds remains in its infancy, ecologically-based targets inspired by the insight that such thresholds exist, could be used to postulate cost- and conservation-efficient management and conservation performance targets. The concept of critical load provides an example of such an approach (Nilsson and Grennfelt 1988). It was coined to define the amount of acidic deposition that the most sensitive elements in an ecosystem could tolerate without significant damage. For the maintenance of biodiversity, critical loss of, for example, habitat structures in relation to the range of natural variation, would be an analogous concept. We stress, however, the need for explicitly recognising uncertainty and, rather than proposing target numbers, there should be a focus on probabilistic targets defined using a variety of indicators, and on the associated “zones of risk” (e.g. Muradian 2001; Phillis and Andriantiatsaholiniana 2001), which will vary among different groups of species and ecoregions.

There is, thus, a need to perform more systematic studies of species' responses to habitat loss. A general procedure for identifying thresholds to be used in the determination of conservation targets in forests was proposed by Angelstam et al. (2004e): (1) Stratify the forests into broad cover types as a function of their natural disturbance regimes. (2) Describe the historical spread of different anthropogenic impacts in the forest region of concern that moved the system away from the reference conditions of naturalness or pre-industrial cultural landscapes. (3) Identify appropriate response variables (e.g. focal species, functional groups or ecosystem processes) that are affected by habitat loss and fragmentation. (4) For each forest type identified in step 1, combine steps 2 and 3 to look for the presence of non-linear responses and to identify zones of risk and uncertainty. (5) Identify the "currencies" (i.e. species, habitats, and processes) which are both relevant and possible to communicate to stakeholders. (6) Combine information from different indicators selected.

From the recent review by Angelstam et al. (2004f) it is evident that there is large variation in the landscape scale habitat requirement of different species depending on the scale and level of ambition for their conservation. Focusing on specialised animals of forest and woodland, the landscape-scale extinction thresholds for 17 species (birds, mammals and one insect) ranged from 10 to 50%, with a mean of 19%. This is consistent with Andrén's (1994) review suggesting that 10–30% of habitat in a landscape is needed to maintain local populations.

Assessment of status and trends

Management of sustainable wood production as well as management for protection, management and restoration of the elements of biodiversity require planning at multiple scales. The approach used in most planning systems for large-scale forestry is hierarchical within a forest management unit (FMU) (Davis et al. 2001). The planning problem is usually divided into three sub-processes: strategic, tactical and operational. Strategic planning means to decide on long-term goals covering an entire rotation and tactical planning to select among different alternatives based on the strategic goals, but on a shorter time horizon. Operational planning involves determining the actual operations. The same logic can be used to build a toolbox of analytic tools for the assessment of the structural elements of biodiversity being the focus in conservation, management, and restoration (Angelstam et al. 2003a).

At the strategic level regional gap analysis is a tool for assessing the extent to which environmental policies succeed in maintaining biodiversity by protection, management and restoration of habitats (Scott et al. 1996). Originally developed in the USA, gap analyses have been used in terrestrial systems to increase society's awareness about conservation needs and to guide the practical implementation of such policies. The rationale for focusing on habitat (i.e. structural elements of biodiversity) is that it serves as a proxy for the maintenance of viable populations of species, vital ecosystem processes and resilience to external disturbance (e.g. Karr 2000). Originally gap analyses focused on representation i.e., that the different types of conservation areas should reflect the natural composition of different ecosystems (Scott et al. 1996). Angelstam and Andersson (2001) and Lõhmus et al. (2004) developed the idea further for Sweden and Estonia, respectively, by combining measurements of the habitat area with information about thresholds for the amount and quality of habitats needed to maintain viable populations within an ecoregion (Table 1).

When a gap analysis has been performed within a particular ecoregion, the forest types for which area gaps have been identified also need to be evaluated as to the extent to which they actually provide functional habitat for the specialised focal species. One approach to evaluate the functionality of existing networks of patches of different forest types is habitat suitability

Table 1. The 'ABC' of regional gap analysis for strategic conservation planning. Given a policy which can be interpreted scientifically, like maintaining viable populations of naturally occurring species, reference conditions (A) such as found in naturally dynamic forests or pre-industrial cultural landscapes, can be quantified for different vegetation types. By comparing the present situation (B) with A for different types of forest and wooded grassland, analyses of representation can be made. Finally, with quantitative policy targets at regional level (C), expressed as a proportion of A, long-term goals for conservation or restoration can be formulated and compared with B. This allows the identification of area gaps for a certain type of vegetation within a region. Next, to assure functional connectivity of the total area, spatially explicit analyses need to be done for the tactical decisions regarding protection, management and restoration at the landscape scale.

Code	Explanation
A	Reference condition
B	The present situation
C	Science-based policy target
A-B	Representation
A*C	Long-term goal for the amount of habitat needed
B - (A*C)	Area gap (if the value is negative)

modelling (e.g., Scott et al. 2002; Angelstam 2003b, 2004d). This means combining spatially explicit land cover data with quantitative knowledge about the requirements of specialised species, and producing spatially explicit maps describing the probability that a species is found in a landscape. With adequate quantitative data defining habitat variables, and parameter values for a suite of particular focal species carefully selected to represent all forest types of concern, a models can be built to assess the functionality of different habitat networks. This requires quantitative information on the habitat requirements of the species at different spatial scales. In general, a habitat model for a given species should build on the following variables: land cover type(s) constituting habitat, habitat patch size, landscape-scale proportion of suitable habitat, and habitat duration (Angelstam et al. 2004d). Using, for example, neighbourhood analysis techniques in Geographic Information Systems, the functionality of the network of each representative habitat (one or several land cover types) can be evaluated. Because a landscape usually contains a range of types of forest vegetation, a suite of species need to be selected and modelled (Root et al. 2002; Roberge and Angelstam 2004)

The procedure suggested above provides a general basis for the assessment and subsequent spatially explicit planning of habitat networks. The development of practical tools using focal species is, however, subject to uncertainty depending on the knowledge about the different parameters included in the models. Another factor influencing the development of practical tools is the thematic and spatial resolution of the land cover data available to the planner (cf. Young and Sanchez-Azofeifa 2004). For example, depicting the habitat of species dependent on dead wood (e.g. many species of woodpeckers, beetles, and wood-decay fungi) require spatially explicit data on the occurrence of this resource across the landscape. Such data is not currently available from forest management maps or satellite images, and therefore additional ancillary data needs to be collected in the field. Until such data become available, surrogate measures such as vicinity to roads as a proxy for the amount of dead wood could be used (Bütler et al. 2004b).

Ideally, focal species should be chosen among the most demanding species for a range of landscape attributes (Lambeck 1997). Since the most demanding species vary among habitats and scales, a given suite of focal species should include representatives from a number of different taxa with different ecologies or functional groups (e.g. Angelstam 1998; Karr and Chu

1999; Nilsson et al. 2001). Finally, each model should be validated in order to test how reliably one can predict occurrences of the focal species in real-world landscapes (Scott et al. 2002).

An example of a successful attempt to apply habitat suitability modelling in operational forest management is provided in Suchant and Braunisch (2004). Using capercaillie *Tetrao urogallus*, a species of high conservation value and of special concern to major actors in biodiversity conservation in Germany, the authors modelled the habitat conditions for the occurrence of the capercaillie in several analytical steps at different temporal and spatial scales. Based on this, management target values were derived and integrated into an operational habitat management model in order to assess habitat suitability. This study is a good example of how wildlife research can be linked to practical habitat management.

Discussion

The need for spatially explicit forest management

Landscapes are not constant (Bengtsson et al. 2003). The variation among different European regions in the trajectories of the development towards the SFM vision is a reflection of this. Because most of Europe's landscapes have an origin as forests or wooded grasslands, forests and forestry must be seen in a landscape perspective (Farrell et al. 2000). Current driving forces affecting European landscapes include the macroeconomic development affecting human population migration from the periphery to centre, the active expansion of the transport infrastructure, and the energy sector. Climatic change is another, but less predictable factor. As shown in the following three examples the effects of different elements of forest biodiversity through changes in the land cover and the spatial configuration are complex.

The implementation of the EU Habitats Directive (Anon. 1992) by establishing a network of conservation areas with a favourable conservation status is one example. The appearing knowledge about thresholds for the amount of habitat viable populations of species need, i.e. reflecting the resources they require, has clear implications for biodiversity management. Even if still under development (Anon. 2003; Halahan and May 2003), it is fair to state that the maintenance of the species listed in the EU Birds and Habitats Directive (Anon. 1979, 1992) using the so called Natura 2000 sites requires functional networks of suitable habitats. Depending on the quality of the conservation areas themselves, and of the surrounding landscape, the amount of area that needs to be set aside will vary. At any rate a toolbox consisting of protection, management and restoration of habitats will be required (Hunter 1999).

Establishment of functional habitat networks may both suffer and benefit from the current land cover changes in Europe. Abandonment of agricultural land in the periphery of economic development leads to increased cover and improved connectivity for forest species (Angelstam et al. 2003c). Early successional stages of deciduous forest in one example in Sweden (Mikusinski et al. 2003). Currently a similar development is taking place at a grand scale in the Baltic States and parts of Poland (e.g. Stålnacke et al. 2003). This can be expected to have positive effects on specialised and area-demanding species. Simultaneously, however, there are advanced plans to extend the transport infrastructure by building highways to stimulate the economic development. Dissection of large and functionally connected forest tracts by transport infrastructure has had well documented negative effects on the efforts to maintain species with large area requirements such as large carnivores and herbivores (Forman et al. 2003). The effects on the future functionality of habitat networks – or "green infrastructure" cannot be understood and planned without spatially explicit analyses at the scale of landscape within larger regions.

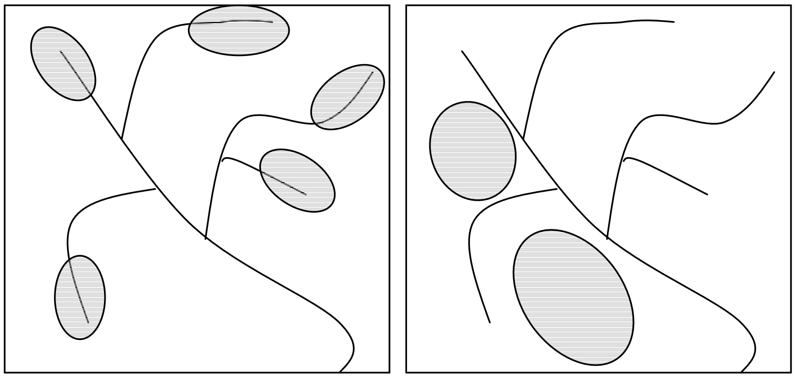

Figure 1. Illustration of how the spatial distribution of retention of deciduous trees and dead wood in a forest landscape can produce very different effects when restoring the ecological integrity of the aquatic system. To the left the litter fall and blow-down of trees left as retention trees and riparian corridors along the first order streams will improve the situation for aquatic species by providing more nutrient input and structural diversity in streams while to the right there will be virtually no effect in the stream.

The positive effect of land cover on aquatic systems is another example (Stålnacke et al. 2003). Wiens (2002) used the term aquatic landscapes to stress this. Interestingly enough, the EC Water Framework Directive (Anon. 2000) has recently reinforced a drainage basin perspective on water issues and aquatic biodiversity. In order to maintain and restore surface and groundwater to “Good Ecological Status” dead wood is a key structure in stream order 1–4 (Marcus et al. 2002). Degerman et al. (2004) studied the relationship between brown trout (*Salmo trutta*) and dead wood in Swedish streams and found a positive relationship between the abundance and size of trout and dead wood. The gap between the present amount of dead wood and the amount found in reference landscape is, however, about 1–2 orders of magnitude (Liljaniemi et al. 2002). With limited resources to leave harvestable wood in the forest to restore the quality of aquatic systems, the spatial effects of retention on the functionality will be important to stream ecosystems (Figure 1).

Finally, because different forest vegetation types host different species, the maintenance of functional networks for species with different specialisations should be seen as separate and not necessarily overlapping green infrastructures. The coniferous and deciduous component in a landscape can serve as an example (Figure 2). While the coniferous forest is managed and forms a stable patch dynamics for at least species not requiring old-growth elements, the deciduous forest originates from abandoned wooded grasslands, and around the lake from the lowering of the lake level in the late 19th century (Rydin and Borgegård 1988, 1991). In contrast to the coniferous forest, the deciduous forest is the result of a number of historical events driven by socio-economic change (Mikusinski et al. 2003). To maintain species of the deciduous forest in the long term, the deciduous component needs to be restored in what is now coniferous forest. However, dense populations of moose and deer severely hamper the recruitment of at least the most important tree species for specialised species (e.g., the genera *Populus*, *Salix* and *Sorbus*) (Angelstam et al. 2000). Another barrier is the poor integration of the management of trees in forest and agricultural landscapes.

Towards integrated and transdisciplinary approaches

Science develops indicators because they are required for the policy implementation process. The MCPFE criteria and indicators focus on the state of a system. However, such results need

Figure 2. Forests usually host multiple green infrastructures. The lake Hjälmaren watershed in south-central Sweden with coniferous (top) and deciduous (bottom) forest. The city to the left of the lake (grey area) is the city Örebro (15°10’ E, 59°20’ N).

to be put into the context of continuous evolution of policies. Indicators should thus be seen as describing the success of a policy implementation feedback loop that begins with a Pressure leading to a State and resulting in a Response (the PSR model; Rapport and Friend 1979). The PSR model, and subsequent elaboration of it, has been successful in helping structure the use of indicators (Linser 2001). The socio-economic context and the associated governance systems drive the state. Based on monitoring of the state of productive functions and biodiversity, and performance targets for the different indicators, assessments can be made. If the outcome of such assessments indicate the need for active response resulting in gradual modification of the state of the landscapes in the desired direction, management must be planned and implemented (Figure 3).

There is a growing insight that there are complex interactions between the parts of different ecosystems and institutions, which require transdisciplinary landscape-scale approaches (Rabeni and Sowa 2002; Schneider et al. 2002; Lazdinis and Angelstam 2004). In Europe the EC Water Framework Directive (Anon. 2000) stresses this. In spite of the presence of relevant tools from the natural and social sciences (Angelstam et al. 2003a; Bryman 2001), effective use of them in a transdisciplinary fashion to facilitate the implementation of sustainable development policies is rare in the real world (Boutin et al. 2002; Duinker and Trevisan 2003). Apparently, working across disciplines in landscape analyses is a major challenge. In a comparison of two case studies Jakobsen et al. (2004) revealed a set of similar

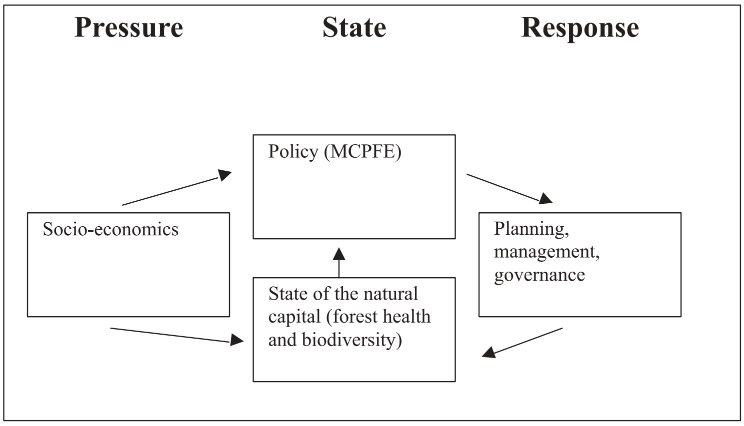

Figure 3. Pressure exerted by socio-economic factors, state indicators of the ecological dimension of sustainability (forest health and biodiversity) in the context of criteria and indicators of sustainable forest management according to MCFPE (Rametsteiner and Mayer 2004), and the response in terms of planning and management under different governance systems.

individual-based, group-based and organisation culture-based barriers. However, even if they proposed a number of recommendations to scientists across disciplines, the limited number of case studies precludes thorough analyses of the effects of ecological, institutional and cultural contexts on both barriers and facilitators to bridge them.

The "Landscape Lab" approach

Even with a widespread insight that spatial forestry encompassing whole landscapes is necessary, the variation in ownership patterns and governance systems may provide both barriers and bridges to the application of spatially explicit assessment and planning. A major challenge is to achieve integration among actors. Researchers and managers accomplish most of their work in isolation, and then present their results to decision-makers. There are hence a number of barriers, in particular when attempting to apply a landscape approach to the conservation of biodiversity (e.g. Gutzwiller 2002). Using landscapes as laboratories is one approach for transdisciplinary research. To describe this Kohler (2002: 212) used the concept 'practices of place' whereby it is "…the arrangement of spatial elements that provides critical evidence of relations between creatures and their environment…". Places are thus to the field ecologist what experimental set-ups are to laboratories. Co-ordinated case studies based on the idea of 'Practices of place' can thus be designed stratified in replicated land use history gradients. This can be made both in time and space. For example, the historical occurrence of species dependent on dead wood can be compared with the decline of dead wood over time (e.g. Linder and Östlund 1998). Additionally, the presence today can be made in landscapes located forest history gradients (e.g. Angelstam and Dönz-Breuss 2004; Angelstam et al. 2004c; Shorohova and Tetioukhin 2004). This is consistent with the combination of case studies and quantitative data termed triangulation used in social sciences (Bryman 2001). This approach may actually make it possible to "look into the future" to see what new pressures on forest ecosystems which can be expected. The gradient of commercial thinning is one example. While this has a very long tradition in Central Europe, this forest history phase reached northern Sweden in the 1960s and is now entering Russia.

Table 2. Europe as a 'landscape lab'. Idealised matrix for selecting case studies with forest and woodland systems having different abilities of hosting forest biodiversity using on the one hand the gradient from cleared forest to a remnants representing biodiversity maintenance visions of "naturalness" and "pre-industrial cultural landscapes", and governance systems on the other.

Forest and woodland system	Governance system			
	Non-Industrial Private	Commercial Company	Public management	Protected
Urban green space	Suburban residence area [1]	Golf course [2]	Park [3]	Urban reserve [4]
Wooded grassland	Agricultural sub-sidies and land abandonment [5]	Ecological meat production [6]	Urban forestry [7]	Recreation forest [8]
Plantations on formerly cleared land	Energy forests [9]	Eucalypt forests in Portugal [10]	Exotic conifer plantations in Scotland [11]	Habitat restoration in Western Europe [12]
Altered tree species composition	Southern Fennoscandia [13]	Intensive forest management for fiber and wood [14]	Transformation of coniferous to deciduous forest [15]	Removal of undesired tree species [16]
Semi-natural managed	Baltic States [17]	Managed forests in Fennoscandia [18]	State-owned forests in Fennoscandia [19]	Protection forests in mountains [20]
Benchmark/ reference	Pre-industrial cultural landscapes [21]	Woodland Key Habitats [22]	Remote parts of Russia [23]	Russian Strict State Reserves [24]

1: Sandström et al. (in press); 2: Tanner and Gange (2004); 3, 4: Sandström et al. (in press); 5: Carey et al. (2003), Stålnacke et al. (2003); 6: Kumm (2002); 7: Konijnendijk (2003), Germann-Chiari and Seeland (2004); 8: Roovers et al. (2002); 9: Richardson et al. (2002); 10: Reed et al. (2003); 11: Summers et al. (1999); 12: Wallis de Vries (1995); 13: Mikusinski et al. (2003); 14: Farrell et al. (2000); 15: Pietsch and Hasenauer (2002); 16: Larsson and Simonsson (2003); 17: Kurlavicius et al. (2004); 18: Raivio et al. (2001); 19: Korhonen et al. (1998); 20: Neet and Bolliger (2004); 21: Rackham and Moody (1996); 22: Hansson (2001); 23: Yaroshenko et al. (2001); 24: Angelstam et al. (2004e).

Even if Europe is becoming more and more integrated in a political sense, the states of the forests and woodland ecosystems are still highly variable in different regions and countries, and range from large intact natural areas to remnants of cultural woodlands. Additionally there is considerable variation as determined by the type of ownership and resulting governance system. The ecological and social dimensions of landscapes should form the basis for a design for communication, research and development towards SFM in case studies representing gradients in social-ecological systems (Table 2). At the pan-European level this matrix would then cover the gradients in regional macroeconomic development, rural-urban transitions and with different sets of stakeholders involve in the management of biodiversity and ecosystem integrity.

To promote this idea we encourage the development of case studies not only as a research tool, but also as arenas for demonstration of bridges to deal with implementation obstacles. One approach is the Canadian Model Forest network, which together forms a partnership between individuals and organisations sharing the common goal of sustainable forest management (see www.modelforest.net). Such a network of forest management units consisting of actual landscapes with their characteristic ecosystems, actors and economic activities can be used as the sites for spreading good examples. Biosphere reserves, national parks and watersheds represent other similar approaches that should be compared and evaluated.

Ideally, active adaptive management teams (Boutin et al. 2002) should be formed. This means that at the level of the actual case study, facilitation of participatory processes with stakeholders including researchers, land managers and policy-makers should lead to joint decisions and responsibilities toward the success or failure of the strategy that they adopt for dealing with different issues. To put this bottom-up reflexive iterative procedure into action and to design management applications, we suggest the development of an international network of active adaptive management teams. This network should be charged with testing different approaches to the management of forest ecosystems that will ensure that biodiversity is restored in areas where it has been lost and maintained where forestry intensification has yet to occur.

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Results from the Project BioAssess – Relation Between Remote Sensing and Terrestrial Derived Biodiversity Indicators

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Abstract

The overall objective of this research was to evaluate the potential of different indicators extracted from remote sensing data to describe and monitor species richness. The different floristic and faunistic data sampled within the BioAssess project provided reference data to assess the indicator value of remote sensing based indices. The sampling of species diversity was carried out in so called land use units (LUUs) along a gradient from old-growth forest to intensive agriculture in eight European countries. In these LUUs the landscape composition was assessed based on remote sensing data. A fused Landsat-IRS image was selected as standard dataset to guarantee the comparability of the results. A hierarchic classification system was defined based on the CORINE database up to four hierarchical levels. Extracted classes from visual interpretation and segmentation-based classification were used to quantify the land use intensity gradient and to calculate landscape indices. In addition, the NDVI was derived from the original Landsat image, and the mean as well as standard deviation values were calculated. For one super test site besides Landsat TM and IRS-1C pan data also Quickbird images and aerial photographs were used. For this test site original pixel values and image grey value derivatives like NDVI, tasselled cap, first and second order texture information, filters and focal algorithms were derived from all aforementioned datasets in addition to the calculation of the landscape indices. Furthermore, a digital elevation model (DEM) in 25 m resolution and a digital surface model (DSM) in 1 m spatial resolution was included in the study using information like slope, aspect, curvature, and texture. The calculated landscape metrics, the direct grey values and described derivatives, as well as the different elevation data were then related to species richness.

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Landscape indices derived from visual interpretation of the Quickbird image correlated the most to species richness of woody plants and birds, while grey value derivatives of the Landsat-IRS image showed the highest correlation to carabid species richness.

Keywords: *biodiversity monitoring; remote sensing; filter techniques segmentation based classification; e-cognition; scale space theory; landscape indices.*

Introduction

The assessment and monitoring of biodiversity is a topic of increasing interest throughout the world, as loss in biodiversity has been identified as a major global environmental problem. The EU and all its Member States are parties to the Convention on Biological Diversity adopted at the Rio de Janeiro 'Earth Summit' in June 1992.

Since then a need to monitor changes in biodiversity has been especially recognised and is explicitly included in, amongst other places, the Convention on Biological Diversity (CBD) as well as the EU Biodiversity Strategy. The CBD requires development of indicators to monitor the status and trends of biological diversity. However, a monitoring programme to assess changes in all components of biodiversity is clearly impossible because of the high number of species present in any place and the number of experts that would be involved. Thus scientists are increasingly expressing the need for indicators of biodiversity and other methods of 'rapid biodiversity assessment to assess the overall biodiversity (Oliver and Beattie 1993). An ideal indicator will accurately reflect changes in most components of biodiversity. There is an urgent need for indicators that reflect those components of biodiversity that are particularly threatened or valued. An ideal indicator of biodiversity should also be one that provides an early warning of changes in biodiversity, particularly in relation to possible threats to biodiversity. Some scientists have suggested that single groups of plants and animals may serve as indicators for overall biodiversity. However, surveys (Prendergast et al. 1993) show highly variable responses of different groups to a disturbance. Therefore, reliance on a single indicator would give a poor measure of overall biodiversity. To overcome this problem, authors have proposed 'predictor sets' of taxa (Stork 1995).

All discussion on biodiversity monitoring is still mainly linked to species diversity. Nevertheless biodiversity is much more than species diversity and from a landscape ecology point of view landscape structures itself are a component of biodiversity (Jedicke 2001). This is also supported by the Convention on Biological Diversity which explicitly includes the diversity of ecosystems. According to Jedicke (2001) the assessment of ecodiversity (landscape diversity) would be a more adequate approach to describe the biological diversity because ecodiversity comprises biodiversity and geodiversity. Many investigations proofed that geodiversity and landcover or land use pattern have a close link to species diversity and abundance (Jedicke 2001). In addition Naveh et al (1998) believes that only through ecodiversity (landscape diversity) the close interrelation between biological diversity, ecological heterogenity and cultural diversity can be expressed. Especially the inclusion of human influence on biodiversity is an advantage of assessing ecodiversity because it is doubtless that long time interactions between man and nature influenced biological diversity decisively. Leser (1997) and Nagel (1998) highlight that diversity of landscapes reflect effects of interactions between abiotic and biotic systems.

If species diversity and species abundance is related to the spatial structure of vegetation complexes then the assessment of vegetation complexes and their spatial distribution seems to be a useful indicator for the assessment of biodiversity. Remote sensing is one data type which can substantially provide information on vegetation. In this article the question how

Figure 1. Overview of BioAssess partner and partner countries.

remote sensing data can be exploited for biodiversity assessment is highlighted. The exploitation of remote sensing data and the linked questions will be discussed on the basis of a research project called BioAssess which has its focus on the assessment and monitoring of biodiversity based on different indicator species and remote sensing data.

BioAssess – biodiversity monitoring based on linked information from remote sensing data and species indicators

In the frame of a European project Biodiversity Assessment Tools (BioAssess) under the Global Change, Climate and Biodiversity Key Action of the Energy Environment and Sustainable Development Programme, a method for rapid assessment of biodiversity was developed on a European level. Test sites in different biogeographical regions have been selected (Finland, Ireland, UK, Hungary, Switzerland, France, Spain and Portugal) (Figure 1). Each European test site consisted of six test areas called land use units (LUU) representing a land use intensity gradient.

The land use units (LUU) in all European partner countries had each a size of 1×1 km and covered the same gradient from relatively natural forests to intensively managed agricultural areas (Table 1).

In all LUU biologists sampled groups of plants and animals (birds, butterflies, soil macrofauna, collembola, carabids, plants, lichens) as indicator species for biodiversity (BioAssess report 2004). Parallel remote sensing images have been acquired covering all selected areas. Within remote sensing a methodology for the assessment of the landscapes and landscape structures was developed as well as diversity indices were calculated. In respect to the qualification of the remote sensing based landscape diversity indices for biodiversity assessment the indices calculated for the indicator species sampled on the ground and the values derived from remote sensing based indices were related to investigate the linkage between remote sensing and ground based methods.

Selection of scale

As mentioned before ecodiversity monitoring is scale dependent. The selection of the aforementioned indicator species for biodiversity monitoring requests an assessment in different

Table 1. Description of the six land use units (LUU).

LSU		Criteria (% cover)
1	Old-growth forest	Old-growth forest >50% Other forests-woodland-shrubland >10% Other land-uses?
2	Managed forest	Managed forest >50% Other forests-woodland-shrubland >10% Other land-uses?
3	Mixed-use dominated by forest or woodland	Forest-woodland-shrubland >50% Grassland >10% Crops >10%
4	Mixed-use not dominated by a single land-use	Forest-woodland-shrubland >25% Grassland >25% Crops >25%
5	Mixed-use dominated by pasture	Grassland >50% Crops >10% Forest-woodland-shrubland >10%
6	Mixed-use dominated by arable crops	Crops >50% Grassland >10% Forest-woodland-shrubland >10%

scale levels. In order to integrate the different scale levels a sampling grid was defined with 16 samples in each LUU unit for the terrestrial assessment of the plant and animal species (Figure 2). Based on the data take at the 16 sampling points in each LUU species biodiversity indicators were calculated by the biologists. The terrestrial results of the calculated indices was provided by the group of biodiversity specialists in the BioAssess project to the remote sensing group.

To find the appropriate remote sensing scale to link these terrestrial data with the remote sensing data it is necessary to know which landscape features are relevant for the selected indicator species respectively which landscape features are in general relevant for overall biodiversity studies. Taking into account the discussion above one approach might be the mapping of habitats (Jedicke 2001). In BioAssess the habitat structures of the indicator species are well defined by the specialists but taking into account the selected species it becomes clear that remote sensing is not the right tool to describe directly the habitat quality for some of the selected species. So what can remote sensing provide? Remote sensing can provide information on land cover types, landscape diversity and landscape structures and their changes over time. The assessment of land cover pattern and structure is linked with ecosystem functions and habitat quality. As Tischendorf (1995) indicates on species and populations level the information on land cover and landscape structure is most relevant to have an approach to the steering biodiversity parameters like habitat quality, isolation, fragmentation and connectivity. Even so it is clear that information on landscape level is closely linked to ecodiversity which is related to biodiversity the question how detailed land cover features have to be assessed is still open. There are no binding lists or recommendations but has to be defined in each case separately. Only if a set of indicator species is defined as a standard set for overall biodiversity monitoring and the interaction of landscape features with the indicator species are investigated then the appropriate answer might be possible.

This approach was also used for the BioAssess project. Based on the habitat requirements described for the indicator species a list of related landscape features has been worked out.

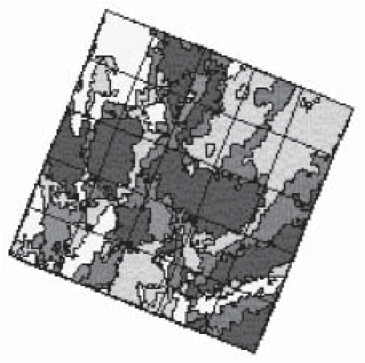

Figure 2. One land use unit (LUU) with the gird of 16 sampling plots.

This list was then adjusted to the CORINE land cover classes to be compatible with the European land use classification system. In a next step the image data sets have been selected. This is the approach most biodiversity projects integrating remote sensing data use for today. Nevertheless this is still a ideal situation because in most cases data selection is restricted by data availability and costs. Therefore the selection of remote sensing data sets is in most cases a compromise on feasibility in respect of costs and size of landscape features which are considered important. Nevertheless for the practical approach to use remote sensing data the most critical point is the existing gap of knowledge how the different landscape features are related to the diversity of indicator species.

In order to investigate the potential of remote sensing images in BioAssess for biodiversity monitoring, three spatial resolutions were investigated. Landsat ETM image was used due to its good spectral resolution (seven channels), good availability, and relatively low price. To improve the spatial resolution of the Landsat data, a panchromatic IRS-1D image (5m spatial resolution) was purchased. These two data sets were used as standard data sets for all LUUs in all countries. In addition for one super testsite in Switzerland a multispectral Quickbird image was acquired, available with a spatial resolution of 2.8 m but a relatively high price and low spectral resolution (3 channels) as well as orthorectified CIR aerial photographs scale 1:10 000 processed as orthoimages with 0.6 m spatial resolution. In selecting the images, caution was taken that none of them was older then 4 years and that they covered the vegetation period from May till August (Figure 3). In respect to the super testsite Switzerland the selected vegetation period can be considered as relatively stable because it is dominated by pasture and forests.

For the investigations in the BioAssess project there are two different approaches a pragmatic one which will use Landsat ETM and IRS 1C pan data for all test sites and a research oriented one with image data of different scales from very high to Landsat ETM resolution in a selected super test site in Switzerland. For the latter the interdependence between different remote sensing scales, remote sensing derived diversity indices and terrestrial diversity indices from indicator species was tested.

Optimisation of image layers for information extraction

Before images can be used several steps of preprocessing and processing have to be provided in order to extract the wanted information. The steps for preprocessing are well known and

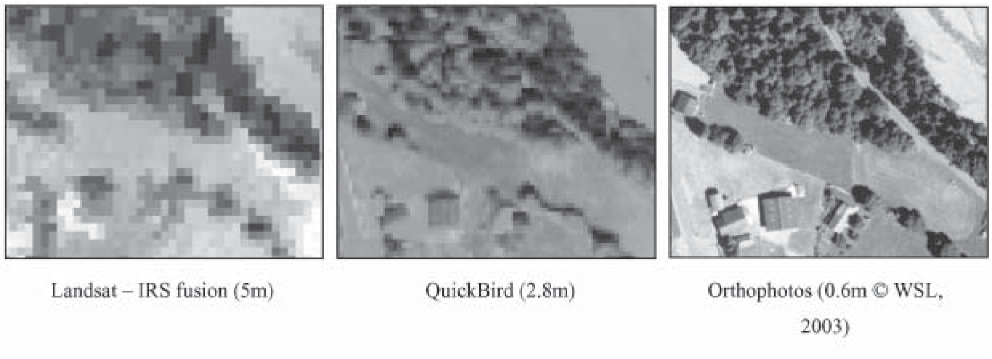

Figure 3. Example of the used remote sensing data.

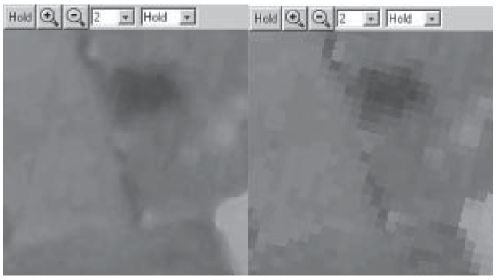

Figure 4. AIF filtered image (left) and original image (right) (Fritz 1999).

will not be discussed here. Having in mind that the BioAssess project is looking for a standardised method to apply it with small adaptations to different data sets over test sites in different bioregions the method has to be transparent and flexible. For the biodiversity monitoring a object based approach was tested. In addition the object based assessment gives better reference to terrestrial mapping or photo interpretation. To achieve an optimisation of information data fusion techniques were applied for the Landsat ETM and IRS 1D pan data. There are different fusion methods available. Pohl (1996) gives an extended overview on different procedures. Most of the fusion algorithms lead to alterations of the multispectral grey values by the panchromatic data set. For classification it is most desirable to keep the original values of the multispectral data set. Therefore the adaptive image fusion filter (AIF) (Steinnocher 1999) was introduced for image fusion. The AIF uses a modified sigma filter for the fusion of the panchromatic (IRS) and the multispectral (Landsat) images. In this method, a local moving window averages those pixels in the higher resolution panchromatic image, which are located within two-sigma distance from the central pixel. Edges are conserved because they do not belong to the two sigma distance. The detected edges of the panchromatic image form the objects. These selected objects are then filled with the values of the averaged multispectral band values. An important advantage of the AIF method is, that no spectral information is transferred from the panchromatic image into the multispectral values (Steinnocher 1999) thus the original radiometry is kept. The use of such a filter sharpens the edges of objects inherent in the high resolution image, while the multispectral grey values within each single object are smoothed (Figure 4). This approach provides the extraction of landscape features within high resolution images without a priori knowledge, but the output is always restricted to the input scale level. For the object identification the pure multispectral values can be used. The sharpening of edges is an important pre-condition for a clear segmentation of objects with decreased distortion of pixel based edge effects.

The major limitation of AIF is the loss of texture information within the objects, which hinders the segmentation of fine details if needed. The local variation of grey values in the higher resolution image unfortunately cannot be reconstructed without distorting the spectral characteristics of the multispectral band. Therefore, an additional processing step was needed to re-import the structural information from the IRS data into the AIF filtered fused image. Fritz (1999) reported good results with the intensity, hue, and saturation (IHS) transformation of the AIF and the high-resolution image. When compared to other methods, the “blockiness” from the Landsat 30 m resolution pixels completely disappeared from the *AIF-IHS* fused results. This is very important for the segmentation of objects, which form the basis for landscape indices calculations. During the IHS transformation, the intensity channel has to be replaced with the panchromatic, high-resolution band, from the IRS band in the respective case. The AIF-IHS transformation is also especially helpful for the visualisation of the data often the basis for land cover interpretation maps.

In addition, several standard procedures to enhance the spectral attributes of the original data. As usual application in the available Software Packages ERDAS Imagine the Principle Component Analysis, Tasselled cap transformation, several ratio procedures like NDVI and other ratios which are appropriate for vegetation mapping, have been processed. The described spectral enhancement procedures have been provided to the standard data set Landsat ETM and IRS 1C pan and as far as possible to the high resolution data sets.

During the process, the user defines the size of resulting objects in each hierarchical level and the algorithm builds a semantic connection between the different levels. Two main hierarchical levels were segmented in the present study, which gave basis for the classification of land cover types.

Segmentation of landscape features

Only recently the segmentation of objects became a practicable tool for image classification. The advantage for segmentation based approaches are manifold. One major advantage is the extraction of homogenous objects which corresponds to our perception of environment. This is also true for biodiversity studies on landscape level which is based on the delineation of basic units like habitats or ecotops. For biodiversity studies landscapes are divided into geometrical units with sharp borderlines. Even so it is well known that sharp borderlines and homogenous units seldom reflect reality it seems to be the only feasible way to come up with a practical approach for landscape diversity characterisation. The problem using segmentation based classifications is the definition of thresholds which influences strongly the delineation of objects. The set of thresholds is based on empirical approaches and requests a thorough a priori knowledge on the kind of units which are of interest. There are different segmentation algorithms available nevertheless all of them request the interaction with the elaborator. The problem with interactions by the elaborator is the problem of standardisation. But having in mind that this influence is inherent to most processes, like terrestrial mapping, sampling of indicator species (location of samples, sampling time, arrangement of sampling a.s.o.) then the factor of individual influence of the elaborator to some extent has to be accepted. The e-cognition software was used for segmentation in the BioAssess project. All the images were segmented using the FNEA approach (Baatz and

Schäpe 1999). This procedure is a bottom up region-merging segmentation technique that operates on several image objects hierarchies. The segmentation was applied to the AIF-sigma-IHS images of the Landsat ETM/IRS 1D pan data in all test sites and in the super testsite in Switzerland in addition to Quickbird and the aerial photos.

Classification of landscape features

For classification the definition of classes is most important. The definition of classes often depends on the expectation of the user, what information they finally need. The expected classes should drive the scale and data type used to extract the information, nevertheless often restrictions due to availability of data drive the selection of data type. Even so it is not possible to get more from a data set then is in a data set there are still differences possible due to the kind of information extraction. For the BioAssess project as for most projects in biodiversity studies the classification system is user driven. Features on landscape level were defined which are assumed to have a relation to the diversity of the indicator species. In BioAssess specialists for the species indicators defined relevant landscape features. These landscape features have been revised according to their feasibility to assess them with the given data sets. The final landscape features selection then has been adjusted to the CORINE classification scheme. A hierarchical segmentation based classification scheme was used, classifying the segments in two levels. On the first level coarse classes have been selected which most possibly can be transferred directly to all test sites in the eight countries (*project level*). The classes are: artificial surface, open spaces, grassland, forest, wetland and waterbodies. On the second level (*country level),* the classes were further classified to follow country specific characteristics (Table 2). Based on the classification mask of the first level, forest areas were further divided into deciduous, conifer, mixed, open forest classes and small forest habitats. Small forest habitats were defined as forest segments smaller then 1ha, surrounded by non-forest areas. The small forest habitats were not further differentiated in the above mentioned categories. Other vegetation classes were grouped into grassland and agricultural areas (Figure 5).

Optimisation task is often driven by the low input information and high quality output problem. Biodiversity studies often require the assessment of many small habitats from remote sensing images, in order to correlate the information with the field data. For this purpose not only the multispectral but also the structural information (texture) from the higher resolution panchromatic band might be useful for the classification. The most often used method to measure texture is the so-called grey level co-occurrence-matrix (GLCM) (Music and Grover 1990; Haralick et al. 1973). Therefore GLCM texture images were also included into the layer stack for the classification.

Statistical Analysis

The statistical analysis presented in this paper are related to data on the super testsite in Switzerland. Landscape metrics quantify the pattern of the landscape, where the landscape itself should be defined inside a logical (economical, social, ecological) boundary. Based on the terrestrial data takes species richness was calculated for all indicator species by the biologists. Due to the terrestrial design the sampling of the indicator species was restricted to the sampling points. That means 96 terrestrial based observations (species richness index) for

Table 2. The used classification scheme.

Level 1	Level 2
Artificial surface	Cities Roads Human construction
Open spaces with little or no vegetation	Open soil Rocks Gravel Rocks and open soils
Grassland	Agricultural
Forest	Broadleaf closed Broadleaf open Broadleaf very open Coniferous closed Coniferous open Coniferous very open Coniferous clear cut Coniferous storm Mixed closed Mixed open Mixed very open Mixed storm
Wetland	Marshland Swamps Peat bogs Mire Moor
Water bodies	Lake River

each indicator species were provided as dependent variables. For the remote sensing based data in a first approach the related landscape for each observation was reduced to a 25 m radius circle and in a second approach to a 100m radius circle around sampling point because a number of indicator species have a very small action radius. This influenced calculation of landscape indices, especially measures like nearest neighbourhood or connectivity. Therefore, simple patch metrics were used instead, which refer to single elements of the landscape and not to the whole area as a contextual unit. The following patch indices have been computed inside the plots:

- area of each classes in a 25 m and 100 m radius plot,
- number of patches in a 25 m and 100 m radius plot,
- number of classes in a 25 m and 100 m radius plot,
- class in which the centre of the plot is located.

Patch indices were calculated within the super test site based on the Fragstats software package for all spatial resolutions described above. Besides patch indices, both original and enhanced pixel grey values were included in the analysis. The grey value derivatives were also calculated for all spatial resolutions (Table 3). Also digital elevation model and digital

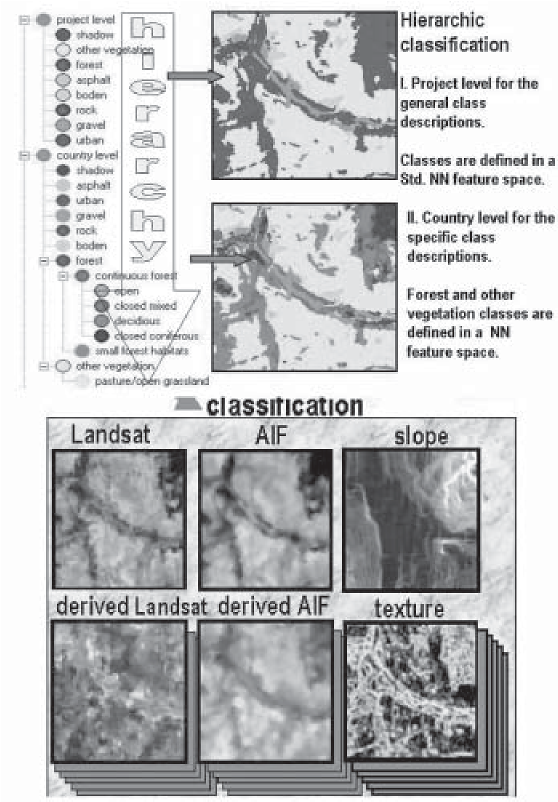

Figure 5. Classification hierarchy and input image for classification (Ivits and Koch 2002).

surface model data were used. This means a set of explanatory variables was created based on calculated landscape indices from classified remote sensing data, from pixel grey value derivates (spectral and textural), from digital elevation model (25 m) derivates and from digital surface model (1 m) derivates.

Before regression model calculation, factor analysis were used to reduce multicollinearity of the explanatory variables of grey values and grey value derivates. This step was done without using the dependent variables. It should be highlighted that no factors were used in the subsequent regression analysis but the variables selected by the factor analysis. This step was necessary since the number of explanatory variables was relatively high related to the number of observations. There was no need of $\log_{10}$ transformation but the variables were standardised to compensate the influence of dominant variables. Factors explaining 90 percent of the total variation and with an eigenvalue above one were extracted. The variables contained in the extracted factors were then used as input for the regression model.

Before regression analysis, also the patch indices were checked in a correlation table (Pearson's correlation). Patch indices with no significant correlation ($p > 0.05$) were excluded from the analysis. Furthermore, between two highly correlated indices ($R > 0.9$) only one was selected to enter the regression model.

Stepwise linear regression was applied to model species richness for woody plants, carabids and birds as dependent variable. Nine regressions were run with each of the remote sensing databases as explanatory variables. The null hypothesis was tested, that variation in species

Table 3. Grey value derivates.

	Channel	Comment
Focal Analysis	Density	Returns number of occurrences of the centre pixel value in focal window
	Diversity	Returns number of different values in focal window
	Rank	Returns the number of pixels in the focal window whose value is less than the centre pixel
	Standard Deviation	Returns standard deviation of pixels in focal window
	Majority	Returns the most commonly occurring value in focal window
	Minority	Returns the least commonly occurring value in focal window
	Minimum	Returns the minimum of values in focal window
	Maximum	Returns the maximum of the data file values in focal window
Textures	Variance	Returns variances in moving window
	Skewness	Returns Skewness in moving window
	ASM	Returns Angular second moment in moving window
	Homogeneity	Returns homogeneity in moving window
	Entropy	Returns entropy in moving window
	Contrast	Returns contrast in moving window
	Dissimilarity	Returns dissimilarity in moving window
	Correlation	Returns correlation in moving window

	Channel	Comment
Spectral enhancement	NDVI	Normalised difference vegetation Index: $\frac{NIR - RED}{NIR + RED}$
	TNDVI	Transformed NDVI : $\sqrt{\frac{NIR - RED}{NIR + RED}} + 0{,}5$
	Vegetation Index	Vegetation Index : $NIR - RED$
	Ratio 1	$\frac{MIR}{RED}$
	Ratio 2	$\frac{MIR}{NIR}$
	Ratio 3	$\frac{IR}{RED}$
	Ratio 4	$\frac{NIR}{RED}$
	Ratio 5	$\sqrt{\frac{NIR}{RED}}$
	Tasseled Cap	Transforms data to optimise viewing for vegetation studies
	First Principle Component	Compresses redundant data values into fewer bands, which are often more interpretable than the source data

Channel	Comment
DEM	Digital elevation model
DSM	Digital surface model
Difference	Absolute elevation of terrain objects (DEM-DSM)
Aspect	Aspect of the DHM and the DSM
Slope	Slope of DHM and DSM
Plan Curvature	Perpendicular to the direction of maximum slope
Profile Curvature	the direction of maximum slope
Curvature	Plan Curvature - Profile Curvature
ASM	Returns Angular second moment in moving window
Homogeneity	Returns homogeneity in moving window
Entropy	Returns entropy in moving window
Focal minimum	Returns the minimum of values in focal window
Focal Maximum	Returns the maximum of the data file values in focal window

richness could not be explained with the remote sensing indices. The alternative hypothesis was defined such that the derived remote sensing indices were predictors of woody, carabid and bird species richness. For rejecting the null hypothesis, the significance of the F statistic was studied. The null hypothesis was rejected on the $\alpha = 0.1\%$ level when the significance value (p) of the F statistic was ≤ 0.001. The coefficient of determination values (R^2) were used to test how much variation in woody, carabid and bird species richness the single models explained. Furthermore, the R^2 values were compared to demonstrate which remote sensing dataset explained the most variation in woody, carabid and bird species richness. The performance of the variables that entered the model was explored based on their partial correlation to woody, carbides and birds species richness and the significance of the correlations.

Accuracy Assessment

Only for the test sites in Switzerland orthorectified colour infrared (CIR) aerial photos scale 1:10 000 were available. The accuracy assessment is based on the aerial photographs and therefore it was only possible to provide a accuracy assessment for the super test site. The aerial photographs allowed to interpret the area very well and the option of collecting points with a GPS during field visit was abandoned. The software eCognition offers two possibilities for object oriented accuracy assessment: in the first case, the user selects objects in the segmented image, which can be assigned to one of the defined classes based on the ground truth material. The software then compares these objects to the classification result and computes the accuracy. In the second case, the user imports a classified thematic layer prepared from any ground truth data, which serves as basis for accuracy assessment. The first approach was selected.

Results

Results on object-based classification

The accuracy assessment in BioAssess proved that the segmentation based classification (Fractal Net Evolution Approach FNEA) commercially introduced by Baatz and Schäpe (1999) provides a good basis for further calculation of diversity indices. The object-oriented classification proved to be very suitable for different landscape types. Accuracy was in general for the first classification level over or around 95% (Table 4). For the standard data set Landsat ETM/IRS 1D pan (Landsat-IRS) fusion image the smallest forest object correctly recognized was 0.022 ha (225 m^2).

In total the aerial photographs showed the highest accuracy followed by Quickbird and Landsat-IRS fusion image (Figure 6). Urban areas and open surface were best classified from the aerial photos and from Landsat-IRS the worst. On the aerial photo these objects are clearly delineated while on Landsat-IRS level urban areas were less successful to separate. This results in a worse segmentation as well as classification of these objects. Continuous forested areas were very well classified from all the three remote sensing datasets. Although the accuracy of the second level classification was generally lower then that of the first level. Nevertheless the second level results were still very good with around 90% of overall accuracy. As before, the aerial photo classification gave the best overall accuracy results on this level. Deciduous and mixed forest were in all the three datasets with a user accuracy

Table 4. Accuracies of the landcover classifications according to level 1and level 2 automatic classifications and to the used remote sensing data.

	Landsat-IRS		Quickbird		Aerial Photo	
Level 1 Overall Accuracy	0.948		0.975		0.994	
Accuracy	User	Producer	User	Producer	User	Producer
Forest	0.995	0.975	1	1	0.998	1
Grassland	0.946	1	0.993	1	0.994	1
Artificial surface	0.733	0.546	0.851	0.722	0.976	0.958
Open spaces	0.852	0.899	0.882	0.921	1	0.973
Level 2 Overall Accuracy	0.900		0.900		0.963	
Accuracy	User	Producer	User	Producer	User	Producer
Broadleaf	0.804	0.895	0.851	0.701	0.991	0.893
Coniferous	1	0.488	0.843	0.989	1	0.891
Mixed	0.833	0.889	0.854	0.676	0.876	0.994
Cities	1	0.872	0.960	0.792	0.934	0.702
Roads	1	0.578	0.882	0.886	0.885	0.958

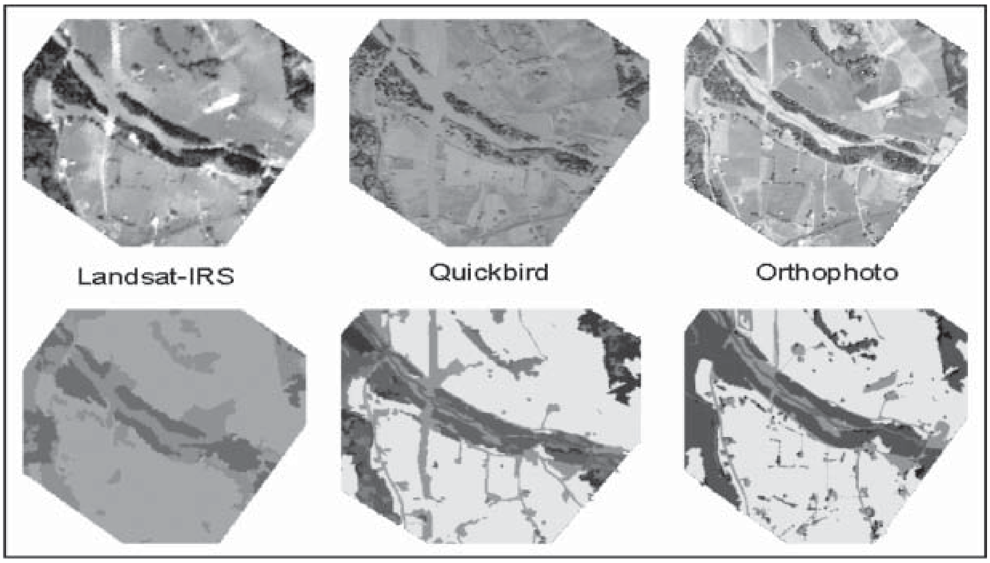

Figure 6. Object-based classification from Landsat-IRS pan fusion image, Quickbird and aerial photograph.

between 80 and 90% classified. These classes seem to be the most stable once which is probably based on the fact that they are most often present in the area. Coniferous forest was from the Landsat-IRS and aerial photo datasets with a much higher accuracy classified as from the Quickbird image. Coniferous forest patches are on the Landsat-IRS resolution very homogeneous objects while on Quickbird images these objects are very heterogeneous. On aerial photo resolution small single coniferous patches (one to few crowns) can be segmented separately from small deciduous patches, that makes their classification more accurate.

In order to get from rather coarse data like Landsat ETM as it was used as standard data set for all test sites in the BioAssess project relative fine structural information a fusion with high resolution panchromatic images with images is needed, as provided in the BioAssess project by IRS 1D pan data. In addition for the classification all information has to be used inherent in the images the structural information from the high resolution images in form of texture image layers as well as the multispectral information. For the multispectral information as input for the classification the AIF filtered images and their derivatives (NDVI etc.) seem to be more appropriate for object classification because the filtered image follow the segments delineated from the high resolution image. With the help of texture and slope images, non-vegetated features were correctly identified. Homogeneity was useful when separating homogeneous from heterogeneous non-vegetated areas like asphalt, bare soil, and gravel surfaces from urban areas. Between homogeneous non-vegetated surfaces, the Standard Deviation and the Dissimilarity measures delivered good results.

Results on correlation between terrestrial based species diversity indices and remote sensing based information

Tables 5 to 7 show the good fit of the models between woody plant, bird and carabid species richness and patch indices extracted from first level segmentation based classifications of the three datasets available for the test sites in Switzerland. In all analyses the models were highly significant ($p < 0.001$), i.e. the hypothesis that species richness for the investigated taxa is uncorrelated with the classified remote sensing data can be rejected.

Table 7 displays model output concerning species richness dependent on grey value derivatives as explanatory variables. All the models were highly significant thus the hypothesis that woody species, carabids and bird species richness does not correlate with the extracted remote sensing indicators can be rejected. Overall grey value derivatives gave similar correlations as landscape indices extracted from the first level classifications, but second level classifications gave better correlation for woody plant and bird species richness. For carabid species richness the landscape indices provided less correlation compared to the grey values.

Conclusions and future research

The use of remote sensing data for ecodiversity studies respectively biodiversity studies is just at the beginning. Until today most biodiversity studies are concentrated on species indicator assessment studies which restricts the studies to terrestrial sample studies. The discussion in the past indicate that the landscape diversity is one important indicator for biodiversity monitoring taking into account that diversity of species is nested in diversity of landscapes. Terrestrial sampling cannot provide the holistic information on a landscape level. This points towards the need to integrate remote sensing methods in biodiversity studies. Nevertheless the methods to use remote sensing data in a way to serve the information needed for biodiversity studies asks for more investigations in order to fill the gaps of knowledge in respect of scale dependent feature extraction, automation of feature extraction and on the interrelation between indicator species diversity and diversity of landscape features.

This study shows first trends. It shows that object oriented classification is a good approach for assessment of landscape features especially in large areas, if the set of needed landscape

Table 5. Linear regression model outputs for variables best describing woody plant, bird and carabid species richness based on level 1 classifications.

Level 1	Landsat_IRS			Quickbird			Photograph		
	Woody species	Birds	Carabids	Woody species	Birds	Carabids	Woody Species	Birds	Carabids
R^2	0.647	0.508	0.209	0.642	0.554	0.226	0.665	0.586	0.253
Significance	0.000	0.000	0.006	0.000	0.000	0.011	0.000	0.000	0.002

Table 6. Linear regression model outputs for variables best describing woody plant, bird and carabid species richness based on level 2 classifications.

Level 2	Landsat_IRS			Quickbird			Photograph		
	Woody species	Birds	Carabids	Woody species	Birds	Carabids	Woody Species	Birds	Carabids
R^2	0.669	0.627	0.394	0.707	0.617	0.293	0.707	0.642	0.293
Significance	0.000	0.000	0.003	0.000	0.000	0.058	0.000	0.000	0.018

Patch indices explaining best woody species richness
Area of grassland in a 25m radius Distance to grassland patches in a 100m radius Area of gravel patches Number of open mixed forest patches in a 25m radius

Patch indices explaining best bird species richness
Area of forest Number of grassland patches in a 25 m radius Area of gravel patches Area of open coniferous forest Number of very open coniferous forest patches in a 25 m radius Number of agricultural patches in a 100 m radius Distance to lakes in a 100 m radius

Patch indices explaining best carabid species richness
The class where the plot is located in Number of gravel patches in a 25 m radius Distance to open spaces in a 25 m radius

features is defined. On one hand, the automatic object based approach is faster than the often used visual interpretation and on the other hand the process is also better to standardise than visual interpretation. Even so automatic segmentation is also influenced by the elaborator the method is compared to a visual interpretation much better standardise and more cost efficient. Therefore it might replace visual interpretations in landscape level studies of biological diversity. Finally the object-based classification allows to provide the information in a form

Table 7. Linear regression model outputs and variables best describing woody plant, birds and carabids species richness based on grey value derivatives.

Level 1	Landsat_IRS grey values			Quickbird grey values			Photograph grey values		
	Woody species	Birds	Carabids	Woody species	Birds	Carabids	Woody Species	Birds	Carabids
R^2	0.644	0.648	0.432	0.562	0.516	0.423	0.647	0.421	0.379
Significance	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.000	0.000

Grey values variables best explaining woody species richness
Variance of the green channel
Focal maximum filter of the green channel
GLCM entropy texture of the near infrared channel
GLCM correlation texture of the blue channel

Grey value variables best explaining bird species richness
Variance of the blue channel of the AIF fused Landsat – IRS image
Focal standard deviation filter of the middle infrared channel of the AIF fused image
Skewness filter of the blue channel of the Landsat image
GLCM Entropy texture of the IRS image
Skewness filter of the middle infrared channel of the AIF fused image
Focal rank filter of the near infrared channel of the AIF fused image
The fourth tasselled cap channel of the Landsat image
Skewness filter of the far infrared channel of the Landsat image

Grey value variables best explaining carabid species richness
First principal component of the six Landsat channels
TNDVI of the Landsat image
Skewness filter of the blue channel of the Landsat image
The fourth tasselled cap channel of the Landsat image

users of map information are accustomed to. All together high classification accuracy was achieved for the selected CORINE classes at all data levels. More detailed classes would have been possible on the basis of Quickbird and orthophotographs.

The derived remote sensing indices showed good potential in predicting species diversity data. Patch indices derived from classified values showed slightly better correlation to species diversity then image grey value derivatives. Only exception was carabid species diversity, where the linear regression model showed better fit based on image grey values. Thus, the study showed that image grey values are possible alternatives of classifications when predicting species diversity. Nevertheless it has to be admitted that the interpretation of the biological meaning for some of the grey value derivates is a problem and therefore in further studies grey value derivates should be carefully revised according to their interpretability. Patch indices derived from second level classifications showed slightly better correlation to species diversity then first level classifications. So more detailed landscape description seems advantageous when assessing species diversity based on patch indices. There was no significant differences between spatial resolutions of the grey value derivatives when predicting woody plant and carabid species richness. In case of birds, however, there was a trend towards better correlations between richness diversity and grey level values with more coarse remote sensing sets which means also based on the used data sets better correlation with the data set of higher spectral resolution. All together it was proved that there is strong correlation between indices derived from remote sensing data and species diversity indices derived from terrestrial data collections.

Nevertheless the authors are aware that the presented results are still very limited. The few selected indicator species investigated until now do not allow a generalisation of the results but rather give indications. Especially for birds the restricted area around the sample points investigated does not reflect the action radius of many bird species. There are still hugh data sources from the BioAssess projects which need to be evaluated further to strengthen the first indications. A follow up project might give the chance to do so also the BioAssess project is finished.

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Implementing Indicators for Forest Monitoring within Alpine Natura 2000 Sites

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Abstract

In this paper three landscape-level indicators have been assessed to support an harmonized forests monitoring and reporting on protected areas: the change in area covered by forests and other wooded land, the proportion of different types of forest ownership and the proportion of mixed forests (conceived here as the opposite of a pure forest). This study contributes to testing the implementation of available European Core sets of indicators proposed in 2003 by the Ministerial Conference on the protection of Forests in Europe (MCPFE) and by the European Environment Agency (EEA). Indeed, those three indicators are identified in those two core sets.

Three protected areas have been selected within the European Alpine Biographical region. They are located from medium to high mountain regions and include or correspond entirely to Natura 2000 sites. Two of them form a transboundary protected area between Italy and France. Remote sensing data and GIS techniques have been used. Results demonstrate the key contribution of multi-temporal Earth Observation data to support management of protected areas for conservation purposes. The implementation of such indicators further require the integration of several GIS data-layers and the study reaffirms the general need of harmonization of scales and legends for forestry maps provided by local authorities and public institutes.

Keywords: Natura 2000; indicators; forest area change; forest ownership; mixed forest; remote sensing; GIS.

1. European Nature and Forest Policy framework

This work aims to support the monitoring of and reporting on forests ecosystems protected within Natura 2000 sites and located within the Alpine biogeographic region. It provides a contribution to research needs associated to the reporting and monitoring issues of the following European legislative instruments:

1. Within the European Union nature conservation policy the Birds and Habitats Directives (‘Council Directive 79/409/EEC on the conservation of wild birds’ and ‘Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora’): the Habitats Directive aims to maintain vital components of the natural heritage at favorable conservation status. To this end, it establishes a European ecological network of protected areas called Natura 2000 (Article 3). Reporting requirements on the conservation status of the sites and their hosting species and habitats are every six years.
2. Concerning the protection of forests, the EU recently approved regulation referred to as Forest Focus (Regulation No. 2152/03 of 17 November 2003 concerning monitoring of forests and environmental interactions in the Community). In addition to the monitoring of the impacts of atmospheric pollution and forest fires, which were already covered by two previous regulations, Forest Focus now includes new development themes, among which forest biodiversity.

Furthermore, specifically targeted to the protection of the Alpine region, the International Convention on the Protection of the Alps, known as Alpine Convention, was signed in 1991 and entered into force in 1995. It requires its contracting parties (Austria, France, Germany, Italy, Liechtenstein, Slovenia, Switzerland, and the European Community) to pursue a comprehensive policy for the preservation and protection of the Alps through application of the principles of prevention, payment by the polluter and cooperation. Among others, the Convention covers the following themes: conservation of nature and the countryside, mountain forests, tourism and recreation.

The progress made towards the goals of the above-mentioned legislative instruments needs to be reported on and to do so, monitoring schemes should be set up. Tools such as indicators together with customized and harmonized spatial databases are essential to achieve consistency among Member States in terms of surveillance methods of protected sites and also as regards trans-border cooperation (Estreguil 2002).

2. Existing indicator initiatives

There is an urgent need to implement currently available indicators for reporting on biodiversity at European levels (Delbaere 2002). Following a short review of policy instruments related to Biodiversity and Nature protection and their respective objectives, Estreguil et al. (2003) identifies available European sets of indicators and clarifies the two user groups for indicators:

1. policy-makers at European level who evaluate the progress made towards objectives set in policies and measures;
2. implementers and managers acting at local and regional levels who provide data to higher decision-making levels and implement EU instruments at the site level.

Moreover, Estreguil et al. (2003) states that cross-relation of indicators developed for implementers and managers at the local and regional levels with policy-making indicators designed for the national/European levels must be ensured to facilitate a common reporting and coordination among organizations operating at local, regional, national and European levels.

This study addressed the specific information needs of Natura 2000 sites' local managers, then developed three indicators for forest monitoring within those sites and cross-related these indicators to the following available European Core set of indicators:

- Regarding specific forestry related indicators: (1) "The improved Pan-European Indicators for Sustainable Forest Management" (MCPFE 2003) that comprise a set of six criteria and 35 quantitative indicators and specific reference was made to indicators set under criteria 1 (forest resources), 4 (biodiversity) and 6 (socio-economic functions); the "Pan-European process", i.e. Ministerial Conference on the Protection of Forests in Europe (MCPFE) and (2) the key factors as potential indicators of European forest biodiversity from the BEAR Concerted action of the 5th EU Framework Programme (www.algonet.se/~bear) that characterize the forest ecosystem according to the major ecosystem attributes (structure, composition and function) and scales (national/regional, landscape, stand) (Larsson 2001)
- Regarding indicators for Biodiversity and Nature Protection, reference to the European Core Set of the European Environmental Agency (EEA 2003), for reporting and policy-making on several thematic environmental topics has been made, particularly on sets for the biodiversity and terrestrial environment.

For the development of the three indicators remote sensing and ancillary data have been used. For each of them the required data, the methodological approach which has been used and the obtained results are described.

3. The Alpine region and forest processes

In case of wide areas, such as the European territory, the general inner complexity and the compositional and structural diversity of ecosystems to be monitored and reported on recommends the adoption of a regionalisation procedure. This is helpful to recognise homogeneous regions with respect to biotic and abiotic properties within which the same methodology for monitoring and reporting can be applied. Moreover it provides a geographic framework to analyse ecosystem/sites in relation to their environment and in which similarities on anthropogenic and natural pressures as well as on responses like management strategies may be expected for similarly defined sites. Article 1 of the Habitat Directive identifies six Biogeographical Regions: Alpine, Atlantic, Continental, Macaronesian, Mediterranean and Boreal. This study focuses on the Alpine biogeographic region where three protected areas were selected. The three indicators assessed in this study were identified by considering the main aspects characterising the natural resources and socio-economic structure of the Alpine region.

The Alps, as the widest European mountain range, contain forest ecosystems that are sensitive to natural and anthropogenic disturbances. They are important sources of timber and for centuries, mountain communities have played a leading role in maintaining a sustainable flow of timber resources to the plains below. However, since the 19th century with the advent of improved infrastructures and socio-economic changes, Alpine forests have been exploited commercially for timber at unsustainable rates and on large spatial scales to satisfy social needs for resources, with negative effects on erosion and slope stability. At the beginning of the 19th century the numerous natural catastrophes lead public authorities to adopt special measures for a more rational use of forest resources and for the maintenance of their protective role, especially in mountain areas (Hamilton et al. 1997). Reforestation programs started in this period. In the

20th century mountain forests on the Alps have become a focus of conflicting interests between economic development and environmental conservation (Kräuchi et al. 2000). Now the protective function is becoming the primary function of forests, especially as a consequence of increasing population density and pressure from emerging tourism in some mountain areas (Grötzbach 1988). However if in some areas of the Alps tourism accounts for widespread population growth at higher altitudes (Bätzing et al. 1996), other areas of the Alps have not been affected by modern development and both the economy and population are declining (Price 1994). Since the middle of the 19th century these areas were affected by changes of the traditional agro-pastoral system (Didier 2001). Some of these areas may even become completely abandoned and socio-economic changes have led to an expansion of forest area. In fact ancient pasture lands and hay meadows, once abandoned, have become covered by tall herbs and by pioneer tree species. This phenomenon is causing the closure of the landscape with potential consequences on tourism frequentation, loss of biodiversity, cultural landscapes, and traditional land use practices. Most recent studies also demonstrate that regeneration can be responsible for the "missing sink" in the global carbon budget (Caspersen 2000).

4. Study areas and available dataset

The three areas selected to apply the three indicators are located from medium to high mountain regions and include or correspond entirely to Natura 2000 sites. They are:

- the National Park of Mercantour (Eastern French Alps) and the Natural Park of the Alpi Marittime (Western Italian Alps) having a common frontier of 35 km and forming a transboundary protected area;
- the Val Grande National Park (Central Italian Alps).

The National Park of Mercantour comprises two parts: a peripherical zone of 146 500 ha and a central part covering an area of 68 500 ha, corresponding to a Natura 2000 site.

The vegetation of the Park is mainly characterized by the presence of silvo-pastoral formations (*pré-bois*) dominated by larches. They form highly dynamic formations whose maintenance is strictly dependant on human intervention. When human intervention ceases, they are exposed to natural dynamics: shrubs invade, new young larches install, the herbaceous component progressively diminishes and other arboreal species such as pines, firs and spruces invade and become dominant, making larch formations to disappear. The result of the process is a gradual trivialization of the landscape with a progressive loss of biodiversity (Sandoz et al. 1998). The National Park of Mercantour is acting in order to maintain the traditional landscapes and high levels of biodiversity. Table 1 summarizes the data available for the National Park of Mercantour

The Natural Park of the Alpi Marittime covers an area of about 27 700 ha entirely corresponding to a Natura 2000 site. The influence of the Alpine and Mediterranean climates makes the area particularly rich of endemic species.

Vegetation is distributed according to three main altitudinal ranges: between 1000 and 1800 m a.s.l. beech woods predominate; at higher altitudes beech is progressively substituted by fir and then by larch or cembrus pines at the subalpine level. The closure of the landscape due to the colonization by pioneer species of abandoned meadows and pastures is particularly evident on the valley bottoms, where pedo-climatic conditions favorite the process. At higher altitudes a change of floristic composition can be observed.

The "Val Grande" National Park extends over about 14 500 ha and ranges from 230 to 2300 m altitude. It mostly coincides with a Natura 2000 site.

Table 1. Available data for the Mercantour National Park.

Satellite images (Landsat)			
Sensor	Scene	Acquisition Date	Spatial Resolution
TM	195/28	23/07/1984	30 m
ETM+	195/28	27/07/2000	25 m
Vector Data			
Layer	Source	Year of Production	Scale
Forestry map	IFN	1985–86	1: 100 000
Forest ownership map	IFN	1985–86	1: 100 000

Vegetation is dominated by broadleaves (chestnut between 250 and 750 m asl, beech at higher altitudes) with a limited quantity of coniferous woods, in which main species are spruce and fir. Larch is scarce because of the climate and the cuttings in the last centuries. Above 1700 m, woods are substituted by Alpine prairies and shrubs.

At the beginning of the 19th century the area was strongly exploited by man for grazing activities, wood production and extraction of building materials. The abandonment of silvo-pastoral activities after the Second World War involved considerably the territory of the Park, mostly due to its inaccessibility, causing a rapid recolonization of pastures by shrubs and trees and the closure of the landscape. This phenomenon has surely been supported by relatively high temperatures and heavy rains characterizing the local climate.

Data available for the Italian test areas are summarized in Table 2 and 3. In this case spatial scales of forestry maps are more detailed than for the French site. This implies the need of a harmonization procedure in scales and legends when comparison between sites is performed, especially for the two transboundary areas.

5. Indicator implementation and discussion

5.1 Land cover change indicator

Since the 19th century many areas of the Alps have been affected by a change in the traditional agro-pastoral system, with consequences on land cover and in particular on forest cover. The area covered by forest within a protected site should be monitored at regular time intervals in order to assess its changes and to understand dynamics of vegetation. Moreover estimates of forest area changes serve as a bench mark state of the environment against which benefits of protection status can be assessed (Prasad 1998).

Within the MCPFE framework, indicators 1.1 (*Forest area and changes*) and 4.7 (*Landscape pattern*) require statistics and spatial distribution regarding changes in the forest cover. From the BEAR factors, direct relevance of the proposed indicator is found to implement national/regional level "*forest area with respect to forest left to free development*" and the landscape level *History of landscape use* factor. From the EEA indicators list, direct contribution is found to implement the *Landscape changes* (BDIV 06) and the *Landscape-level spatial pattern of forest cover* (BDIV 06A) indicators.

Table 2. Available data for the Alpi Marittime Natural Park.

Satellite images (Landsat)			
Sensor	Scene	Acquisition Date	Spatial Resolution
TM	195/28	23/07/1984	30 m
ETM+	195/28	27/07/2000	25 m
Vector Data			
Layer	Source	Year of Production	Scale
Forestry map	IPLA	2001	1: 25 000
Forest ownership map	IPLA	2001	1: 25 000

Table 3. Available data for the Val Grande National Park.

Satellite images (Landsat)			
Sensor	Scene	Acquisition Date	Spatial Resolution
TM	194/28	20/08/1985	30 m
TM	194/28	24/06/1999	30 m
ETM+	194/28	06/10/1999	30 m
ETM+	194/28	09/09/2001	25 m
Vector Data			
Layer	Source	Year of Production	Scale
Forestry map	IPLA	2001	1: 25 000
Forest ownership map	IPLA	2001	1: 25 000

For both the National Park of Mercantour and the Natural Park of Alpi Marittime, we assessed changes over 16 years time-frame in forest and other wooded land area using Landsat satellite images (195/29) from July 23, 1984 and July 27, 2000. Images were geometrically and radiometrically preprocessed. Then a hybrid classification procedure based both on a spectral and spatial analysis was applied to derive a land cover change map (Maggi et al. 2003).

Estimates derived from this land cover change map show that the area interested by changes within the two Parks amounts only for a little percentage of their territory, respectively the 11.5% for the Mercantour and the 7.7% for the Alpi Marittime Park (Table 4). Of the area interested by changes the greatest part corresponds to an increase of woody vegetation outlining trends associated to land abandonment; smaller percentages refer to increases of herbaceous vegetation and to decreases of woody and herbaceous vegetation. Changes related to herbaceous vegetation correspond in some cases to areas submitted to a different land use in 1984 and 2000. Additional images would be useful in order to distinguish these areas from those where the increase of herbaceous vegetation occurred due to a decrease of pastoral activities. A decrease of woody vegetation has occurred were cuttings have been done.

Table 4. Comparison of changes in the Mercantour National Park and Alpi Maritime Natural Park.

Mercantour National Park	%	Alpi Maritime Natural Park	%
No Change	86.0	No Change	88.8
Not Processed	2.5	Not Processed	3.5
Change	11.5	Change	7.7
Class of change	%	Class of change	%
Increase of woody vegetation	60.0	Increase of conifers	35.6
		Increase of broadleaves	1.4
		Increase of pioneer vegetation	47.7
Increase of herbaceous biomass	15.0	Increase of herbaceous biomass	3.7
Decrease of woody vegetation	10.0	Decrease of vegetation (woody and herb.)	11.6
Decrease of herbaceous vegetation	15.0		

Land cover change maps are currently under validation by using a set of aerial photographs. However, visits on the fields, consultations with the parks managers and visual analysis of original satellite images have already confirmed the reliability of the obtained results.

In the case of the Val Grande National Park, satellite images over the same 15 years time-frame are currently being processed and trends of land abandonment have been reported in the literature with the loss of 46% of pastures and their replacement by woody vegetation between 1951 and 1991.

5.2 Forest ownership indicator

The distribution of forest ownership within a study area is an important social indicator. The number and type of forest holdings can have a strong influence on a sustainable forest management. Indeed a private ownership usually implies less financial resources, more parceled forest and a less coordinated forest management.

Within the MCPFE framework, indicators 6.1 (*Forest holdings*) is linked to forest ownership. From the BEAR factors, a direct link can be established with the national/regional level *Forest ownership* structural key factor.

Computing this indicator relies on the availability of digital maps of forest ownership. These maps were available for our three test sites and were crossed in a GIS with boundaries of protected areas. This enables a comparison of the forest ownership structure within our protected sites. Percentages of different categories of ownership are illustrated in Figure 1.

The distribution of forest ownership in the Mercantour National Park reflects the French national situation, with a prevalence of private ownership. In the Italian areas the communal and demanial properties prevail, despite at the national level the private owners are the most common. Another interesting exercise could be to overlay the forest ownership map with the land cover change map to report on possible cause-effect relation-ships.

5.3 Mixed forest indicator

In the context of this study a mixed forest is conceived as a forest containing more than one species either coniferous or broadleaved, i.e. the opposite of a pure forest. The distinction

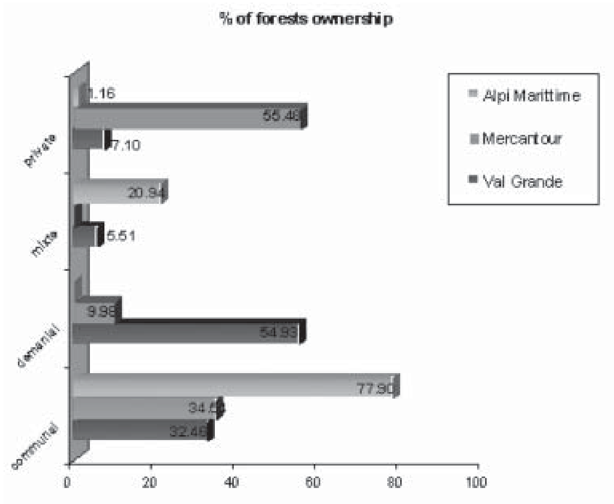

Figure 1. Forest ownership distribution in the three study areas.

between mixed and pure forests was important to highlight the structural diversity of forests between study areas. This structural variety is assumed to be an advantage in case of damages and a better support for biodiversity. Mixed forests are thought to approximate natural forests and natural forest functions (MCPFE 2003).

Within the MCPFE framework, the presence of mixed forests is of interest to the implementation of the indicators 4.1 (*Tree species composition*). In the BEAR factors, a direct relevance of this indicator is found with the *Tree species* structural key factor.

To derive this indicator, forestry maps available for each study area have been reclassified and then crossed in a GIS with boundaries of protected areas. Percentages of pure and mixed forests within each protected site were derived (Figure 2).

Diagram of Figure 2 shows that for the Alpi Marittime Natural Park and the Val Grande National Park there is a prevalence of pure forests (up to 70% for both study areas), while for the Mercantour National Park mixed forests are prevailing. The differences in scales (Figure 2) and legends between forestry maps used for the Italian and French study areas may have a clear incidence on the values found.

6. Conclusions

Indicators to be used for monitoring protected areas should be identified on the basis of local and regional conservation priorities and management needs. In the context of Europe, the implementation of a same indicators set over different protected areas, facilitates obtaining comparable harmonized results to derive trends at European level and to monitor progress towards European instrument goals. Firstly, in such a wide and diverse territory like Europe, a regionalization procedure is mandatory to discriminate homogenous regions within which the same monitoring strategy and associated indicator set can be applied. Moreover, the cross-relation and consistency of indicator set defined by managers at local levels and for policy-making and reporting at European levels must be ensured. The feasibility of this cross-link was demonstrated in this study.

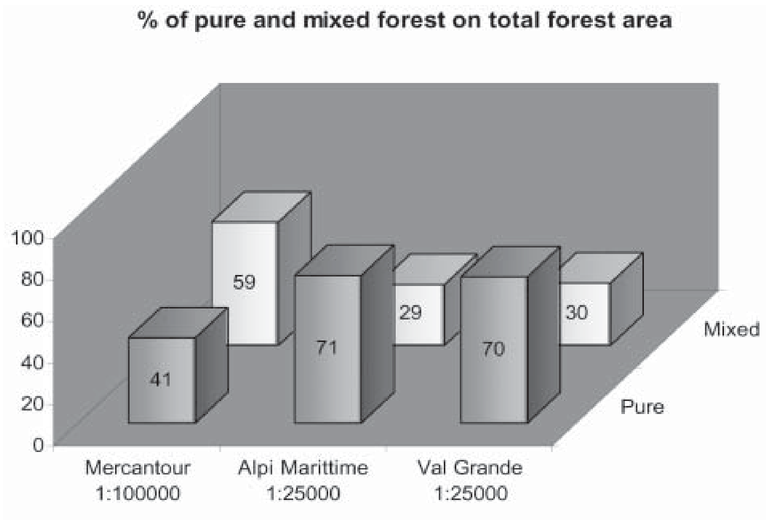

Figure 2. Distribution of pure and mixed forest within the three study areas.

This study focused on the Alpine region, identified and implemented three forest related indicators (forest change, forest ownership, mixed forest) on three Alpine Natura 2000 sites. This was achieved using remote sensing and available ancillary digital data.

It is assumed that the results related to land abandonment and associated forest regrowth are representative of the medium to high mountain range of the Alpine region. Changes of the landscape structure, resulting from changes of traditional socio-economic structures and land use practices since the 1850s, need to be further investigated being strictly related both to landscape diversity and species richness. The use of high resolution Earth Observation data (Landsat) enables the assessment of land cover changes at the landscape level. Derived maps are detailed enough to stratify the protected area in areas of changes of interest, then enabling down-scaling operations using field data or more detailed maps on forest species. Park managers and technicians consider maps of change reliable enough for operational use and useful for the development of their management plans and conservation purposes.

In the case of both the forest ownership and mixed forest indicators, digital maps are provided, when available, by national authorities or public institutes. This study reaffirms the urgent need of harmonization in scales and legends at least for protected sites belonging to the same bio-geographical region. This is particularly true for transboundary areas.

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Spatial Analysis of Natural and Semi-natural Habitats of the Natura 2000 Network in the Sicani Mountains (W Sicily, Italy)

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Abstract

Currently, at the European level, interest in environmental problems is well expressed by the *Habitat Directive* (92/43/EEC) that proposes the constitution of the European Natura 2000 Network. According to this directive, a high number of particularly important ecological sites have been detected at the European scale.

The aim of this work is to analyse the spatial distribution of some natural and semi-natural habitats within the Sicani Mountains Natura 2000 Network, in order to determine their fragmentation level.

Furthermore, this work has allowed to verify the presence of the habitats listed within the Directive and to find out some new ones within the test areas.

A habitat inventory has been carried out through orthophotos interpretation and habitats classified according to the Corine Land Cover classification system, and following the standard classification of the forest and pre-forest types of Sicily. Moreover, forest and pre-forest communities have been analysed at a landscape level using spatial indices (connectivity, fragmentation, diversity) for each patch of the Natura 2000 sites and between different sites of the network.

The most represented habitats within the studied sites are the perennial grasslands among the pre-forest vegetation units, while holm oak woods are the most widespread forest units.

The semi-natural vegetation communities present within the examined territory show a high level of heterogeneity, which seem to enhance biodiversity.

Keywords: Landscape analysis; forest biodiversity; forest fragmentation.

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1. Introduction

Natural and semi-natural areas have progressively been reduced by improper use of territory for human activities causing a high fragmentation of the territory. The consequent reduction of the territorial "matrix" has influenced the ecological processes and functionality in the remaining fragments, triggering the alteration of natural dispersal mechanisms and the progressive decline of the habitat quality both for single animal and vegetation species and for the communities in which they live. Such events cause a "cascade effect" on the whole environmental system over time (Wilcox and Murphy 1985; Janzen 1986; Wilcove et al. 1986; Bright 1993).

In order to reduce this problem, the European Union, through the "Habitat" (92/43/EEC) and "Birds" (79/409/EEC) Directives, has developed the Natura 2000 Network, an Ecological Network in which there is harmony between the various environmental and social-economic processes. The principal objective of this initiative is to limit the erosion of natural resources through the identification of "Special Conservation Zones" (SCZ) and "Community Importance Sites" (CIS). Such areas guarantee the presence, maintenance and restoration of the most threatened European habitats.

The Natura 2000 Network, including protected forests as major ecosystems, is currently involved, as classification system, in the PROFOR (Protected forest areas in Europe – analysis and harmonisation) of Cost Action E4 project. Its objectives are to describe, analyse and harmonise the wide range-of Protected Forest Area categories used in European countries within the context of existing international systems of protected areas.

The study and knowledge of the natural and artificial landscape of these sites is fundamental for the definition of the guidelines for their management, necessary to reach the European Union goals.

This study provides a contribution to the knowledge of the habitats of principal Natura 2000 Network sites in the Sicani Mountains, through a detailed analysis of the actual structure, the integrity level and the degree of fragmentation of the habitats themselves. More in detail the study has allowed the classification of land use and forest and pre-forest types of the investigated sites, the delimitation, classification and integration of habitats listed in the Directive 92/43/EEC within the sites, the definition of habitat fragmentation both at a site and at a landscape scale and the analysis of habitat composition and spatial structure for site planning and management purposes.

2. Study area

The Sicani Mountains are located in south-west Sicily. This area is bounded by Monte Cammarata and Monte Genuardo in the East-West direction, and Monte Carcaci and Monte Telegrafo in the North-South direction, defining a natural border between the Agrigento and Palermo provinces. The area is constituted primarily by a carbonatic complex, constituted by calcareous or limestone-dolomitic rocks which formed during the Mesozoic and Tertiary era.

The Sicani Mountains include 15 Natura 2000 Network sites. For this study were chosen only 11 of these sites since they are located in proximity to each other, while the other sites are more distant (7 CIS, 3 CIS + SCZ and 1 SCZ: see Table 1 and Figure 1). In total, the sites occupy a surface area of 32 871 hectares.

This area is also interesting due to the presence of 4 Regional Natural Reserves (Table 2) listed in the Official list of Italian Protected Areas. The limits of these reserves coincide only in some cases with those of the Natura 2000 Network sites.

Table 1. List of the CIS and SCZ of Natura 2000 Network of the Sicani Mountains.

Type	Code	Denomination	Surface (ha)
CIS	ITA020011	Rocche di Castronuovo, Pizzo Lupo, Gurghi di S. Andrea	1761
CIS	ITA020022	Calanchi, Lembi Boschivi e Praterie di Riena	752
ZCS/CIS	ITA020025	Bosco di Sant'Adriano	6824
ZCS/CIS	ITA020028	Serra del Leone e Monte Stagnataro	3738
CIS	ITA020029	Monte Rose e Monte Pernice	2523
CIS	ITA020031	Monte D'Indisi, Montagna dei Cavalli, Pizzo Potorno e Pian Del Leone	2380
ZCS/CIS	ITA020034	Monte Carcaci, Pizzo Colobria e Ambienti Umidi	1759
ZCS	ITA020035	Monte Genuardo e Santa Maria del Bosco	2631
CIS	ITA040005	Monte Cammarata - Contrada Salaci	2104
CIS	ITA040006	Complesso Pizzo Telegrafo e Rocca Ficuzza	5288
CIS	ITA040007	Pizzo dell'Apa, Bosco di Santo Stefano Quisquina	3111
Total			32 871

Table 2. List of the Regional Natural Reserves of the Sicani Mountains.

Code EUAP	Denomination	Surface (ha)
1123	Monte Cammarata	2049.37
1136	Monti di Palazzo Adriano e Valle del Sosio	5862.07
1137	Monte Carcaci	1437.87
1140	Monte Genuardo e Santa Maria del Bosco	2552.91

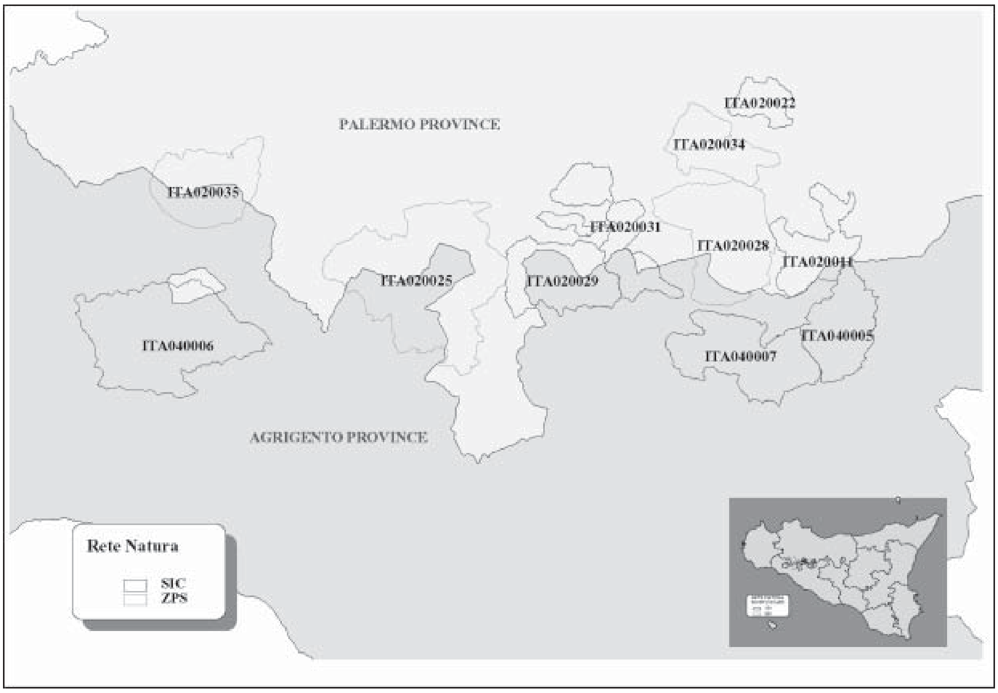

Figure 1. Geographical location of Natura 2000 sites.

The Sicani Mountains are characterised by numerous peaks; the tallest ones are Monte Cammarata (1546 m a.s.l.) and Monte Rose (1436 m a.s.l.).

The prevalently mountainous morphology of the territory causes sudden altimetric changes with narrow river valleys, of clastic nature such as *Valle del Sosio*, while the mountain peaks often show karstic morphologies.

According to the Soil Map of Sicily (Fierotti et al. 1998), in the territory under investigation there are 11 soil associations.

Among these, those most representative are no 4 (Lithic Xerorthents. Rock outcrop. Lithic Haploxerolls), frequent on the calcareous-dolomitic rocks of the sites "*Monte d'Indisi*, *Montagna dei Cavalli*, *Pizzo Potorno e Piano del Leone*" and "*Monte Cammarata – Contrada Salaci*"; no 20 (Typic Xerocrepts. Calcixerollic Xerochrepts. Lithic Xerorthents), present at the sites "*Monte Genuardo e Santa Maria del Bosco*", "*Bosco di Sant'Adriano*" and "*Monte Rose e Monte Pernice*"; no 23 (Typic Xerochrepts. Calcixerollic Xerochrepts. Typic and/or Lithic Rendolls), present at the sites "*Monte Cammarata - Contrada Salaci*", "*Pizzo della Rondine, Bosco di Santo Stefano Quisquina*" and "*Rocche di Castronuovo, Pizzo Lupo, Gurghi di Sant'Andrea*".

The study area shows quite heterogeneous climatic patterns. The mean monthly temperature oscillates between 9 °C and 25 °C, while the mean annual rainfall ranges between 485 mm and 870 mm. According to the bioclimatical classification (Rivas Martinez 1994), the Sicani Mountains have a prevalently "*Mesomediterranean lower subhumid*".

The general features of the flora of the study area are well known, thanks to the collection and the data supplied by different botanists (Gussone 1842–1845; Lojacono-Pojero 1888–1903). A recent study (Marcenò et al. 1985), regarding a wide area of the Sicani Mountains, provides a list of 783 taxa including several endemic or rare species.

The mountain pastures are among the most representative; they preserve many species of particular taxonomic and phytogeographic interest the chasmophytic flora, present in rocky environments, includes many very rare and endemic species (*Anthemis cupaniana*, *Scabiosa cretica*, *Brassica bivoniana*, *Dianthus rupicola*) and endemic or site specific species (*Erysimum bonannianum, Alyssum siculum*). A further increase in the biodiversity is favoured by the presence of screes which encourage the emergence of pioneer species such as *Centranthus ruber*.

Species richness is also increased by the presence of cultivated areas (orchards, vineyards, olive-groves, arable and pasture lands), which give hospitality to many widespread synanthropic species.

3. Materials and methods

A land use map of the Sicani Mountains Natura 2000 sites has been obtained by video interpretation of digital black and white orthophotos (1: 10 000 scale), concerning the period 2000–2002. This first step was followed by an on field investigation to verify the map and classify areas of difficult interpretation.

The legend of the *Atlante dell'uso del territorio* (*Servizio Conservazione Natura, Ministero dell'Ambiente e della Tutela del Territorio 2003*), created to provide guidelines for the management of Natura 2000 Network sites, was used for land use classification.

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This legend has allowed the classification of the natural and semi-natural areas, on the basis of classes derived from a close examination of the Italian fourth level of the Corine Land Cover classification (this is further improvement and detailing based on European CLC third level). Besides, the Corine classes have been joined with the Habitat classification systems used at the European level (Corine Biotopes, Natura 2000, Eunis Habitat Classification - III classification level). Furthermore, from the land use map, the natural and semi-natural community have been linked to the pre-forest and forest types of Sicily (for further information consult the methodology on the regional classification systems of the pre-forest and forest typologies in Italy) (Del Favero 2000; La Mantia et al. 2000, 2001; Marchetti and Cullotta 2003).

Unclassified types that had a purely local character (not classified to regional level) have been inserted as new Types or Variants. The sparse plant communities, such as those of the rocky walls, screes and calanchis, have been inserted as new categories.

A unique code has been attributed to the artificial areas land use categories (artificial surfaces, agricultural surfaces, etc.), considering the surface of the sites as matrices only constituted by forest and pre-forest patches.

Finally, the natural and semi-natural habitats present in the sites (Appendix. I of the Directive 92/43/CEE) were classified following the forest and pre-forest types classification (La Mantia et al. 2000, 2001).

In the Sicani Mountains Natura 2000 Network sites there are 10 habitats of which 2 are priority ones (6210, 6220). The “Pure or mixed *Quercus pubescens* s.l. forests” habitat (92XX) is not inserted in the Directive but it has been considered because it is an important natural or semi-natural community within the Sicilian territory.

Spatial analysis has been carried out on the habitats map, both on the single site and landscape scale.

Analysis has been conducted applying a series of spatial indices (McGarigal et al. 1994, 2002; Riitters et al. 1995) that describe the composition, the shape and the configuration of the patches (Gustafson 1998; Rutledge 2003) that together allow the quantification of fragmentation.

Composition indices, used for quantifying the fragmentation, are the *Number of Patch* (*NP*), *Mean Patch Size* (*MPS*) and *Patch Standard Size Deviation* (*PSSD*).

Shape indices quantify the complexity of the patches, which is of fundamental importance to understand the different ecological processes taking place. These indices are the *Mean Perimeter Area Ratio* (*MPAR*) and *Mean Shape Index* (*MSI*).

Configuration indices measure the degree of connection, maintenance or isolation along and among landscape patches. The notion of connection/isolation is closely related to the “Island Biogeography” theory (MacArthur and Wilson 1967).

The configuration index used is the *Mean Nearest Neighbour* (*MNN*) that measures the distance between two relatively close patches.

Among the inherent spatial composition indices (McGarigal et al. 1994), an important role is played by the *Core Area* (*CA*), obtained by applying different levels of buffer toward the inside of the habitat, increasing 20 m at a time, starting from 0 m (Figure 2) to reach a level at which the habitat terminates (Dawson 1994). Some specific indices have been applied to the *CA* (Table 3).

The *MNN* index and those related to the *CA*, *TCA* (*Total Core Area*) and *TCAI* (*Total Core Area Index*) have been calculated for the habitats corresponding to thermophilous (6220, 5332) and mesophilous (6210) grasslands, shrubs (5330, 5333), deciduous (92XX) and evergreen (9340) oak woods, while these indices have not been calculated for the remaining habitats due to their high fragmentation (evident from the calculation of the preceding indices above mentioned).

Table 3. Habitats of Nature 2000 Network in Sicani Mountains.

Code	Denomination
3260	Water courses of plain to montane levels with *Ranunculion fluitantis* and *Callitricho-Batrachion* vegetation
3290	Intermittently flowing Mediterranean rivers of the *Paspalo-Agrostidion*
5330	Thermo-Mediterranean and pre-desert scrub
5331	*Euphorbia dendroides* – dominated communities
5332	*Ampelodesmos mauritanicus* – dominated garigue
5333	*Chamaerops humilis* – dominated garigue
6210	Semi-natural dry grasslands and scrubland facies on calcareous substrates (*Festuco Brometalia*) (*important orchid sites)
6220	Pseudo-steppe with grasses and annuals of the *Thero-Brachypodietea*
8130	Western Mediterranean and thermophilous screes
8210	Calcareous rocky slopes with chasmophytic vegetation
92A0	*Salix alba* and *Populus alba* galleries
92XX	Pure or mixed *Quercus pubescens* s.l. forests
9340	*Quercus ilex* and *Quercus rotundifolia* forests

4. Results and discussion

4.1 Land use, forest and pre-forest types

The study area present 8 land cover categories (Corine Land Cover II, III and IV level): agricultural area (1123, 23.40% of the total surface area); broadleaved woods (311, 37.80%); coniferous woods (312, 21.70%); natural lawns-pastures and grasslands (321, 8.10%); heaths and shrubs (322, 2.97%); areas with sclerophyllous vegetation (323, 5.30%); open zones with rare or absent vegetation, 4.50% (33).

According with forest and pre-forest types list of Sicily (La Mantia et al. 2000, 2001) there are 10 Categories, 29 Types and 14 Variants.

Natural Woods (Categories 8 and 9) cover a total surface area of 5456 ha and include evergreen oak stands (16.4%) and deciduous oak pure or mixed woods dominated by *Quercus pubescens* s.l. (10.2%).

Artificial Woods (categories 16 and 17) cover a surface area of 7218 ha (23.1%), the 21.7% of these are dominated by conifer, while deciduous exotic broadleaved correspond to 1.4%.

Riparian vegetation (category 15) cover the 9.7% while rocky cliffs (20), screes (21) and calanchis vegetation (22) cover the 4.5% of the total surface area.

Maquis, shrubs and garigues cover a surface area of 1495.9 ha. They have been divided in two categories, “Maquis and garigues of the mesic and/or warm-arid environments” (category 14), which deal the 5.2% of the whole territory and “Maquis and garigues of the mesic and/or warm-arid environments subject to intense succession processes” (Category 2), which cover 2.7%. The grasslands, with a surface of 2547 ha, belong to the second category.

4.2 Natural and semi-natural habitats

The area occupied by the 13 habitats is 17 201 ha, the 53% of the total surface area of the sites (Table 3 and Figure 2).

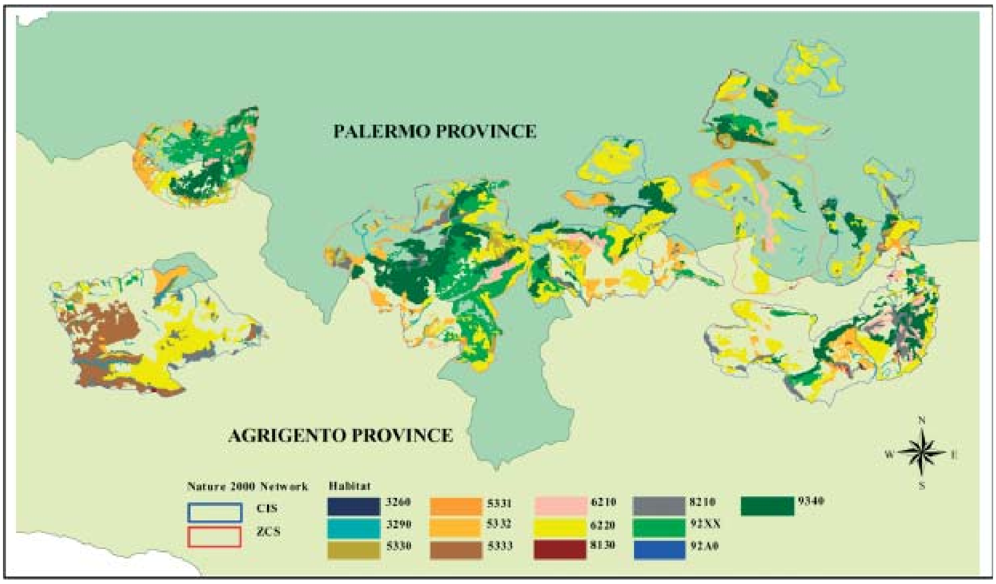

Figure 2. Classification of Natura 2000 habitats in Sicani Mountains.

The most widespread habitats are grasslands (Figure 3) that altogether cover 23% of the total surface area; therophitic grasslands occupy 16.7% of the surface area and they are represented by the habitats 6220 and 5332, that cover 5.1% of the surface area.

The mesophilous grasslands (habitat 6210; cover 2% of the territory) are found in mountain and sub-mountain zones, from more than 1000 m a.s.l.. up to the top of the mountains. The total surface of *Ampelodesmos mauritanicus* (5332) and *Brachypodium* sp. pl. grasslands (6220) is 7166.7 Ha, while the mountain pastures (6210) occupy a rather small surface area (627.4 ha).

6% of the surface area is covered by shrub formations with habitat 5330, habitat 5331 and habitat 5333. They develop where climatic conditions are more xeric, with a total surface area of 1173 ha.

A significant section of the territory (approximately 17% of the whole surface area) is covered by Mediterranean oaks, with a total area of 5624 ha.

Of these, the habitat 9340 covers 10.1% of the surface area (3394 ha), while that 92XX covers 6.8% of the total (2230 ha).

The ridges, rocky walls or screes are, however, covered by 8210 and 8130 habitats; they cover 4% of the territory with a total surface area of 1366 ha.

The habitats with riparian vegetation (3260, 3290, 92A0) cover less than 2% of the total surface area (509 ha).

4.2.2 Spatial analysis

Patch analysis has underlined a high fragmentation level for some of the habitats (3260, 3290, 92A0, 8210, 8130, 5331, 6210). From an ecological point of view, the effects of fragmentation, caused by the new form of landscape use, have caused a large loss of natural habitats, the reduction of the patches remained dimension and the increase of isolation.

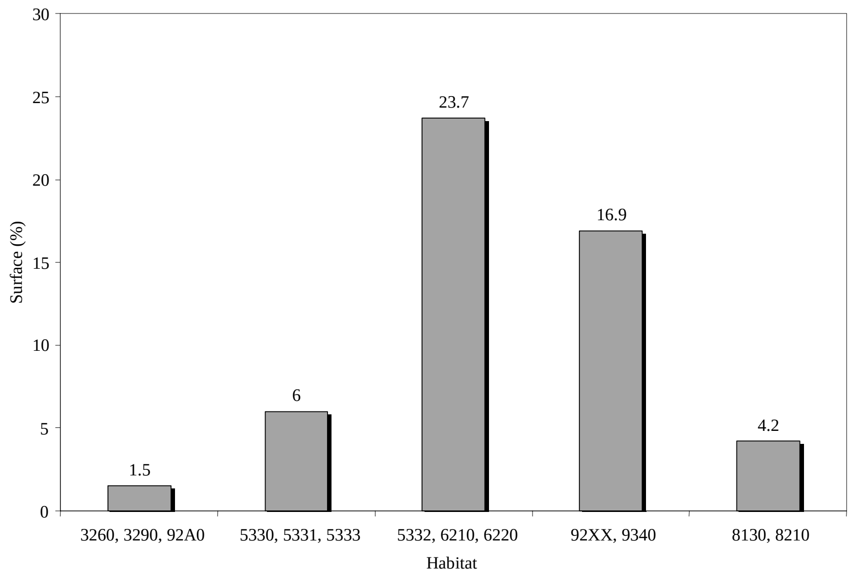

Figure 3. Surface of class habitats.

Sicani Mountains habitats result rather heterogeneous, both in composition and in form; they are inserted in antrhropic environmental context where 25% of the total area are constituted by reforestations, while the agricultural land occupies 22% of the total surface area.

In this study a total of 1303 patches has been analysed, which have a surface area of 17 201 ha. Among the thermophilous grasslands, habitat 6220 is less fragmented although it is subject to intensive human action (fire, agriculture and pasture). It is localized along the border of the sites and is in continuous evolution. In fact, in contrast to habitat 5332, habitat 6220 is constituted by a high number of patches (Table 4; Figure 4) of great size and regular shape: the former *NP* is 307, *MPS* 6 ha (Figure 5) and *MPAR* 311, in the latter, despite the surface area being almost a third of the former, *NP* is equal to 128, *MPS* is 4.5 ha and *MPAR* is greater (350). Habitat 6120 covers a small area (*CA* 627 Ha) although it is located in the centre of the analysed sites, characterizing mountain pastures and areas, which are subject to intensive pasturage. It has a *NP* relatively large (59) and a irregular surface (*MPAR* 409).

A preliminary analysis of the forest habitats has highlighted a greater fragmentation of the deciduous oaks (92XX) compared with the evergreen oaks. In fact, the former covers approximately 7% of the total area, with a *NP* of 151 and a *MPS* less than 5 Ha. The evergreen oaks habitat (9340), however, even though it covers a greater surface (10%), has a lower *NP* (130) and above all, a *MPS* greater than 10 Ha.

The *Standard Patch Size Deviation* (*PSSD*) of the deciduous oaks also shows a higher value compared to that of the evergreen oaks (Table 4); this is due to the small and similar dimension of the deciduous oak patches. This reflects the greater human influence on the deciduous oaks; in fact, compared with evergreen oaks, they have a preference for more fertile soils and result more sensitive to agriculture, intensive pasture etc. On the contrary, *MPS* of deciduous oaks increases on the sites of “Monte Carcaci, Pizzo Colobria e Ambienti Umidi” (13 Ha), “Monte Genuardo e Santa Maria del Bosco” (15 ha), “Bosco di Sant’Adriano” (17 ha) and “Monte Rose e Monte Pernice” (24 ha) where this habitat is found very often and extends over large areas (cover superior to 60%) because the soil is very acid and uneven.

Table 4. Results of landscape indices applied to the habitats.

Code	CA (ha)	P%	NumP	MPS (ha)	MSI	MPAR	MNN (m)	TCA (ha)	TCAI (%)
3260	26	0.1	4	3.1	4.3	656.4	-	-	-
3290	3.72	1.1	125	1.8	3.8	982.6	-	-	-
5330	829	2.5	121	2.3	1.7	455.6	314.2	104.5	15.2
5331	139	0.4	28	4.4	1.7	363.1	707.5	16.5	13.7
5332	1672	5.1	128	4.5	1.9	350.2	231.0	408.2	26.7
5333	1026	3.1	15	17.6	2.0	259.9	329.3	324.3	38.2
6210	627	1.9	59	2.9	1.8	409.2	166.0	99.3	18.8
6220	5494	16.7	307	6.0	1.9	311.9	108.0	1424.4	38.7
8130	51	0.2	28	1.2	1.6	507.9	-	-	-
8210	1314	4.0	195	1.8	1.8	556.1	-	-	-
92XX	2230	6.8	151	4.7	1.9	367.2	156.3	514.9	30.6
92A0	111	0.3	12	5.9	4.5	600.9	-	-	-
9340	3304	10.1	130	10.3	1.9	262.2	214.2	787.2	27.0
Total	17201	51.97	1303	-	-	-	-	6761.0	109.0

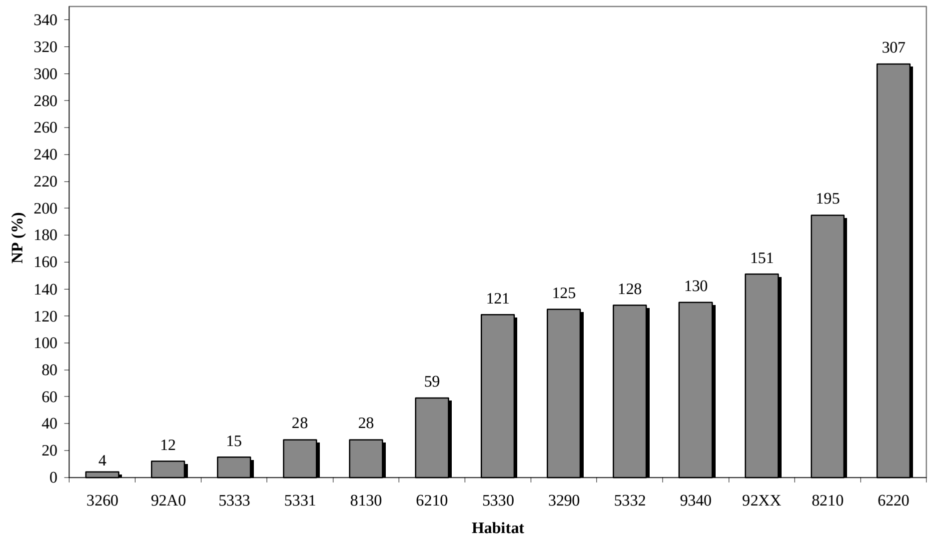

Figure 4. Number of patches (NP) of each habitat.

The shrubs (5330; 5331; 5333) show rather heterogeneous and complex results. Habitat 5330 is constituted by a *NP* equal to 121, a *MPS* of approximately 2 ha and a *MPAR* of 455. *Euphorbia dendroides* shrubs (5331) are present at sites characterized by thermophilous rocky environments where the patches are small and fragmented (Table 4). The habitat constituted by *Chamaerops humilis* scrubs (5333) represents a particular situation and it is present only at the "*Monte Telegrafo e Rocca Ficuzza*" site. It covers 20% of the site with few and compact patches.

Habitats 8210 and 8130 are less representative because they cover only small and discontinuous areas (respectively 4% and 0.2%).

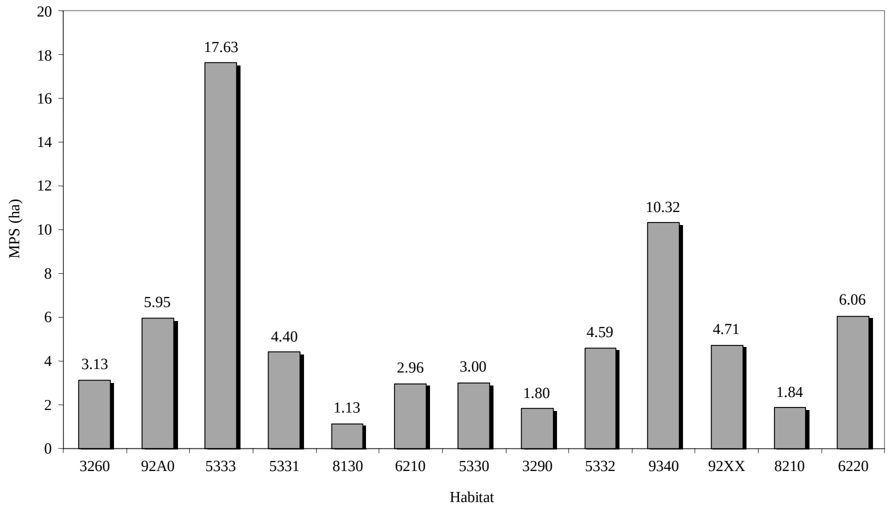

Figure 5. Mean Patch Size (MPS) of each habitat.

From the analysis of habitat composition of the grasslands and the evergreen oaks, it is deduced that these habitats are less fragmented in comparison with the others; the deciduous oaks are more fragmented and more vulnerable; the thermophilous shrubs (5330) and the *Euphorbia dendroides* shrubs (5331) are reduced to small areas with an elevated degree of fragmentation.

Concerning the shape, habitats, on the landscape scale, introduce a *Mean Shape Index* value greater than 1 (Table 4), highlighting an irregular mean patch shape. In particular, the most irregular patches are those constituted by riparian vegetation (3260, 3290, 92A0), being linear formations and therefore very long. Results of landscape indices applied to the habitats patches (5330, 5331) are less irregular in shape compared with the broad-leaved forest (92XX, 9340). The patches of *Chamaerops humilis* – dominated garigue (5333) are, however, more irregular than the other shrubs.

The index relating to the patch configuration (MNN) has been calculated, on the landscape scale, for grasslands, shrubs and broad-leaved forest habitats, excluding riparian vegetation (3260; 3290; 92A0), the thermophilous screes communities (8130) and the chasmophytic vegetation (8210) because these habitats show a more irregular shape (long and narrow).

This index, which measures the distance between the patches, has revealed a high value for the shrub habitat, particularly for the habitat 5331, while habitats 5332 and 5333 show lower values (Figure 6).

The shorter distance between the patches is found for thermophilous (6220) and mesophilous (6210) grasslands, and for the deciduous oaks (92XX). The latter are shown to be more connected than the evergreen oaks (9340) despite occupying a smaller surface area and being more fragmented.

4.2.2 Core Area

In the thermophilous grasslands habitat (5330) the reduction in *Core Area* (*CA*) is gradual (Figure 7) until a buffer of 200 m, after which there is a sudden change.

The *CA*, reduced by approximately one third at 200 m, has a drastic decrement varying from 8.45 ha to 3.15 ha; it is reduced to zero when the buffer reaches 280 m.

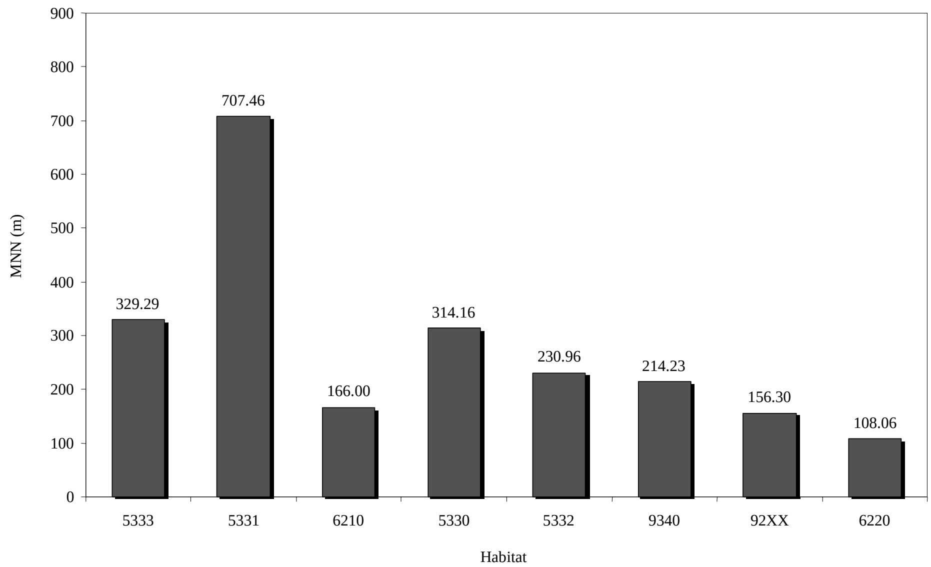

Figure 6. Mean Nearest Neighbour (MNN) of each habitat.

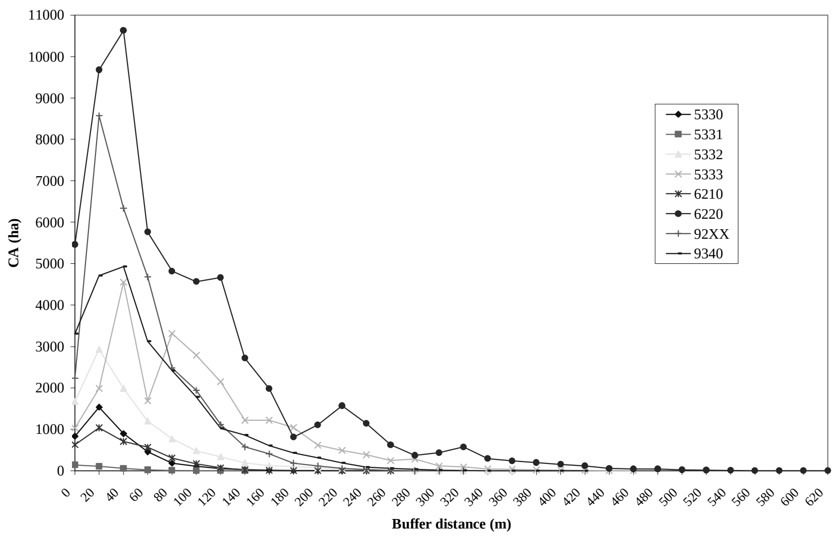

Figure 7. Change of Core Area (CA) of each habitat.

Habitat 5331 is present in thermophilous environments with a surface area of 139 ha. The progressive increase in the buffer causes irregular diminution; with the buffer at 20 m the habitat is already halved. Between 60 m and 80 m of buffer the *CA* reduces from 20 ha to 6 ha and the *TCA* from 16.57 ha to 6 ha creating a critical situation for the habitat that, however, it is reduced to zero with a buffer of 140 m.

In the "*Ampelodesmos mauritanicus* – dominated garigues" habitat (5332) the reduction of the *CA* is gradual, reaching critical values between 260 m and 280 m of buffer, where the surface is reduced from 15 ha to 6 ha and subsequently it terminates for 360 m of buffer.

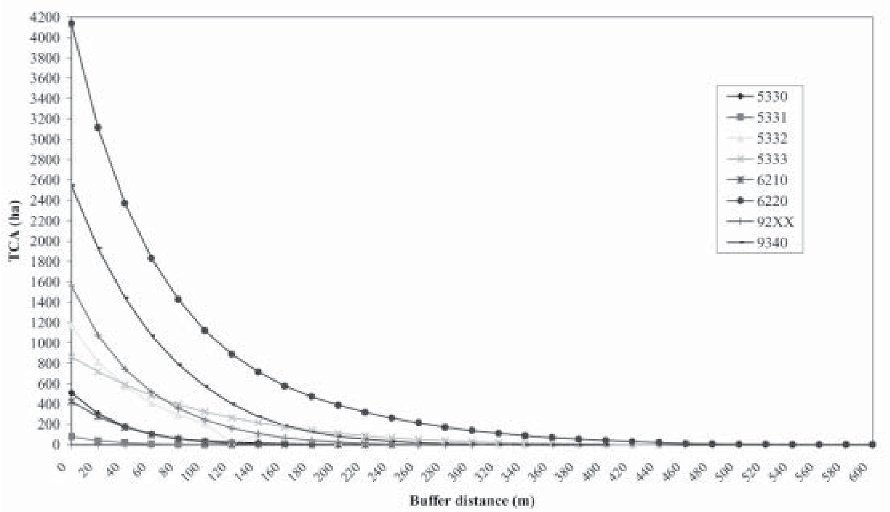

Figure 8. Change of Total Core Area (TCA) of most representative habitats.

The mesophilous grasslands habitat (6210) reduces slowly up to 120 m of buffer followed by a sudden change for the next buffer. The *CA* reduces from 76 ha to 19 ha and the *TCAI* reduces from 7% to 3%. The habitat terminates when the buffer reaches 260 m.

The deciduous oaks habitat (92XX) also shows a gradual reduction for elevated buffers only reaching critical thresholds between 240 m and 260 m of buffer, where the *CA* reduces from 24 ha to just 6 ha. The *TCA* is reduced from 9.64 ha to 2.72 ha and the *TCAI* from 1.23% to 0.40%. The habitat surface is reduced to zero with a buffer of 320 m.

The thermophilous grasslands habitat (6220) has quite a discontinuous *CA* trend. Among the habitats analysed, habitat 6220 most withstands external disturbances; in fact, it is the habitat that terminates with the highest buffer (620 m).

The *Total Core Area* (*TCA*) (McCarigal and Marks 1995; Schumaker 1996) has a hyperbolic type trend for all of the habitats, showing a regular decrement only for some (Figure 8).

For the 6220, 92XX and 9340 habitats the curve decreases quickly and gradually, slowing between 140 m and 200 m of buffer, then followed by a slow and long descent. The behaviour of these three curves denotes a certain tendency to be attacked by external phenomena such as grazing, agriculture or fires in these habitats, but also the elevated ability to withstand such disturbances forming rather compact, dense and regular surfaces.

Habitat 5332 shows a steep, but not gradual, trend that suffers a collapse at 140 m of buffer (Figure 7). This can be explained by the lower integrity of this habitat, compared to those above, and from a greater susceptibility to human disturbances.

Habitat 5333 has a slow and long regression, due to an extended and compact *CA*. Furthermore, given its isolated orographic position, the only significant disturbance is grazing.

Habitat 6210 has a similar trend to 5333, but given its modest surface area, the *CA* is reduced to zero very quickly.

Habitat 5330 only has a strong decrement between 20 m and 60 m of buffer. This phenomenon is caused by the high dynamism of this habitat that represents a community evolving towards more stable formations such as forest.

Habitat 5331 shows a gradual and short decrement terminating quickly because of the precarious ecological conditions in which it develops.

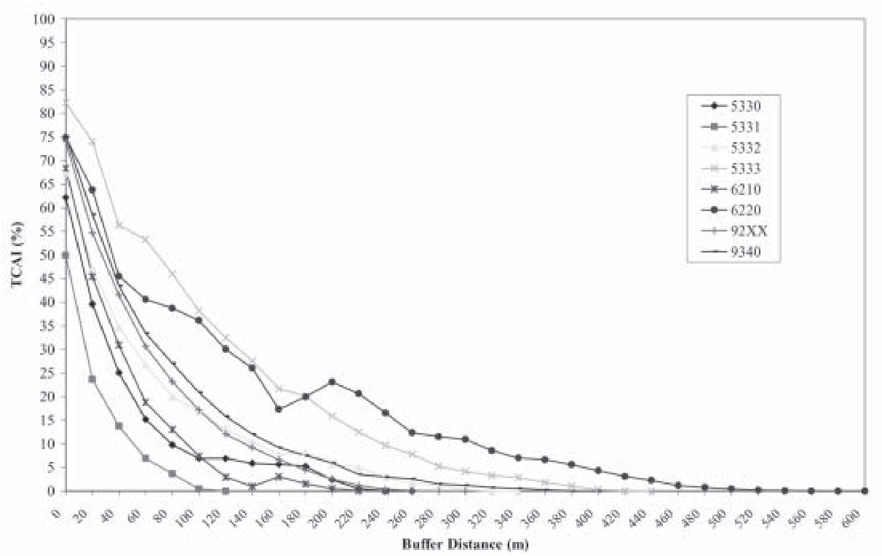

Figure 9. Change of Total Core Area Index (TCAI) of most representative habitats.

The *Total Core Area Index* (*TCAI*) (McCarigal and Marks 1995) has a discontinuous trend in almost all of the habitats, except for forest habitats (92XX and 9340) where the decrement is slow and gradual (Figure 9).

Habitats 5332 and 5333 have an almost regular trend. Habitat 6220 first has a steep decrement, up to 60 m of buffer, then gradually decreases until an abrupt descent between 160 m and 180 m of buffer; it subsequently shows a sudden rise followed by a slow and irregular descent. Habitats 6210, 5330 and 5331 show a steep descent for small values of buffer, but, while 5331 decreases until its termination, 5330 is stable between 140 m and 200 m of buffer before then quickly decreasing and terminating. Habitat 6210, however, has a quickly decrement between 140 m and 180 m of buffer before continuing its brief descent.

Core Area analysis has highlighted the greater vulnerability of habitats with limited surface area, irregular shape and peripheral position within the site such as “Thermo-Mediterranean and pre-desert shrubs” (5330) or “*Euphorbia dendroides* – dominated communities” (5331), compared to those that occupy regular, central and vast surfaces such as “*Chamerops humilis* – dominated garigue” (5333) “Oaks to *Quercus ilex* and *Q. rotundifolia* forests” (9340), “*Ampelodesmos mauritanicus* – dominated garigue” (5332) or the thermophilous grasslands (6220) that represent the least fragmented habitat. *Quercus pubescens* s.l. forests (92XX), despite intense disturbances, have a moderate fragmentation; the mesophilous grasslands (6210) show similar patterns.

5. Conclusion

Landscape can be read, in ecological terms, as a group of interacting ecosystems of various dimensions. Accordingly, the analysis of its components is an essential indicator to understand its level of fragmentation.

From an ecological point of view, the excessive fragmentation, which causes the reduction of the dimensions of the remaining patches and an increase in isolation due to the new forms of land use, is a dynamic process that could cause conspicuous loss of habitats within a landscape especially when the surface of the patches does not guarantee their ecological functionality.

The natural and semi-natural habitats represent heterogeneous areas in which human activities have caused some changes over time that have limited their surface area, reducing them to small distant fragments (Gkaraveli et al. 2001).

Spatial analysis of the habitats present in the investigated sites, has allowed an understanding of the spatial organisation of the single patches that compose such habitats, to evaluate the composition and the degree of aggregation of the territory. The applied landscape indices have underlined a different degree of fragmentation within the natural and semi-natural habitats of the Sicani Mountains sites; in fact, they are very heterogeneous, both in composition and in shape, and inserted in an almost natural environmental context, where 23% of the total area is covered by forest plantations, while the agricultural surfaces only cover 22%.

The comparison between the different classification systems, adopted in the present research, has been useful for this preliminary study aiming to evaluate the state of the habitats of the Sicani Mountains Network. The database, constituted by land use, forest and pre-forest types, is a tool of fundamental importance for the Natura 2000 sites planning and management.

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Extracting Forest Patch Attributes at the Landscape Level Using New Remote Sensing Techniques – An Integrated Approach of High-Resolution Satellite Data, Airborne Lidar Data and GIS Data for Forest Conservation

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Abstract

Methods for identifying attributes of forest patches using high-resolution satellite data and airborne lidar data at the landscape level and for integrating this information with GIS data were presented. GIS data concerning forest management units are useful for understanding the disposition of forest patches, but the attribute data are sometimes unreliable because of the delay of data renewal and the lack of confirmation in the field. High-resolution satellite data and airborne lidar data were used to identify forest patch attributes. For high-resolution satellite data, a local maximum filter was used to detect treetops in plantations and profiles of digital numbers along hypothetical lines that were set on forest patches were compared to comprehend structures of diameter classes in natural forests. Airborne lidar data were used to obtain three-dimensional information. Individual tree height was determined from DCM (digital canopy model) derived from both first and last pulse data of laser scanning. Adjoining spaces with 1 m wide along a certain direction in the stand were assumed and all airborne lidar data within every space were projected to a corresponding vertical plane to comprehend a canopy structure. These attributes obtained from new remote sensing techniques were used to renew attribute data for GIS.

Keywords: *airborne lidar data; high-resolution satellite data; local maximum filter; vertical projective plane; GIS*

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1. Introduction

Forests have various functions such as conservation of biodiversity, water conservation, carbon dioxide fixation, material production, and regeneration. However, some plantation areas have been abandoned after final cutting in Japan due to the decline of domestic timber prices against imported ones. Soil erosion and consequent loss of biodiversity are our concern at present because such areas are normally characterized by steep slopes. The patch structure of a forest, such as disposition, size and context, influences its regeneration. It is very important to understand patch structure when we consider forest conservation at the landscape level. The rapid construction of GIS data concerning forest management units which were generated from forest maps and forest registers has given us some understanding, but attributes about stand factors are not necessarily reliable because they are sometimes input without field confirmation.

Since first launch of the earth observation satellite, Lansat-1 in 1972, satellite remote sensing has played an important role in environmental monitoring. Despite their advantage of having been obtained from wide and periodic observations for a long time, satellite remotely sensed data are rarely employed for forestry. One of the reasons is that we cannot interpret individual tree crowns from these data, either visually or automatically. Recently, they have been utilized for national forest inventories for the Kyoto protocol and for assessing forest biodiversity at the regional, national and continental levels.

Advances in sensors and techniques have opened a new era for taking forest inventories and assessing forest biodiversity with remote sensing at the landscape level (Pitkänen 2001). Commercial high-resolution satellite data such as IKONOS or QuickBird data are expected to identify forest patch attributes (Flanklin et al. 2001). Ground resolution of these satellites has reached one meter. As a result, observations from space have shifted from forest stands, clusters of trees or mixture of some land-cover types to individual trees.

Lidar (light detection and ranging) remote sensing is an expansion from two-dimensional observation to three-dimensional measurement and we expect to improve its accuracy to estimate the potential of productivity and biomass through the acquisition of three-dimensional data. There has been growing interest in the utilization of lidar remote sensing in forestry. Pervious studies showed that stand parameters such as tree height, number of stems, and stand volume could be estimated accurately (Magunussen et al. 1999; Næsset and Bejerknes 2001; Næsset and Økland 2002). Recently, individual tree attributes were also derived from airborne lidar data with high sampling density (Hyyppä et al. 2001; Persson et al. 2002; Brandtberg et al. 2003). It has been also applied to ecological studies (Hinsley at al. 2002; Lefsky et al. 2002). Thus, ecologists are now interested in lidar remote sensing as a tool for expanding their studies to spatial understandings.

We can get accurate and detailed information from airborne lidar data but it is more expensive than other data. The utilization of high-resolution satellite data is occasionally restricted by atmospheric condition or topography. Therefore, the combined utilization of multi-source data is cost-effective to understand forest patch structure. Here, we present methods for identifying attributes of forest patches using commercial high-resolution satellite data and airborne lidar data at the landscape level and for integrating this information with GIS in the study.

2. Materials and methods

2.1 Study area

The study area is located in national forests managed by the Shimanto District Forest Office in the Shimanto River basin, Shikoku Island, Japan. 86% of the area is covered with forests, 68% of which are artificial. Sugi (*Cryptomeria japonica*) and hinoki cypress (*Chamaecyparis obutusa*) trees have been planted actively since the end of World War II because of the increase of timber demand, and natural forests have been increasingly segmented. These areas are characterized by steep slopes and complicated patch structures that are formed by broad-leaved forests and plantations.

36 study plots with the size of 20 m × 20 m were established in evergreen broad-leaved forests and plantations by vertical zone to investigate the efficiency of high-resolution satellite data for identifying forest patch attributes. DBH (diameter at breast height) of whole trees was measured and species were recorded. Stand density was calculated in each plot based on the number of standing trees per 0.04 ha.

The study plot for airborne lidar data was set up in the Ichinomata national forest at approximately 600 meters above sea level. The study plot included an evergreen broad-leaved forest and a plantation of hinoki cypress. The plot area was 1.5 ha (150 m × 100 m). A differential GPS was used to determine the geographic coordinates of four corners of the plot. A 100 m × 100 m area of the plot is occupied by a plantation of hinoki cypress, which is divided by a ridge from the rest, which is an evergreen broad-leaved forest. DBH of whole trees was measured and species were recorded. Tree heights in half of the plot were measured to examine the accuracy of DCM derived from both first and last pulse data of laser scanning.

2.2 GIS data and remotely sensed data

GIS data were generated from forest maps and forest registers for whole national forests of the Shimanto District Forest Office.

We used two kinds of remotely sensed data in the study; high-resolution satellite data and airborne lidar data. Panchromatic data of IKONOS and QuickBird satellite with 1-meter ground resolution were acquired from 2001 to 2003 in the area and a total of 7 cloud-free images were prepared for the study. The data were geo-registered to the UTM coordinates.

The ALMAPS (Asahi Laser Mapping System, Aero-Asahi Co., Tokyo, Japan), which consists of the ALTM 1225 laser scanning system (Optech, Canada), the GPS airborne and ground receivers, and the inertial measurement unit (IMU) reporting the helicopter’s roll, pitch and heading, was used to acquire the lidar data. The laser scanner system transmits the laser pulse at 1064 nm (near-infrared) and receives the first and last echoes of each pulse. The elapsed time between transmission and reception is measured to calculate the distance between the system and the object. The position of the helicopter and the scan angle are calculated using GPS and IMU. The distance between the helicopter and an object is measured by the laser range finder. Kinematic GPS is used after the flight to determine the position of the helicopter with high accuracy.

The airborne lidar data were acquired on 28 October 2002. We selected a helicopter as a platform because it has advantages for flight speed and flight altitude to acquire intimate data. The flight altitude of the helicopter above the ground was about 250 meters and the average flight speed was approximately 14 m/sec. The pulse repetition frequency was 25 kHz and the scan frequency was 25 Hz. Maximum scan angle (off nadir) was 12°. The beam divergence was 1.2 mrad. Overlap of scanning between neighboring flight lines was about 40%. Measurement density was 26.6 points/m^2. Therefore, the footprint diameter was approximately 30 cm and the average distance between neighboring footprints was about 20 cm. Both first and last pulse data were acquired to reconstruct forest canopy structure and topography.

2.3 Identification of attributes

Stand density of a plantation is a key factor to understand forest patch attributes. We can estimate tree height and stand volume from it if we know stand age from other information sources, such as GIS data. While aerial photographs have been used to interpret stand factors for a long time in forestry, there is a limit to its use because experts are required to interpret stand factors from them.

We are anticipating that the digital analysis of high-resolution satellite data will supplant the interpretation of aerial photographs. In the present study, we applied a local maximum filter for panchromatic data of a high-resolution satellite to estimate stand density (Hirata et al. 2002). If a digital number of a certain pixel is the biggest of surrounding pixel numbers, the local maximum filter changes this pixel value to 1. If the biggest value is found in surrounding pixels, this pixel value is changed to 0. The biggest value shows the brightest point in the interest area, and it can be considered to correspond to the top of a tree. The applicability of this filter depends on the distance between tops of neighboring trees. Theoretically, a stand density of less than 2500 individual trees (50 × 50 individual trees) per hectare could be detected if we would apply a 3 by 3-local maximum filter to 1-meter resolution data, but each tree does not grow straight in a real stand. In addition, suppressed trees cannot be detected from the processing.

We used a profiling method to understand canopy structure in evergreen broad-leaved forests. The compositions of diameter classes and their species are considerably different even in stands with similar stand density. The acquisition of a profile of digital numbers in panchromatic data from a high-resolution satellite along a hypothetical line in a forest patch makes it possible to estimate its attributes. A wavelength in a profile of digital numbers shows canopy structure, and amplitudes in a profile are not important because they fluctuate due to many factors such as sun elevation, topographic conditions, and atmospheric conditions. Trees of large diameter classes have large crowns, so if a hypothetical line passes through these crowns, long wavelengths are found. If a profile passes on the edge of crown, the wavelength becomes short. Therefore, we could find repetition of short wavelengths in young stands such as regenerated secondary forests or poor stands. On the other hand, a mixture of short and long wavelengths is found in mature or old-growth stands. Some hypothetical lines were drawn on high-resolution satellite data to take profiles in forest patches involving study plots and these profiles were investigated.

Accurate three-dimensional data of first and last pulses for the plot were obtained from airborne lidar data with post-processing. A DEM (digital elevation model) with 50 cm-pixel size was produced having a minimum value of the last pulse data in each pixel. A DSM (digital surface model) with 50 cm-pixel size was generated having a maximum value of the first pulse data in each pixel. A DCM (digital canopy mode) was calculated from the difference between the DSM and the DEM. A local maximum filter was used to detect treetops. The value of the DCM at each treetop was regarded as individual tree height. Tree height derived from airborne lidar data was compared with tree height measured in the plot.

Adjoining spaces with 1 m wide along a certain direction in the stand were assumed and all first and last pulse data within every space were projected to a corresponding vertical plane to comprehend the canopy structure. A series of vertical projective planes with scattering points

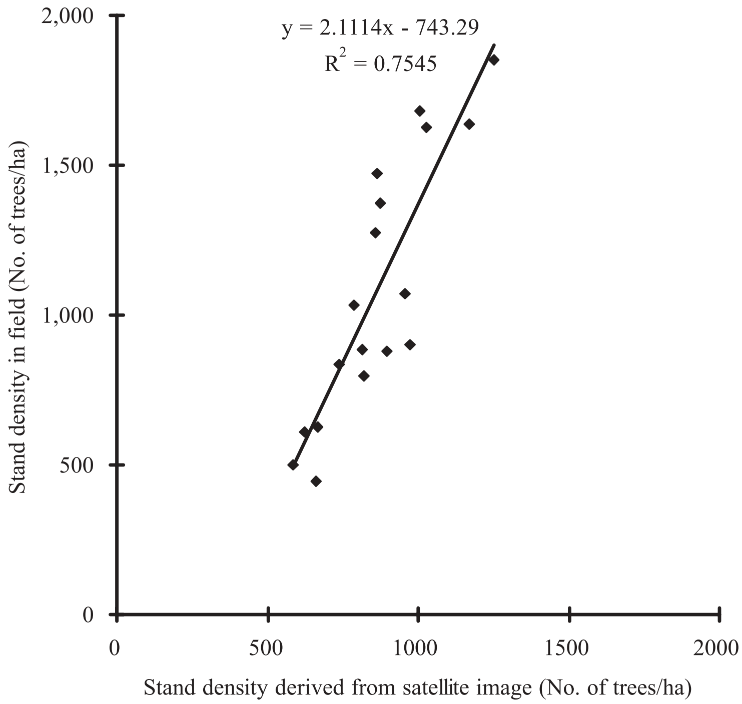

Figure 1. Estimated stand density of study plots plotted against field measurements.

were prepared and the domain of dense points was identified as canopy layers and topography in each plane (Hirata et al. 2003; Hirata et al. 2004).

3. Results and discussion

Treetops were detected using the local maximum filter in forest patches involving study plots in plantations. Stand densities of the 18 study plots in plantations were regressed against the predictor variables derived from high-resolution satellite images (Figure 1). The coefficient of determination (R^2) was 0.75 and the correlation coefficient was 0.87. The estimation of stand density was adequate in mature stands, while it was underestimated in young stands. In the mature stands, the crowns of individual trees have enough area and their forms are sharp, therefore the difference of brightness at the top of a crown and in its edge is quite large. On the other hand, some treetops do not appear to have a canopy surface or they cannot be distinguished from each other in young stands. The filter processing to detect stand density worked well in stands of approximately less than 1000 individual trees / ha.

Profiles of digital numbers on high-resolution satellite data along lines set into forest patches involving study plots were compared with forest patch attributes such as distribution of DBH in evergreen broad-leaved forests (Figure 2). As a result, it was found that composition of wavelengths strongly had relation to the composition of diameter classes.

Both results showed that high-resolution data could be applied to estimate forest patch structure and it was possible to separate forest patches with different structures in terms of density, species, and distribution of DBH for forest management and evaluation forest functions for conservation purposes.

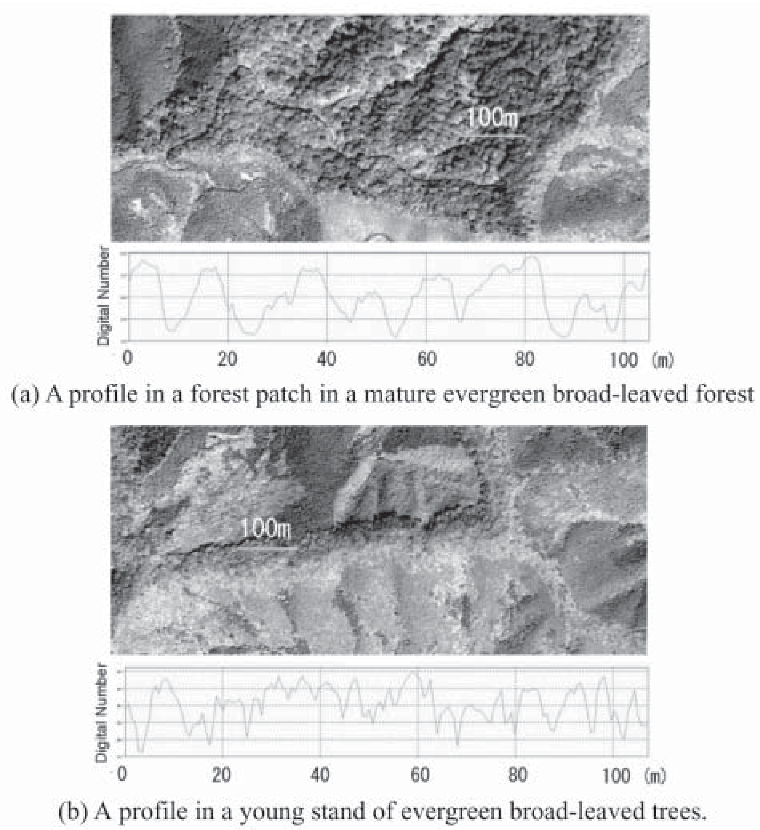

Figure 2. Profiles of digital number on high-resolutions satellite data.

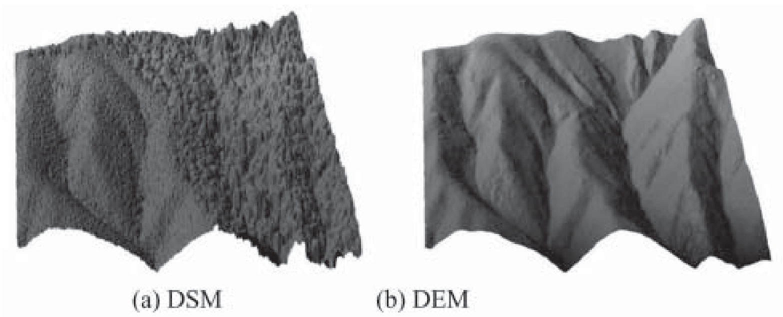

Figure 3. DSM and DEM derived from the first and last pulse data.

DSM and DEM for the study plot were generated from the first pulse data and the last pulse data respectively (Figure 3). Figure 4 shows DCM with gray scale and a contour map derived from DEM for the study plot. This figure demonstrates that the trees in an evergreen broad-leaved forest have large crowns and the height of hinoki cypress trees depends on topography. By overlaying DCM on DEM, it was found that the growth of trees along the valley was better than the growth along the ridge in the hinoki cypress stand. In Figure 5, tree height derived from airborne lidar data was plotted against tree height measured in the field for sample trees. The line fitted to the data using the least-squares method had a slope of 0.89

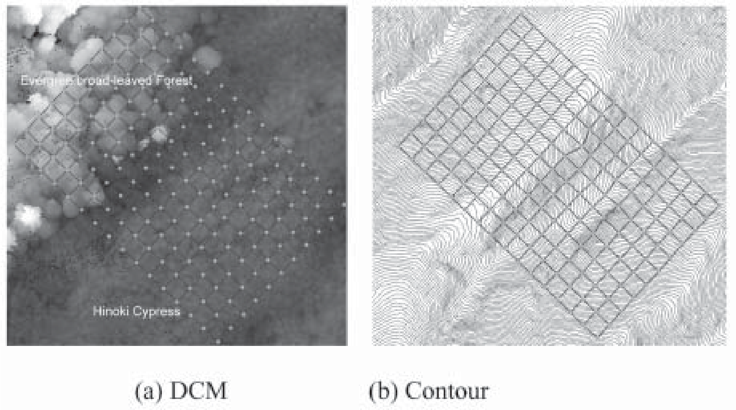

Figure 4. DCM with gray scale and Contour derived from DEM for the study plot.

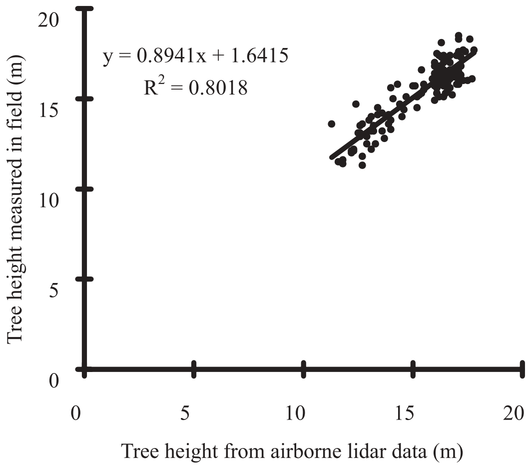

Figure 5. Estimated tree height of 143 sample trees plotted against field measurements.

and an intercept of 1.64 m. The coefficient of determination (R^2) was 0.80 and the correlation coefficient was 0.90. The differences between tree heights derived from the DCM and ones from field measurements in the hinoki cypress stand were quite small. On the other hand, the differences between tree heights derived from the DCM and ones from field measurements in the evergreen broad-leaved forest was very large. This reason was that broad-leaved trees normally leaned to the valley side on a steep slope. We have to note that each crown height in DCM was overestimated on the valley side because of topography.

Using a series of 100 vertical projective planes that were prepared (Figure 6), the canopy layer could be interpreted and distinguished. The stratification of canopy layers could be analyzed by extracting dense domain of scattering points from each projective plane and connecting them according to geographic coordinates. Gap structure, which is closely associated with regeneration, was also clear from these planes. The understory vegetation could also be identified by interpretation of scattering points in them. This approach made it possible to investigate ecological phenomena spatially.

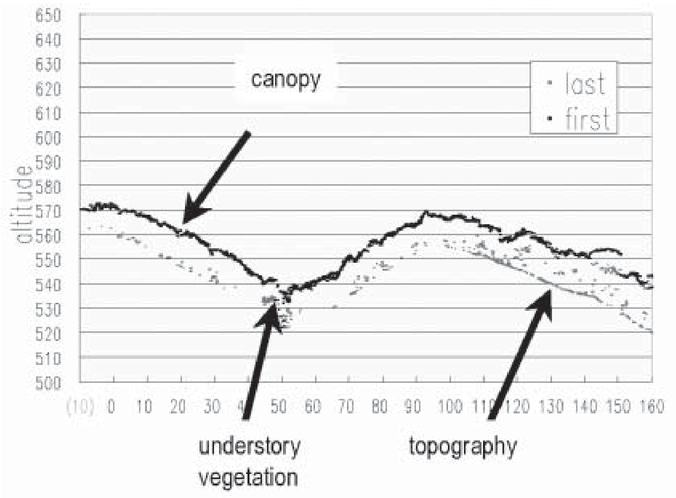

Figure 6. All first and last pulse data within every space were projected to a corresponding vertical plane to comprehend the canopy structure.

4. Concluding remarks

We have described new remote sensing techniques for understanding forest patch attributes at the landscape level. Using high-resolution satellite data, stand density could be detected in old-growth and mature stands, but it was underestimated in young stands. The structure of diameter classes in natural forest could be estimated from it. In GIS data, information concerning natural forest is often unreliable because of the difficulty of forest inventory. Therefore, identification of stand attributes using the satellite data is very useful and often effective to renew the GIS data.

Individual trees could be extracted and the heights could be derived from airborne lidar data. The stand structure, particularly canopy structure as well as understory vegetation, could be identified from the data with high sampling density. The knowledge concerning it becomes important for biodiversity conservation.

These results indicated that there are several opportunities to use high-resolution satellite data and airborne lidar data as the alternative method of forest inventory and for the renewal of GIS data concerning forest management units.

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Birds and Mammals as Indicators of Changes in Biodiversity in Italy

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Abstract

Some animals require a particular habitat, and the occurrence of these animals in a place can be an important indicator as to how animal species are related to individual ecosystems. Birds by their wide distribution and the abundance of their species can be used as indicators of environmental quality as there is a close connection between the overall biodiversity of an environment and the complexity of bird populations. Birds are also important ecological indicators. Mammals occur in various types of environment and this has favoured their use as indicators of biodiversity. The overall aim of bioindicator studies is to develop a method based on interdisciplinary co-operation to set about organising and managing the environment.

Keywords: biodiversity; monitoring; ecological indicators; functionality ecosystem.

Introduction

The United Nations Conference on Environment and Development, UNCED, was held in Rio de Janeiro, Brazil, from 3 to 14 June 1992. One of the results it achieved, the Convention on Biological Diversity (CBD), was particularly important. The aim of this Convention was to preserve biological diversity (as its name suggests), to foster the sustainable use of the components of biological diversity and to work for the fair and right distribution of the benefits deriving from the use of genetic resources.

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According to the CBD, biodiversity means: ‘the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems.’

The twentieth century has seen a marked reduction in biodiversity, due to various factors such as the spread of agriculture, large-scale deforestation and profound anthropogenic changes in the natural habitat.

Research conducted in France shows that since the start of the 20th century there has been a 30% reduction in plant species (of which 12% is threatened with extinction), while 40% of floricultural plants, 52%, of mammal species, 40% of bird species and 40% of reptile species have disappeared (Hermitte 1992).

A recent report of the International Union for the Conservation of Nature (IUCN) states that world-wide 11 167 animal and plant species are threatened with extinction, 121 more than in 2000. The animal species at risk are 1137 mammals, 1192 birds, 742 fish species, 293 reptiles, 157 amphibians, and 1932 invertebrates.

The situation appears even more serious if it is considered that all animal species, even those that seem less important, may be fundamental in maintaining the functional properties of a system, and may modify it drastically if they should disappear. Natural systems are diversified and highly evolved wholes. Each species in such a system has adapted itself to fill a special role in it, and the entire system depends to a certain extent on each species to fill its role. It follows that any reduction in the biodiversity of a system will disturb its equilibrium and may impair its functioning (Wilson 1988). At the same time it is also true that if some species disappear or are reduced in numbers, others take over their ecological role, so that the system as a whole continues to function. This functional complementarity does not however mean that species can be considered supernumerary and hence that some are expendable. They have different but not identical roles in the ecosystems, so that the functional complementarity permits the renewal of the structure and the resume of the evolutionary process after the ecosystem has been changed.

Apart from the various definitions so far given, biodiversity is also from an ecological point of view the outcome of a competition between living species for survival. On this competition then depends the whole mosaic of plant and animal species that make up each ecosystem. It is a process that naturally evolves towards an ever more refined adaptation of the individual to its environment (Rosa 1988).

Biodiversity generally is described, quantified, managed and exploited at three levels. The first of these levels considers variation in the genetic patrimony between individuals of different populations of a species, and between the populations within that species; the second, variation between species, which ascertains the name, relative frequency or rarity, and the endemicity of a species; and the third, variation between ecosystems and the manner in which species interact with each other and with the environment in which they live (Burley 2003).

Italy has not yet approved a national plan for biodiversity, but there have been two reports, dating from 1998 and 2001, on how to implement the Convention. The work carried out in connection with these reports has included the identification, monitoring, and evaluation of biodiversity in Italy and a study of plants and animals that can be used as indicators of biodiversity.

The aim of this paper is to show how some higher vertebrates (birds and mammals) can be used as bioindicators to monitor habitats and the countryside. Animal ecology studies how animal species are related to individual ecosystems. Some animals require a particular habitat, and the occurrence of these animals in a place can be an important indicator that such a habitat exists there.

Fauna and environment in Italy

Italy has a notable diversity of natural landscapes, due in large part to the length of the country, which comprises mountainous areas and a wide variety of other geographically diverse regions. This has made it rich in animal and plant species, so that it was, and perhaps still is, the country in Europe that has the greatest biodiversity from a genetic, specific and ecosystematic point of view. To give one example of this, during the breeding season there are 514 species of nesting birds in Europe, of which 240 are found in Italy. Europe counts 145 species of terrestrial mammals (insectivores, bats, rodents, lagomorphs, carnivores and artiodactyls) of which 95 are also (or exclusively) native in Italy.

From the earliest times parts of Italy have been inhabited by human communities who exploited the natural resources and adapted the environment to their own needs. Present-day biotic communities are thus the result of an evolution of techniques of cultivation in field and forest going back thousands of years.

Since the middle of the last century agriculture has gradually undergone two distinct changes, one relating to the lowlands, the other to the more hilly and mountainous areas. The lowlands have seen an intensification of production, with the increasing importance of industrial agriculture, which has drastically reduced tree and brushwood cover and forage cultivation, while in the higher areas there has been a gradual abandonment of farming, especially animal husbandry, and silviculture; and this has led to a rapid diminution of open fields and fields used for pasture, with a substantial increase in forested areas and ecotones.

Forests have also undergone profound changes in flower composition with a considerable reduction in the genetic variability of individual tree species. Broadleafs represent more than 80% of all forests and stands of varying aspect and complexity, from high forests of beech to the mediterranean maquis dominated by holm oak, at a time when agricultural changes and reclamation have led to the almost total disappearance of riparian forests and lowland stands of mixed broadleafs.

The changes and reduction in the forest cover have also had a strong impact on the species composition of the fauna, with a marked reduction in ‘specialist’ species limited to specific habitats, and an increase in the more opportunist and adaptable species that profit from the changes that have taken place. This has modified the carrying capacity (K) of these habitats, which has in turn led to substantial changes in the quality and quantity of the wildlife populations.

The slow natural evolution of agricultural lands favoured the formation of associations between plants and animals that adapted well to semi-wild environments, such as that of field-and-pasture. This *habitat* ensured a high biodiversity and the preservation of animals and plants characteristic of the habitat.

The gradual diminution of the rearing of animals in the wild or semi-wild state, and the cessation of traditional agricultural activity has strongly reduced, and in some cases led to the disappearance of animals associated with field-and-pasture. At the same time there has been a significant increase in the numbers of some species of ungulates typical of forest environments and ecotones: red deer, roedeer and boars. These animals benefited from the abandonment of extensive marginal lands and from the increase in forests.

In this way there has been a substantial change in the composition of the fauna in hilly and mountainous areas, especially over the last few decades. The animal species that used to be characteristic of these areas have slowly but surely been replaced by those more typical of a forest habitat. In Italy more than two thirds of all forests have been grown for timber production and this has caused great changes in the species of broadleaf chosen for planting. The more vigorous species have become more common while the less flexible ones have tended to disappear. Shorter intervals in the timber-production cycle have prevented late-fruiting trees such as cerris (*Quercus cerris*), beech (*Fagus sylvatica*) and pubescent oak (*Quercus pubescens*) from producing more fruit, which is an irreplaceable source of nutrition for many wild animals. A lack of competition with domestic cattle and the new management of coppices have led to a steep rise in the number of large wild herbivores and this is causing considerable damage to vegetation and forest plants.

It is clear that the preservation of biodiversity is closely linked with the state of the environment, and that whenever extinctions occur there is a lessening of biodiversity and a consequent impairment of the environment. Extinctions are to be sure in some cases a quite natural phenomenon. Long before the arrival of man there were mass extinctions of species unable to adapt to climatic changes for example. In other cases however the extinction of species was due mainly or exclusively to man-made factors, especially when they altered or destroyed certain habitats.

One of the most pressing problems today is to preserve the environment and particular habitats in it. There is no doubt that in conservancy areas biodiversity is more secure; however, conservancy cannot be achieved only by increasing the number of conservancy areas; what is required is to combine biological preservation with territorial planning that safeguards the ecology of the countryside.

In this matter financial constraints make it necessary to set priorities, one of which will have to be to identify particular areas where biodiversity is still high, with numerous surviving species, and to concentrate on these areas. In practice they will be the smaller isolated fragments of an original larger ecosystem, and which can continue to serve as a refuge for animals and plants. In these smaller areas it will be easier to identify and save species threatened with extinction, to promote reforestation and to find ways to enable these species to coexist with an active human presence (Memoli 2003).

Animal species as indicators of biodiversity

Man-made environmental change tends to have negative repercussions on the ecosystem, making it necessary to take measures to protect and restore it. This is done by monitoring the ecosystem and reconstructing how it evolved over time.

In this context an evaluation of biodiversity becomes particularly important. The biodiversity of an ecosystem among other things gives a good idea of the functionality of that ecosystem (the whole network of its ecological relations) and of the effect that natural or anthropogenic changes have had on it. It is not always easy to evaluate biodiversity because environments vary very greatly among each other. To consider only the forest environment, for example, each type of forest is a distinctive system very different from the others. Each of these systems contains plant and animal species with their own peculiarities. When evaluating the biodiversity of an area therefore many variables must be considered, the most important of which are described by Burle and Gauld (1994) as follows: ‘long-term changes, short-term seasonal changes, the diversity of species in an ecosystem, variations in the number, abundance and rarity of individual species at different periods of their life, the mobility of animals causing them to move from one ecosystem to another, the extent of community development, the physiological situation of the ecosystem, and lastly the geographic scale, global, regional, national, ecosystemic, residential or individual plot.

Among the most reliable indicators to measure environmental quality in an ecosystem are birds and mammals since their presence depends upon particular environmental preconditions. Birds because of their mobility and the large territory they occupy during their life cycle are particularly useful as indicators of the relations between animal communities and vegetation in forest environments. They have this further advantage that there are many species of birds and they are easy to observe. Bird behaviour gives an indication of how a given bird specie thrives in different habitats (faunal peculiarity) and of how similar these habitats look to the birds of a species (ornithological similarity). Mammals can also be important bioindicators, especially of the ecology of a territory, since the idea that they only indicated levels of chemical pollution in the environment and nothing else is no longer accepted. Mammals adapt themselves ethologically and physiologically to changes in the ecosystem around them and this makes them reliable indicators of the quality of forest habitats.

The distribution of animal species can be conditioned by the way in which an original territory is broken up. The break-up of a territory can occur in three ways, each of which reduces the biodiversity of the original habitat: parts of the habitat can be lost, the remaining parts can gradually become smaller, and these parts can become increasingly isolated from each other (Bani et al. 1998). These authors state that if the remnants of a habitat fall below a threshold size it begins to reduce the size of populations in excess of that expected from the mere reduction in habitat area alone. This threshold size is between 10 and 30% of the useful *habitat* for both birds and mammals. The existence of thickets, rows of trees, hedges and uncultivated areas with spontaneous vegetation is important because, apart from offering refuge, they provide faunal corridors linking the remaining pockets of habitat that enable animals to undertake daily and seasonal movements and migrations or dispersals, and to extend their range (Malchevschi et al. 1996). In this way biodiversity is favoured and the ecosystems remain more functional because the genetic exchange between different subpopulations is maintained, and species from one area move to support those in another that are threatened.

Birds too, by their wide distribution and the abundance of their species can be used as indicators of environmental quality since the composition of bird species is closely linked to the habitat. There is a close connection between the overall biodiversity of an environment and the complexity of bird populations in terms of both the number of species (species richness) and the number of individuals.

Changes in the environment and in the land use in Italy have led to the disappearance of natural populations of the partridge (*Perdix perdix*) and to very great reductions in the Greek or rock partridge (*Alectoris graeca*) and the red-legged partridge (*Alectoris rufa*). Cultivated steppes, the *habitat* of the partridge, have almost completely disappeared. These were an ecosystem consisting of pasturelands and fields alternating with extensive cereal crops, interrupted by hedges and thickets, which spread all the way from the plains to the mountains. This was an ideal habitat for the partridge, with its non-migratory habits. The abandoned fields have now been taken over by a phytocoenosis that is deficient for this bird from a nutritional point of view, and the relentless spread of monocultures has among other things led to the disappearance of hedges and bushes that were an irreplaceable necessity for partridges to reproduce.

The abandonment of pasturelands and the failure to manage the forest environment have led to the disappearance of a suitable habitat for the woodcock (*Scolopax rusticola*) and for many other migratory birds that made their habitat in wooded areas. A study in the Apennines of Tuscany and Romagna carried out in the three-year period 1998–2000 (Casanova and Memoli 2001) found that the woodcock has a marked preference for pasturelands. In areas that still had their pasturelands the average occurrence score of these birds was more than twice what it was in areas without pasture. And indeed, in pastureland there are notable quantities of earthworms which are a fundamental part of the woodcock diet. Granval (1988) calculated that in intensively grazed and manured pasturelands there was a production of 2200 kg of earthworms per ha whereas areas under cereal cultivation yielded a mere 38 kg per ha.

The importance of birds in forest ecosystems is well known. They transport and assist in the germination of seeds, they have a role in the energy flows, and they help control plant-eating insects. To mention one example, 12 cuckoos (*Cuculus canorus*) and 10 golden orioles (*Oriolus oriolus*) were able in the space of a month to capture in a single row of mulberry trees more than 50 000 larvae of hyphantria (*Hyphantria cunea*), a polyphagous leaf-eating moth originating in North America that has very few other natural indigenous predators limiting its spread. But as Dickson (1979) reports, besides conferring these benefits, birds are also important ecological indicators.

James and Wamer (1982) and Erdelen (1984) studied the relationship between forest structure and the species composition and density of bird communities. Forest structure has a decisive effect on bird communities because birds when choosing a habitat are guided by vegetation characteristics. The nuthatch (*Sitta europea*) chooses fairly extensive forests of high quality to dwell in. Wiens (1989) found that older forests have a greater structural complexity than younger forests (more strata, more abundant undergrowth, etc.) and harbour more complex bird populations. The decision to convert the most public forests to high forests has led to a general ageing of these forests, which has benefited the avifauna, especially some of the more selective and characteristic species, such as in Tuscany: the great spotted woodpecker (*Picoides major*), the nuthatch, the tree-creeper (*Certhia familiaris*), and the creeper (*Certhia brachydactyla*).

A study carried out from 1992 to 1997 on behalf of the Tuscan Regional Government on nesting birds in the Casentino forests (Tellini Florenzano, 1999) found that the growing infrequency of particular types of environment such as cultivated and uncultivated fields and pasturelands, especially when they were located in close proximity to woodlands, adversely affected the survival of many species of birds and mammals which earlier had in those same areas found a food supply and places suitable to reproduce. It is no accident that in the north-central Alps some bird species whose natural *habitat* is in pasturelands have disappeared: the titlark (*Anthus campestris*), the whitetail (*Oenanthe oenanthe*) and the water-pipit (*Anthus spinoletta*), while the ortolan (*Emberiza hortulana*), a species closely linked to the agricultural and pastoral landscape, has been much reduced in numbers. The partial or complete disappearance of bird species naturally dwelling in fields and pasturelands is not a strictly Italian phenomenon but occurs also in other European countries.

Another important consideration is the fact that the loss of nesting birds from these areas is not being compensated by new species moving in to replace them, so that what is being witnessed is a drastic reduction in the nesting fauna of these areas.

Allochthonous conifers also have a negative influence on the richness of bird populations. Forests of pine and Douglas fir are non-native and recently introduced to the country, and hence the bird species for which they provide a *habitat* are relatively few. The beech-woods growing on the Tuscan slopes of the forests of Casentino consist of young trees with almost no undergrowth, and they therefore support only a limited number of bird species at a low density. Similar findings come from a study on Alpine avifauna carried out in the forest of Tarvisio in the eighties (De Biasio 1999). The type of forest with the smallest diversity in bird species was the beech-wood on account of the scant amount of shelter and food it provided. However, even a modest number of firs interspersed among other trees was sufficient to cause the bird population to react to the wood not as a beech-wood, but as a conifer stand.

While the ‘Statistical study on the migratory flow of some populations of thrushes and finches in Tuscany’ was being carried out, Casanova and Memoli in 1998–1999 undertook a parallel study on black pine and beech in forests of the state forest Giogo-Casaglia to ascertain the species composition of nesting birds and its seasonal variations. The entire area studied belongs to the Mugello mountain group astride the ridge of the Apennines. With an altitude that ranges from 600 to 1100 m above sea level it has a climate suitable for the beech and chestnut woods with which it is covered. Although the purpose of the parallel study was to explore the relationship that migratory bird species establish with non-migratory birds, it also revealed how bird communities utilised the resource of space for feeding and for song. Among the species examined, particularly interesting were some in the genus *Parus*, the great tit (*Parus major*), the coal titmouse (*Parus ater*) and the marsh titmouse (*Parus palustris*), all of which are insectivorous,. The behaviour of all these species was similar throughout the year and they fed on the same trees, but each species exploited a different part of the tree so that no competition arose between them. In the period from autumn to spring, the great tit preyed for food essentially in the lower parts of pine trees or on the ground for seeds and beech nuts. The coal titmouse captured its insects on the branches of black pine and along the beech trunks at the inserts of the larger branches; its food consisted of beech nuts which it found on the higher and thinner branches. The marsh titmouse had feeding habits very like those of the coal titmouse but it preyed on the more external and thinner lower branches of these trees. Of particular interest is what occurred in the second year of the study, when food in the territory occupied by the tits was scarce owing to unfavourable weather. As a result the boundary lines demarcating the feeding areas of the first year became less clear-cut and tended to disappear, and the three species invaded all areas that held out the prospect of sufficient food, even though this greatly increased the amount of competition for food between them, and also reduced reproductive success.

Papi (1995) studying a beech-wood on the Serra Lunga (Aquila) found that some bird species tended to select their reproductive *habitat* on the basis of some forest-structure parameters such as height, diameter, crown density, vertical distribution of the crown or stratification. Since some birds supply this information about the structure and the degree of naturalness of their forest they are a valid indicator for monitoring the forest environment. In a beech-wood at Collelongo (Aquila), Masci et al. (1999) carried out a similar study and found significant correlations between certain bird species and given types of forest structure.

In the investigations at Serra Lunga and Collelongo the species were grouped in ecological groups or guilds. When birds are studied with a view to determining the structure of the forests the term guild is used to express the concept that the population dynamics of an individual species are liable to be affected by any modifications that occur in the rest of the system.

The results obtained confirmed the importance of bird communities in forest ecosystems, and their possible role as environmental indicators. This is a matter of importance also from a management point of view, that’s why the U.S. Forest Service has an obligation based on precise rules to develop forest plans that will favour the vitality of all species of autochthonous vertebrates in their areas, and to improve the habitats of such species as are indicator species in forest management (Masci et al. 1999).

Mammals occur in various types of environment and this has favoured their use as indicators of biodiversity. The first thing to be noted here is that while for birds it is relatively easy to move to a more suitable *habitat* because they can fly, the break-up of a territory has an adverse effect on the spatial distribution of mammals.

Italian forest ecosystems contain small mammals such as insectivores and rodents, and some carnivores and ungulates of a larger size. Insectivores are sensitive indicators of the completeness continuity of the alimentary chain in that they are both predators (of insects) and prey (of the carnivores). Rodents on the other hand feed almost exclusively on vegetable matter and that makes them useful in monitoring pollution caused by the release of chemical substances. Carnivores, which occupy the top of the alimentary chain are the best indicators of how functional an ecosystem is. In particular the members of the weasel family (*Mustelidae*) such as the pine marten (*Martes martes*), the polecat (*Mustela putorius*), and the badger (*Meles meles*) indicate the degree of naturalness of an environment, and especially the effect of anthropogenic changes and the effectiveness of ecological relations as a whole.

The fox (*Vulpes vulpes*) by contrast is a poor bioindicator. Its increasing occupation of strongly urbanised environments has had a profound effect on its feeding habits. The availability of alternative sources of food in domestic refuse bins near centres of human habitation has freed the fox from its natural predatorial link with its prey. It is now a nearly ubiquitous animal able to move into almost any territory and survive there. The same goes for the more wide-spread mustelids such as the weasel (*Mustela nivalis*) and the beech-marten (*Martes foina*).

The larger ungulates present a rather different picture, especially the boar. These animals, which seems to be expanding rapidly, impair both the biodiversity and the stability of agricultural and forested areas. The boar (*Sus scrofa scrofa*) is an ecologically valuable animal that adapts its diet and its feeding habits to the food available in the environment it finds itself in. Its impact on the forest is very much linked to the nutritional choices it makes, and the response of an ecosystem to boar grazing is more resilient if it has a more complex structure. In general boars cause a considerable decrease in the plant biomass, but the richness of the flora, i.e. the number of plant species, is hardly affected (Massei and Toso 1993).

The boar may also cause changes in the relative proportions of the plant species in the territory in which it feeds. The result is that particularly in fields and pasturelands the gramineae, which the boar prefers to feed on, are gradually replaced by less favoured plant species. In their search for tubers, roots and small animals, boars root up the soil and this if it is drastic enough can seriously degrade the herbaceous vegetation and impair the stability of the soil. Lastly, when important food sources (chestnuts, acorns and beech nuts) become scarce the boar enters in competition with other animals, the red deer (*Cervus elaphus*), the roedeer (*Capreolus capreolus*) and the mouflon (*Ovis musimon*), whose nutritional requirements are the same. In this competition the boar is always successful on account of its adaptability.

The roedeer is a good indicator of the functionality of forest systems. The numbers of this animal have increased in the hilly parts of the central and northern Apennines in part because of the disappearance of their natural predators, the lynx and the wolf, but its current distribution over much of Europe is mainly due to changes in its social organisation which has evolved in adapting to the different *habitats* it has occupied. The best *habitats* of the roedeer are broadleaf forests rich in undergrowth, timberwoods (with their clearings), cultivated and uncultivated fields, pasturelands, and mediterranean maquis that supports so many wild animals. The optimal forest for this animal is probably the Turkey oak stand in its various forms, with its rich and varied vegetation consisting of both bush and herbaceous plants, and with an abundance of undergrowth which is fundamental for the nutrition of this animal. Owing to the roedeer's large energy and nutritional requirements its feeding strategy is one of 'concentrated selection'. Its bodily fat reserves supply it with only 10% of the energy it needs to survive the winter, so that it must make up the deficit from its daily food intake (Andersen et al. 1998). In spring and summer it preferentially feeds on legumes, some graminaceae and almost all broadleafs, saplings and bushes that grow well in Turkey oak plantations and chestnut groves. These latter however form an important part of its diet only when there is not enough herbaceous pasture, as has occurred several times in recent years owing to persistent summer droughts that hampered herbaceous growth. In winter on the other hand the roedeer like the other ungulates feeds exclusively on bushes and on natural regeneration, with a particular preference for native species of higher value, sycamore, common ash, mountain ash, silver fir, chestnut, and the like. The damage caused to natural regeneration is greater in mountain forests than at lower altitudes (Eiberle and Nigg 1987); this is because in the mountain and subalpine areas natural regeneration occurs in cycles that extend over several years and in their initial stages proceed very slowly. The exposure of the new plants to browsing at critical points in the cycle, especially by too many animals, can impair their regeneration for a number of years and cause a gradual shift in the species composition of the area.

Conclusions

So far the indications are that birds and mammals are important bioindicators of the environment. Monitoring those species of animals that are most suitable as bioindicators can show up any negative effects that man has on biodiversity.

The overall aim of bioindicator studies is to develop a method based on interdisciplinary co-operation to set about organising and managing the environment. This will not be an easy task, not least because the time required for scientific research cannot always keep pace with the immediate needs of the territory, so that research-based recommendations for action often arrive too late. Perhaps too it is better not to try to deal with all the aspects of biodiversity at once, but to focus more narrowly on a few of the elements that seem more worth while, even if this means taking action at the local or regional level.

In this connection the habitat map of the project Interreg IIC 'Base de données et cartographie' is of particular interest. This map covers two French and seven Italian administrative regions, and one of its aims is to map biodiversity at the regional level. The map among other things also gives useful information about potential habitats for certain animal species and possible corridors for animals to pass through. At present there are various methods of measuring biodiversity and they mostly vary in relation to the purpose they are intended to serve. But all these methods have a single purpose behind them, which is to preserve nature and to improve the management of the territory. These overriding aims have become vital ever since at some time in the last century man began to transform ecosystems and to make this transformation too an part of human development. In this way he has given rise to a process of unnatural selection in which plant and animal reproduction and survival depend ever more on whether they are compatible with human activity.

In this sense the aim of preserving biodiversity as expressed in the Rio Convention is important as it will lead to attempts to protect the environment by increasing our understanding of the ecological characteristics of various animals and habitats. This understanding must be resolutely pursued, in the conviction above all that the problem of the environment resides not in the environment itself, but in the excessive demands placed on the environment. It may well be that Aristotle was referring to these things when he wrote in the *Politics*: 'The greatest crimes are committed by men not in the name of necessity, but in the name of what is superfluous. People do not become tyrants to keep out the cold'.

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Session 6: Methods for Data Collection and Evolution of Monitoring Schemes

Ecological Multicriteria Evaluation as Fuzzy Prior Probability Supporting Forest Type Mapping on a Regional Scale

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Abstract

Sound ecosystem management depends on accurate, complete, and concise information regarding the extent, condition and productivity of natural resources. Distinctively, in order to define the elementary patch units within natural and seminatural forest ecosystems according to a "per habitat" approach, suitable reference can be made to forest types. The basic concept is to provide forest managers with an operational tool for discrimination of types of forests under different ecological-management conditions. High resolution remotely sensed data are not able, alone, to provide enough information for such a classification on a regional level. The present study deals with an experimentation carried out in central Italy for forest types supervised mapping based on the integration of multitemporal Landsat imagery (7 ETM+ and 5 TM) and spatial ecological information. The purpose of the study was to assess the effect of the inclusion of the prior probability information derived by forest type suitability analysis into the classification process, with the aim of developing the best classification method to automatically map forest types by remotely sensed images. MultiCriteria Evaluation (MCE) methodology was used to guide a modified maximum likelihood algorithm based on fuzzy logic. Classification accuracy is evaluated against a visually photointerpreted forest type map of the same area.

Keywords: forest type mapping; remote sensing; image classification; prior probability; multi criteria evaluation.

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1. Introduction

Sustainable forest management, especially in complex environments (such as Mediterranean ones), requires accurate, complete, and concise information regarding the extent, condition and productivity of natural resources. Distinctively, thematic maps are of basic importance as a tool to support forestry: knowledge about forest spatial characteristics are fundamental for updated forest management planning, and they also represent an essential basic information source for many modelling tasks such as animal habitat suitability assessment, hydrological applications and carbon sink modelling. (Corona and Marchetti 2000).

Traditionally, forest management has used information visually extracted from aerial photointerpretation. Such an approach is well-understood, properly used, and easily integrated with field surveys. Instead, information extracted from digital remote sensing data is operationally rarely used in forest management (Franklin 2001). But forest management is changing, and the shifts from the traditional forest management approach, focused on timber values, to sustainable forest management, focused on a wide spectrum of ecological, social and economic values, are becoming more and more demanding as concerns information, both from a qualitative and quantitative point of view (Bricker and Ruggiero 1998; Noss 1999; Simberloff 1999).

In such a context, forest classification is one of the main potential application of Earth Observation (EO). To support forest policies at regional and National level forest maps have to be based on systems of nomenclature able, at the least, to distinguish main forest species composition. Remotely sensed data adequate and feasible for such kind of reference scale (1: 50 000 to 1: 250 000), such as Landsat data, are usually operatively photointerpreted by skilled operators. In GIS environment photointerpreters use false colour composite (usually based on red and infrared bands) of one monotemporal image or two or more multitemporal images with other ancillary information (such as elevation and aspect from DTM). Using such datasets skilled operators are able to classify forest on the basis of main species composition. In order to accomplish the labelling process the operators mentally link spectral (colour), textural (shape) and contextual (proximity to water body, elevation, aspect, etc.) properties of forest patches with their own recognition keys.

Following such well known methodology in a project funded by the Italian Ministry of Environment a forest map of Italy at the reference scale of 1: 250 000 (minimum patch size of 50 ha) was completed on the basis of multitemporal Landsat images acquired around 1998. The map is developed starting from the CORINE Land Cover database, the only available at national scale, increasing its original thematic level from the third to a fourth level. Completed the project the accuracy assessment is now running.

One of the main problem in the operative diffusion of EO techniques in forestry is that supervised or unsupervised classification methods still don’t have the same performances of expert photointerpreters. In other words, the system of nomenclature successfully (in terms of classification accuracy) used in automated classification are not enough thematically detailed for forest management purposes (Franklin and Woodcock 1997).

The present contribution is therefore a first step to develop a classification system able to simulate the skill and experience of photointerpreters, based on objective and replicable rules. The prior information for such application is derived by CORINE database which is the only one available at national level. The procedure could be anyway repeated using other kinds of vegetation maps or inventory data.

The basic aim of the present work is to test the inclusion of ecologically-based prior probabilities in Landsat imagery classification in order to match a predefined forest type list based on main forest species composition. Ecological prior probabilities simulate the recognition process of photointerpreters. If spectral and shape properties of forest patches do not help the operators in the labelling process, the operators work presuming that in a certain area a forest category is more probable than an other on the basis of ecological requirements of forest species (in terms of soils, elevation, climate, etc.). Results achieved by standard Maximum Likelihood and fuzzy Bayesan classification methods are contrasted with results achieved by the same methods driven by prior probabilities.

2. Materials and methods

The study area, 5900 km^2 wide, is located in central Italy, between the administrative regions of Umbria, Marche, Lazio and Abruzzo, prevalently within the Mediterranean biogeographical region. The area was selected because of its topographic and vegetation complexity. The mean elevation is 893 m varying from 96 to over 2400 m. The mean slope is 25%, and all aspects are equally represented. The mean annual temperature is around 10 °C. Mean annual precipitation is 1133 mm, ranging from 670 to over 2000 mm. In mountain and hilly areas, where forests are prevalently located, soil parent material is mainly calcareous.

According to the CORINE Land Cover (CLC) database (EC 1992) less then 2% of the study area is covered by urban infrastructures, less then 30% is cultivated, while the remaining 68% is forested.

2.1 Data acquisition

Two Landsat images were available for the test area: one from Landsat 5 Thematic Mapper acquired on the 17^{th} July 1998 and one from Landsat 7 Enhanced Thematic Mapper Plus acquired on the 31^{st} July 2000 (Figure 1).

A national digital elevation model (DEM) of 75-m resolution was acquired from Italian Ministry of Environment.

From the same source raster images of mean, minimum and maximum monthly mean temperatures and monthly mean precipitation derived by the spatialization of local weather stations were also available (Maselli in press).

An “Ecological-soil map” developed by Joint Research Centre of the European Commission for the Italian Ministry of Environment in scale 1:250.000 was available. In such a map, each patch has a complex multi-scale system of soil classification (ESB 2002).

A “Vegetation cover and land use map” (1: 250 000 scale) developed by the integration of different information sources (Landsat imagery, digital orthophotos, forest management maps, etc.) was available. The map implements a 4^{th} thematic level of the CLC database (Figure 2). The original CLC project classifies forest in three classes: broadleaves, conifers and mixed (conifers+broadleaves). Instead, the 4^{th} thematic level is based on 14 classes, here assumed as forest types: 7 for pure and 7 for mixed forest types (Table 1). For further details regarding the datasets and methodology followed for the implementation of the 4^{th} thematic level in the CORINE database refer to Chirici et al. (2002).

2.2 Data processing

Landsat images were orthocorrected using 50 ground control points and the 75 m resolution DEM in order to reduce positional Root Mean Square Error to 10 m.

All information layers were projected in the same geographical system (UTM 33N, European Datum 1950) and resampled to match the same resolution of Landsat imagery (30 m).

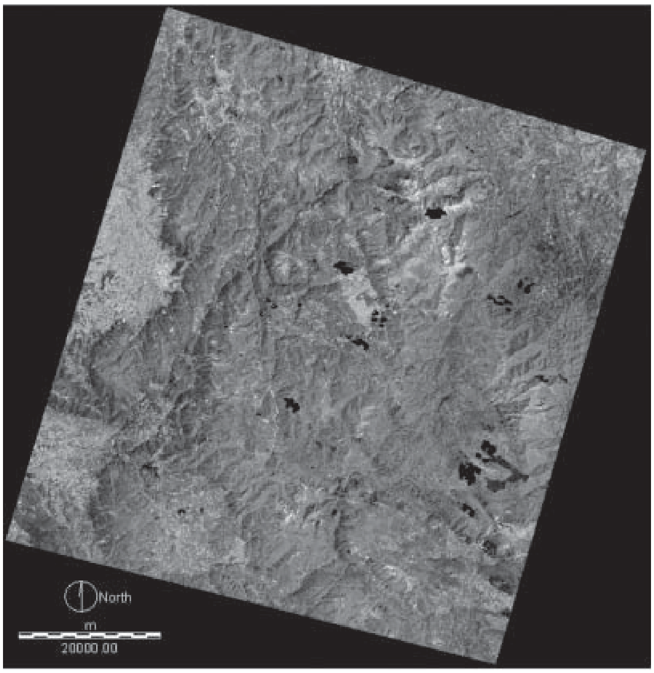

Figure 1. False colour composite (RGB: 432) of Landsat 7 ETM+ imagery acquired in summer 2000. Clouds and relative shadows are masked out.

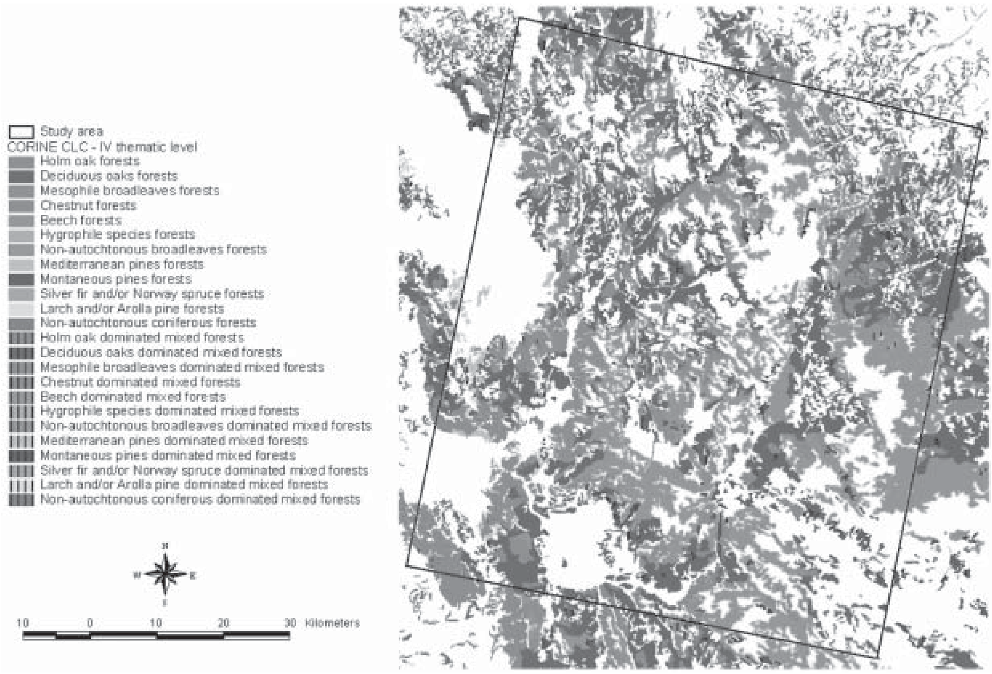

Figure 2. Fourth thematic level of CLC database for the study area (black square).

Table 1. Nomenclature system of 3rd and 4th forest thematic levels of CLC database.

CLC 3rd level	CLC 4th level
Broadleaves forests	3111 – Holm oak forests 3112 – Deciduous oaks forests 3113 – Mesophile broadleaves forests 3114 – Chestnut forests 3115 – Beech forests 3116 – Hygrophilous species forests 3117 – Non–autochtonous broadleaves forests
Conifer forests	3121 – Mediterranean pines forests 3122 – Mountainous pines forests 3123 – Silver fir and/or Norway spruce forests 3124 – Larch and/or Arolla pine forests 3125 – Non–autochtonous coniferous forests
Mixed forests (broadleaves+conifers)	31311 – Holm oak dominated mixed forests 31312 – Deciduous oaks dominated mixed forests 31313 – Mesophile broadleaves dominated mixed forests 31314 – Chestnut dominated mixed forests 31315 – Beech dominated mixed forests 31316 – Hygrophilous species dominated mixed forests 31317 – Non–autochtonous broadleaves dominated mixed forests 31321 – Mediterranean pines dominated mixed forests 31322 – Mountainous pines dominated mixed forests 31323 – Silver fir and/or Norway spruce dominated mixed forests 31324 – Larch and/or Arolla pine dominated mixed forests 31325 – Non–autochtonous coniferous dominated mixed forests

Cloud coverage, and relative shadowed areas, was removed with a mixed approach. An unsupervised classification (with isodata algorithm) was refined by manual digitalisation.

Slope and aspect maps were generated from the DEM, and a raster image of distance from sea was created from vector coast line (as simple linear distance).

DEM was also reclassed in 300-m-classes of altitude, while aspect maps were classified in three classes (north-facing, south-facing and intermediate conditions). These images were then intersected to produce an integrated altitude-aspect image.

2.3 Creation of prior probability maps

Prior probability maps were derived using a Multi Criteria Evaluation process (Eastman et al. 1993), the same usually used for fuzzy land suitability analysis. The first step for the analysis is the definition of the relationship between environmental factors and forest type suitability. Two different sources of information were considered: ecological knowledge from literature and direct evidences evaluated on the basis of environmental information available in the area as GIS layers. A fuzzy approach was chosen in order to express forest type suitability as a continuous value ranging from 0 (area not suitable for the considered forest type) to 1 (optimal area for the considered forest type).

Frequency distributions of all the possible combinations between examined environmental factors and mapped forest types from CLC database were analysed. By a reiterative approach

Table 2. Environmental factors considered in forest type suitability modelling.

Forest type	Environmental factors	
	Continuous	Discrete
Holm oak forests	July precipitation Distance from the sea	Elevation–Aspect Soil
Deciduous oaks forests	Mean minimum temperatures of January July precipitation December precipitation	Elevation–Aspect Soil
Mesophile broadleaves forests	Mean annual temperature	Elevation–Aspect Soil
Chestnut forests	Mean annual temperature Mean January temperature Mean maximum temperatures of July	Elevation–Aspect Soil
Beech forests	Mean minimum temperatures of January Mean maximum temperatures of July	Elevation–Aspect Soil
Mediterranean pines forests	Mean maximum temperatures of July July precipitation	Elevation–Aspect Soil
Mountaineous pines forests	Mean maximum temperatures of July Mean minimum temperatures of January	Elevation–Aspect

and on the basis of bibliographic references, just the most important factors for each forest type were selected (Table 2).

Fuzzy membership functions were derived by fitting the observed pixel frequency distribution of each considered forest type as a function of the values of each selected continuous environmental factors. The fitting was based on the minimisation of squared residuals (Figure 3).

For discrete datasets (such as soil map), a frequency-to-probability transformation was performed: higher the frequency of a specific class, higher the resulting suitability of such class.

Resulting maps show the forest type suitability of each pixel with respect to each environmental factor for each modelled forest type (Figure 4). Multi Criteria Evaluation was then used to aggregate all these resulting maps to create an overall suitability layer for each forest type. Such layers were used as prior probability maps for classification tests.

2.4 Image classification

The purpose of the present work was to assess the effect of the inclusion of the prior probability information derived by forest type suitability analysis into the classification process, with the aim of developing the best classification method to automatically map forest types by remotely sensed images.

In the light of this, a robust standard maximum likelihood algorithm was chosen and applied both with the hard and the fuzzy approaches. Training sites were acquired by a semi-automatic method. Spectrally homogeneous pixels within CLC forest polygons were digitised in order to acquire just those pixels spectrally representative of each CLC forest class. In fact,

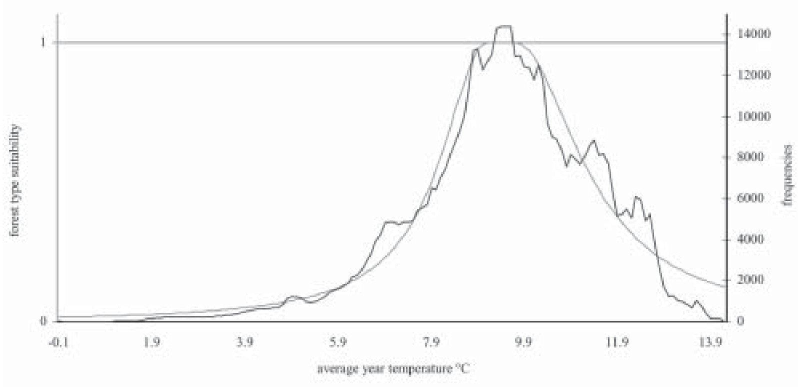

Figure 3. Fuzzy membership function for "Mesophile broadleaves" forest type suitability with respect to the average year temperature. Black line refers to the right y–axis representing pixel frequencies, while the left y–axis show fuzzy values referred to the best modelled membership function (grey line).

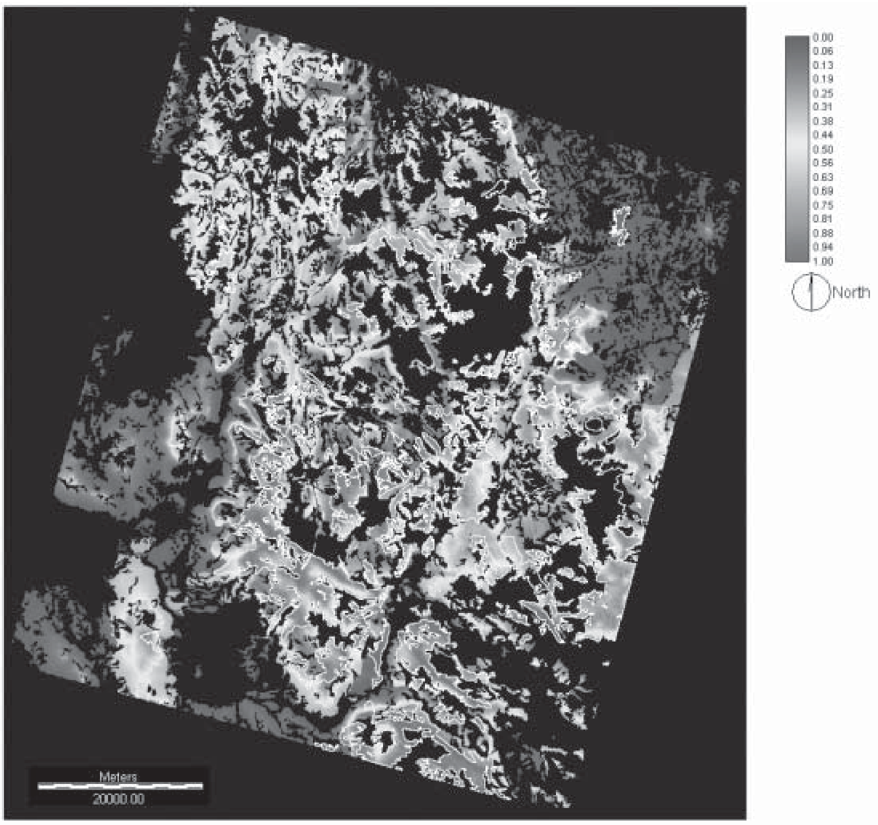

Figure 4. Example of fuzzy prior probability map for the beech forest type. White boundaries outline the polygons assigned to the class 3115 (Beech dominated forests) in the forth thematic level of CLC database.

Table 3. Classification results (“hard” = maximum likelihood classifier; “fuzzy” = Bayesian soft classifier; Kappa Index of Agreement is calculated on splitted test sites).

Tested bands	Classifier	Prior probability	KIA
Landsat 5 TM (1998)	Hard	No	0.62
Landsat 5 TM (1998)	Hard	Yes	0.83
Landsat 7 ETM+ (2000)	Hard	No	0.66
Landsat 7 ETM+ (2000)	Hard	Yes	0.85
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA	Hard	No	0.64
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA	Hard	Yes	0.82
Landsat 5 TM (1998) + DEM	Hard	No	0.77
Landsat 7 ETM+ (2000) + DEM	Hard	No	0.75
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA + DEM	Hard	No	0.78
Landsat 5 TM (1998)	Fuzzy	No	0.66
Landsat 5 TM (1998)	Fuzzy	Yes	0.85
Landsat 7 ETM+ (2000)	Fuzzy	No	0.70
Landsat 7 ETM+ (2000)	Fuzzy	Yes	0.88
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA	Fuzzy	No	0.68
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA	Fuzzy	Yes	0.84
Landsat 5 TM (1998) + DEM	Fuzzy	No	0.78
Landsat 7 ETM+ (2000) + DEM	Fuzzy	No	0.79
Landsat 5 TM (1998) PCA + Landsat 7 ETM+ (2000) PCA + DEM	Fuzzy	No	0.81

CLC database, along with its nomenclature system, is not conceived on spectrally homogeneous classes and it was acquired by visual interpretation: for this reasons, it cannot be used directly to derive training sites but it can be used to help their acquisition. CLC forest classes taken into consideration and assumed as forest types are shown in Table 2.

Digitised areas were randomly splitted in training (30% of selected pixels) and test sites (70%).

Training areas were used to create spectral signatures for each considered forest type. Hard maximum likelihood and fuzzy Bayesian classification (Richards 1986) methods were applied with or without the inclusion of prior probability, using monotemporal and multitemporal approaches. The possibility to include synthetic bands derived from DEM was also evaluated: to this end, the integer DEM map was linearly rescaled in 256 values (8bit) and considered together with Landsat bands. Concerning the multitemporal approach each Landsat set was compressed by Principal Components Analysis, and for each scene the first 3 principal components were used.

Classified maps were filtered with a 7 × 7 modal moving window and the classification accuracy was evaluated against test sites and against the original CLC database.

3. Results

On the basis of the accuracy analysis of classification carried out (Table 3) the separability of some broadleaves was confirmed as quite low. Especially the classes 3113 (mesophile broadleaves forests) and 3111 (deciduous oaks forests) are very similar. Indeed, such forest types cover more or less the same environments under an ecological point of view, at least at

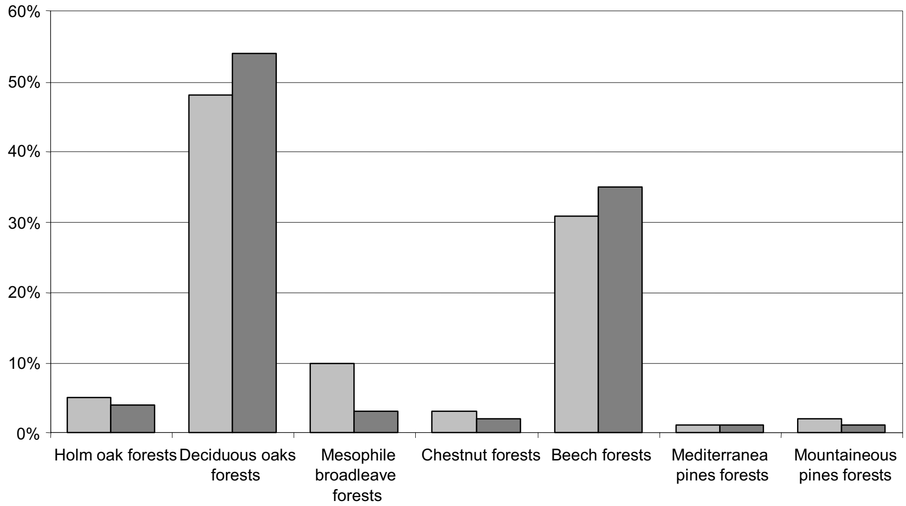

Figure 5. Classification results in labelling forest polygons of the CLC database: light grey is for the original dataset classified by visual photointerpretation; dark grey is for the labelling by the prevalent pixels classified with the best tested method (fuzzy classification with inclusion of prior probability of Landsat 7 ETM+ imagery). The scale is in percentage of the total forest area analysed.

the considered scale level. That's why the inclusion of prior probability has a limited impact on overall accuracy of the tested classifiers in the examined study area where mesophile broadleaves forests and deciduous oaks forests are widespread. In other situations, such as those where beech (3115) and chestnut (3114) types prevail, prior probability greatly enhance classifiers performances because of the ecological separability of those species. As concern conifers, whose spectral signatures are very different from broadleaves, the inclusion of prior probability proved to be able to discriminate between Mediterranean and Mountaineous pine types.

All the classifications based on Landsat 7 ETM+ dataset achieved better results then the same ones based on Landsat 5 TM imagery, and those ones based on fuzzy approach achieved better results than those ones based on the hard classifier (Table 3).

Even if with the canonical accuracy analysis (classified map again splitted test sites) of the best classification method (fuzzy approach with the inclusion of prior probability on a Landsat 7 ETM+ dataset) indicated very satisfying results, the cross-tabulation analysis with original CLC database is still weak (KIA=0.35). Contrasting results between original CLC and present classification datasets were quite clearly due to the fact that the first one is based on visual photointerpretation, where the operator labels digitised homogeneous patches, while the second is based on a "per pixel" approach. To better understand differences and analogies of these two datasets, original CLC polygons were labelled on the basis of prevalent forest types mapped by tested per-pixel supervised classifier. Close to 70% (in area) of CLC forest polygons were classified in the same way by photointerpreters (the original CLC database) and by the best classifier tested (Figure 5).

Since thematic and geometric accuracy evaluation is not yet available for original CLC database differences between Corine and Landsat classification based maps should also partly due of photointerpretation errors in Corine dataset.

4. Conclusions

To support sustainable forest management and close-to-nature silviculture, forest main species composition should be mapped, at least on a regional level. Target classes and nomenclature systems used in remotely sensed imagery classification should be defined by user requirements and not restricted by technical limiting issues. In other words, the best classification achieved could not be useful in forest applications even if it could generally satisfy remote sensing experts (Franklin 2001).

A discrimination between some forest species merely based on spectral responses is problematic using spectral and geometric resolutions such as those ones provided by Landsat sensors, at least under Mediterranean conditions. To derive useful classified maps from satellite imagery, elaboration algorithms should include methods that more close simulate complex human visual interpretation, for such purposes the application of segmentation techniques (Lobo 1997; Green 2000) and the inclusion of textural information (Yuan et al. 1991; Franklin et al. 1997; Gougeon 1995; Hay et al. 1996) are recommended and will be object of further analysis.

The purpose of the present work was to assess the effect of the inclusion of the prior probability information derived by forest type suitability analysis into the classification process, with the aim of developing the best classification method to automatically map forest types by remotely sensed images. The use of prior probability based on ecological preferences of main species composing those forest types to drive classification proves to highly increase the accuracy of produced maps.

For such a kind of application, the development of more complete databases describing relationships between spatial distribution of main forest species and environmental factors should be considered as a prioritary target, and the meaning and implications of such relationships should be the subject of further investigation using more in-depth statistical analysis procedures. Research is therefore in progress on the topic, using the same and different datasets.

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Baselines of Biodiversity – the Plant Species Richness of Natural Forests as a Target for Nature Conservation

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Abstract

Baselines of biodiversity are standards, reference values and targets for nature conservation. In forests the biodiversity of natural stands may serve as an appropriate baseline which helps to assess human impacts on ecosystems. In this paper the practicability of baselines is discussed for one component of biodiversity, the species richness of vascular plants.

Two assumptions have to be fulfilled if the species richness of natural forests is used as a baseline: Naturalness is given more weight than species richness and naturalness correlates with species richness. The relationship between naturalness (or management impact) and species richness is of key importance for the development of baselines based on the state of natural forests.

Negative, positive or unimodal relationships between naturalness and plant species richness are thinkable. An unimodal relationship is predicted by the intermediate disturbance hypothesis. A positive relationship between naturalness and species richness may be the result of microclimatic continuity in natural forests. A negative relationship is anticipated when increasing naturalness results in low light availability and therefore unfavourable growth conditions for ground-layer vegetation.

Baselines of plant species richness derived from studies in natural forests have to consider these different directions of the relationship between naturalness and species richness and thus depend on the forests under consideration. In many cases a species richness lower than the baseline (implying a negative relationship between naturalness and species richness) and in some cases higher than the baseline (implying a positive relationship) are appropriate targets for conservation.

Keywords: disturbance; forest management; naturalness; standard.

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1. Introduction

Effective biodiversity management needs three elements: (1) verifiable targets, (2) information on biodiversity status and trends, and (3) a plan for corrective measures (United Nations Environmental Program 1997). These elements are linked as, without a target, it is impossible to develop appropriate management practises for the maintenance of biodiversity. Targets, standards, benchmarks or baselines of biodiversity are urgently needed. Moreover policy makers want scientists to deliver quantitative data (Spellerberg and Sawyer 1996; United Nations Environmental Program 1997; Korn et al. 2001).

Leibundgut (1959) proposed using the natural dynamics of pristine forests as a reference for the sustainable management of forests. This approach has been supported in recent publications (e.g. Angelstam 1998; Schnitzler and Borlea 1998; Kuuluvainen 2002; Uotila et al. 2002). If it is possible to set up baselines for "natural" states, a comparison with these baselines will help to assess the impact of forest management practises. A baseline is defined in this study as the desirable state or value of a variable characterizing biodiversity. The term "biodiversity" is interpreted in different ways (Gaston 1996; Mayer et al. 2002). In this paper "species richness" is used as a surrogate of biodiversity (Bunnell 1997), where the variable of interest is the species richness of vascular plants on the spatial level of forest stands. We want to use the species richness of plants in natural forests as a standard of comparison for managed forests (see Roberts and Gilliam 1995; Scherzinger 1996; Parviainen et al. 2000).

A baseline for species richness based on investigations in natural forests only makes sense if two assumptions are fulfilled:

- There is a correlation between naturalness and species richness. Otherwise baselines of species richness cannot be connected to naturalness.
- Naturalness is given more weight than species richness. The target is to achieve a species richness comparable with that in natural habitats and not a maximum in species richness.

In this paper we (1) describe the underlying assumptions necessary for deducing baselines of species richness from studies in natural forests, (2) discuss the relationship between naturalness and plant species richness, and (3) assess the practicability of using baselines of plant species richness for guiding nature conservation in forests. We restrict the discussion on the spatial level of the forest stand, which is in the focus of forest managers, and do not consider coarser scale effects on the landscape level.

2. Species richness, naturalness, and nature conservation in forests

2.1 Variables and criteria for evaluation in nature conservation

Species richness and naturalness are measurable attributes of ecosystems. Species richness is the number of species per unit area, it increases with larger sampled areas (e.g. Rosenzweig 1995; Grace 1999). Naturalness is complementary to human impact, with a low human impact implying a high degree of naturalness (e.g. Dierschke 1984; Odum 1989). Others define naturalness as a pre-industrial state (Hayes et al. 1987) or, for North America, as the state before the arrival of European settlers (Swanson et al. 1994). Naturalness, which is usually given on an ordinal scale, is more difficult to quantify than species richness because many ecosystem properties are affected by human impact and have to be considered in a measure of naturalness (Angermeier and Karr 1994; Uotila et al 2002). Grabherr et al. (1998) assessed the naturalness of Austrian forests with criteria such as species composition in relation to potential natural forest association, disturbance indicators in understory vegetation, and the intensity of forest management impact. Trass et al. (1999) proposed for the assessment of forest naturalness a combination of different characteristics such as landscape unaffectedness, tree age, logs and their decay status, and the latest intensive cutting.

Species richness and naturalness are also criteria for evaluation in nature conservation. The variables can be assigned a specific value, e.g. "a high degree of naturalness has a high value for nature conservation". Deciding on these normative settings is outside the scope of the natural sciences and should ideally include intensive expert discussion (Plachter 1994). Often species richness as well as naturalness are positively correlated with conservational value: the higher the species richness and the higher the naturalness, the higher the value for nature conservation (Seibert 1980; Ammer and Utschick 1984; Scherzinger 1996).

2.2 Nature conservation targets in forests

The target of ecologically sound forest management has to be clear in order to evaluate possible alternatives (Davis et al. 2001; Failing and Gregory 2003). In the context of this study the following question has to be answered: is the target to achieve a maximum of species richness, a maximum of naturalness or an optimum of both?

There is no general answer to this question; the answer depends on the ecosystem under study and the answer implies value judgements and normative settings (Callicott et al. 1999; Failing and Gregory 2003). For temperate forest ecosystems in Central Europe we have to point out that:

- Forests usually are, compared to other ecosystems in the region (e.g. oligotrophic grasslands), relatively poor in vascular plant species per unit area (Seibert 1980). Therefore the main reasons for protecting forests in Central Europe will not be to ensure a high number of plant species in a relatively small area.
- Other organisms, such as wood-decaying fungi (Sippola et al. 2001) and epiphytic lichen (Goward 1994), depend on forests with a low human impact. In these the species richness of these organisms is high. Thus a high degree of naturalness has potential advantages for other components of the ecosystem besides vascular plants. These other components may be threatened if habitats with low human impact become scarce.
- Without human impact most of Central Europe would be covered by forests, with the exception of riverbanks, mires, rock outcrops, steep slopes, and areas above the tree line in mountains (Fischer 2002). Therefore forests have, compared to other ecosystems, a high degree of naturalness, which increases automatically when human impact is reduced (Dierschke 1984; Seymour and Hunter 1999).

As forests are ecosystems of relatively high naturalness, and as the area of ecosystems unaffected by human activities is declining, the reduction of human impact has priority in nature conservation in forests (Scherzinger 1996; Schmidt 1997; Reif 1999/2000; Plachter et al. 2000). The primary goal is not to maximize species richness but to protect natural habitats (Hunter 1999). Willson (1996) writes that "the goal is not to maximize diversity but rather to maintain natural levels of diversity despite human impacts".

The target of a high degree of naturalness is not valid for all forest types. In intensively managed landscapes forest types exist which depend in their structure and species composition on traditional, intensive forest management (e.g. coppice woods). In these forests not a maximum in naturalness but the maintenance of appropriate levels in species richness is the main target (Lux 2000). However in forests which are not subjected to traditional intensive management, e.g. in North America, Scandinavia, Siberia and higher altitudes of Central Europe, a minimization of human impact and thus a high degree of naturalness is an appropriate target.

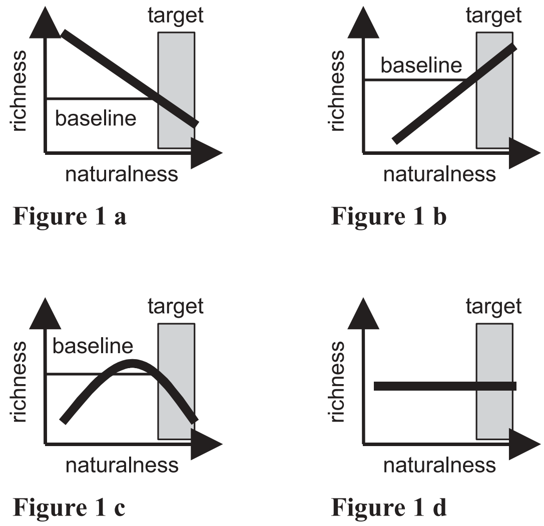

Figure 1. Scenarios for the relationship between naturalness and species richness. The species richness at high degrees of naturalness is considered as the target. The baseline is the species richness at the lower limit of this target.

3. Relationship between naturalness and plant species richness

3.1 Scenarios

In this section we discuss four scenarios for the relationship between naturalness and species richness, treating naturalness as the independent variable and species richness as the dependent variable (Figure 1). The state in natural forest stands is considered as the target. The baseline is the plant species richness at the lowest level of this target (Figure 1).

Scenario 1: the higher the naturalness, the lower the species richness (negative linear relationship).

A negative relationship between naturalness and vascular plant species richness was found in the studies (temperate and boreal forests) of Halpern and Spies (1995), Brunet et al. (1996), Schmidt and Weckesser (2001), Mayer (2002), and Oheimb (2003, see Table 1). In this scanario a species richness higher than the baseline should be avoided (Figure 1a).

Scenario 2: the higher the naturalness, the higher the species richness (positive linear relationship).

A higher plant species richness in natural forests was found by Økland et al. (2003, see Table 1). In this scenario a species richness lower than the baseline should be avoided (Figure 1b).

Scenario 3: unimodal relationship between naturalness and species richness.

An optimum of species richness at intermediate levels of naturalness is expected by Wulf (2001), but we found no example for forests in the literature. In the scenario of an unimodal relationship a richness higher than the baseline should be avoided. This statement is correct only at high degrees of naturalness where species richness decreases with increases in naturalness (Figure 1c).

Table 1. Comparative studies of vascular plant species richness in unmanaged and managed forests. The overview is restricted to temperate and boreal forests.

	Author, year	Dominant tree species, climate zone	Description of treatments
Plant species richness higher in managed than unmanaged forests	Halpern and Spies 1995	*Pseudotsuga menziesii*, temperate	Unmanaged: mature old-growth stands before clear-cut managed: same plots after clear-cut and slash-burning
	Brunet et al. 1996	*Fagus sylvatica*, *Qercusrobur*, temperate	Six classes from unmanaged for one decade through heavy thinning to clear-cut.
	Schmidt and Weckesser 2001	*Fagus sylvatica*, *Picea abies*, temperate	Differences in species composition in the tree layer from purely *Fagus sylvatica* (natural vegetation) to purely *Picea abies* (planted)
	Mayer 2002	*Picea abies*, *Fagus sylvatica*, temperate	Classes: 1) unmanaged for at least 25 years with closed canopy 2) unmanaged for at least 25 years with open canopy due to bark beetle attack 3) selection cutting 4) clear-cut
	Oheimb 2003	*Fagus sylvatica*, temperate	Stands without management for several decades compared with managed stands, ages of the tree stands at the sites were approx. the same
Plant species richness lower in managed than unmanaged forests	Økland et al. 2003	*Picea abies*, boreal	Classes: 1) old-growth without traces of logging 2) minor selectively cut 3) major selectively cut 4) formerly clear-cut Tree stands at all sites were 60 years old or older.
No differences in plant species richness	Graae and Heskjaer 1997	*Fagus sylvatica*, temperate	Six classes from untouched for more than 100 years to clear-cuts and planting. All stands were mature.
	Bobiec 1998	*Carpinus betulus*, *Tilia cordata*, temperate	Classes: 1) old-growth untouched for more than 70 years 2) old-growth with selective cutting 3) pioneer forest after clear-cut 70 years ago 4) 30-year-old plantation
	Abs et al. 1999	*Carpinus betulus*, *Tilia cordata*, temperate	Old-growth stand untouched for more than 70 years compared with adjacent managed stand

Scenario 4: there is no detectable pattern in the relationship between naturalness and species richness.

Species richness was not dependent on naturalness in the studies of Graae and Heskjaer (1997), Bobiec (1998), and Abs et al. (1999, see Table 1). In this scenario the two criteria have to be treated separately. It would then not be possible to use the species richness of natural forests as a baseline (Figure 1d).

3.2 Implications of disturbance theory

Naturalness is a “management term” which is used in ecosystem management and nature conservation but not in ecological theory. Thus there are no explicit predictions in ecological theory for the relationship between naturalness and species richness. But naturalness is linked with the theoretical concept of disturbances: high level of naturalness reflects a low intensity or frequency of human disturbances. Disturbance theory is central to forest management because “management activities are simply different forms and intensities of disturbance” (Roberts and Gilliam 1995).

Disturbances are considered as one of the main factors controlling species richness (e.g. Connell 1978; Sousa 1984; White and Pickett 1985). One general theoretical model for the relationship between disturbance and species richness is the Intermediate Disturbance Hypothesis (IDH, Grime 1973; Connell 1978; Huston 1979). The IDH predicts a maximum of species richness at intermediate levels of frequency or intensity of disturbance as well as at intermediate time intervals since the last disturbance. The IDH postulates that diversity is highest during intermediate stages of succession because enough time elapses for a variety of species to invade, but not enough time for any species to dominate (Roberts and Gilliam 1995). Thus if forest management activities change the stage of succession to intermediate stages, an increase in plant species richness is predicted (we discuss this further later).

There are three major problems with adapting IDH to forest management. First, it is difficult to define an intermediate level of disturbance (Hobbs and Huenneke 1992; Mackey and Curie 2000). Second, it is not clear which spatial scale the IDH refers to (Roberts and Gilliam 1995). An application at all levels of spatial scale at the same time is impossible because what is intermediate at one level may be minor at another. An intermediate disturbance intensity over a whole landscape could reduce species richness because species dependent on extreme conditions might be eliminated (Roberts and Gilliam 1995). A finding related to scale effects is that of Mackey and Curie (2000), who showed that the relationship between species richness and disturbance intensity depends strongly on the sampled area and the sampling intensity. Third, the IDH does not take into consideration differences in site conditions which may have, besides disturbances, a major influence on species richness (e.g. Ricklefs 1977; Grace 1999). This is a problem because forest managers have to deal with very different site conditions within their management units.

Theoretical assumptions about the relationship between natural disturbance regimes and diversity have led to the emulation of natural disturbance regimes in some approaches to forest management (Hunter 1993; Angelstam 1998; DeLong 2002). This is especially common in boreal countries.

3.3 Effects of human disturbances on plant species richness in forests

The effects of human disturbances on plant species richness in forests can be categorized as either initial and short-term or longer term (Halpern and Spies 1995). We discuss in this section influences on plant species richness, according to the two categories.

Initial effects

The destruction of plants through vehicle traffic, timber hauling and slash burning are initial effects that immediately result in a decline in plant species richness (Halpern and Spies 1995). Forest vehicles do not only disturb the plants but also the soil. Disturbed soil in forests is a regeneration site for many plant species (Harper 1977) and fine-scale increases in plant species richness after soil disturbance have been observed by Rydgren et al. (1998), Kobayashi and Kamitani (2000), and Mayer (2002). Forest vehicles transport propagules, e.g. in soil material attached to the tyres, and thus potentially contribute to an increase in plant species richness (Bonn and Poschlod 1998).

Light conditions, temperature and water may be changed by forest management measures. Timber harvesting is likely to produce gaps because trees are removed. Canopy gaps with high light conditions were found to be important for species richness in general (Packham et al. 1992) and for plant species richness in particular (Schmidt et al. 1996; Stone and Wolfe 1996). Maximum temperatures in gaps and on clear-cuts are higher than in closed forests, which may harm sensitive plant species (Meier et al. 1995). The increased water availability in gaps because of reduced tree transpiration allows the establishment of moisture-demanding plants and thus may result in an increase in plant species richness.

Longer term effects

Planting, thinning and harvesting selectively in managed forests affect the tree species composition. The ground-layer species composition may be affected by measures to control competition, e.g. the application of herbicides. How these measures affect plant species richness depends on the specific conditions.

Nitrogen fertilization influences plant species composition and diversity (Aerts and Bobbink 1999). After nitrogen fertilization Thomas et al. (1999) recorded an increase in overstory canopy cover and a decrease in understory vegetation cover and diversity. In contrast Qian et al. (1997) observed that plant species diversity increased with increasing soil nitrogen. These results are not contradictory because the typical response of plant species diversity and richness to nutrient availability is a “humped-back” curve (Grime 1979). Species richness is increased by fertilization at relative low nutrient levels and decreased at high levels.

The length of harvest rotation and the resulting maximum stand age have an effect on plant species richness. An increase in plant species richness with increasing stand age has been recorded by Halpern and Spies (1995), Qian et al. (1997), and Ewald (2002). In managed forests stand age is reduced because younger stands are more productive than older ones (Kramer 1988).

Stand heterogeneity may be increased or reduced by forest management measures. Stand heterogeneity is increased with small-scale selection cutting and the creation of small gaps. Heterogeneity is reduced when stands are transformed into single-species, single-age stands through thinning, planting and clear-cutting (Kuuluvainen 2002). Heterogeneity is positively linked with plant species richness (Giller 1984; Shmida and Wilson 1985; Pickett et al. 1997) and management practises which increase heterogeneity are expected to result in an increase in plant species richness.

Other long-term effects of human disturbances in forests, albeit on coarser spatial levels than the forest stand, are changes in spatial extension and habitat fragmentation, i.e. the alteration of previously continuous habitat into spatially separated and smaller patches (Dale et al. 2000). Hobbs (1988) and Benítez-Malvido and Martínez-Ramos (2003) found that plant species richness correlates positively with forest size. Given these results and according to the theory of island biogeography (MacArthur and Wilson 1967), a reduction of forest area is likely to result in reduced plant species richness. Forest roads divide forests into smaller sections and therefore may contribute to a reduction in plant species richness. But a positive effect of forest roads on plant species richness was found by Skov (1997), who claims this is due to the establishment of light-demanding flora along the forest roads and the way roads act as corridors for dispersal. Edges may be more abundant and sharper in managed forests in comparison with natural ones. In a review of edge effects in forests, Murcia (1995) reports both higher and lower plant species richness along the forest edges.

In conclusion, some effects of human disturbances in forests are likely to result in a decline in plant species richness. But there are at least an equal number of effects which promote plant species richness.

3.4 Comparative studies of plant species richness in managed and unmanaged forests

Studies of direct correlations between plant species richness in forests and naturalness are rare. In contrast, there are many studies investigating the effects of other factors on plant species richness. These factors are not directly related to forest management impacts but they may be changed by management activities (see section 3.3). Comparative studies of managed and unmanaged forests are somewhere in-between in terms of number as well as of suitability for assessing management effects on species richness. An overview of differences in plant species richness for temperate and boreal forests is given in Table 1.

The problem with comparing managed and unmanaged forests with respect to plant species richness is that management activities change different factors and affect plant species richness indirectly. One question is whether we describe e.g. changes in canopy cover or spatial extent as management impacts or as the impacts themselves. The effects on plant species richness of different management measures are subtle if all environmental variables are held constant. On the other hand, effects can be described with some confidence if changes in the environmental variables caused by management activities are included in the investigation. In the investigation of forest management effects of Schmidt and Weckesser (2001) tree species composition differs, in the studies of Halpern and Spies (1995), Brunet et al. (1996) and Mayer (2002) light conditions and successional stages differ. All these studies found a significant higher plant species richness in managed than in unmanaged forests.

4. The plant species richness of natural forests as a target for conservation?

Many empirical results indicate that management activities can lead to increases in vascular plant species richness. Theoretically, at least, an increase in plant species richness can be expected if a certain level of management intensity or frequency is not exceeded. Thus, if a high degree of naturalness is the aim in forests, too many plant species rather than too few seem to be inappropriate. But before we can provide reliable reference points for management targets, several problems have to be solved.

The biggest problem when setting up baselines for species richness is the large variation between sites, regardless of the extent of human impact. Species richness is a complex phenomenon (e.g. Sheil 1996; Grace 1999) and many hypotheses to explain its variation have been proposed (e.g. Ricklefs 1977; Grime 1979; Huston 1994).

Related to this point is that differences in species richness between successional stages, which are not caused by human impact, have to be considered. Untouched forests need not be dark stands with very old trees, a picture that many people have in their heads (Scherzinger

1996). Roberts and Gilliam (1995) argue that it is unrealistic to use only the species richness of late successional phases as a standard of comparison for managed forests. The early phases of the regeneration cycle in treefall gaps and after windthrow or insect outbreaks are essential components of pristine forests and differ in species composition and species richness from later stages. Changes in forest structure have been classified by several authors into four main stages (Leibundgut 1978; Bormann and Likens 1979; Oliver 1981; Peet and Christensen 1988; Korpel´ 1995). In the first "establishment stage" competition is minimal and plant species richness is high (Halpern 1988; Peet and Christensen 1988; Schoonmaker and McKee 1988). After canopy closure in the "thinning stage", plant species richness declines dramatically. It then remains at a low level in the "optimal stage". In the "decay stage" species richness is high again, after fallen trees have created gaps in the canopy (Peet and Christensen 1988). The point is that there is not one "typical" level of species richness in unmanaged forests, but rather several, which differ according to stage (Fischer et al. 2002). Aplet and Keeton (1999) tackle the problem with the *"historical range of variability"* concept (see also Landres et al. 1999; Davis et al. 2001), but restrict its application to the proportion of habitats in a landscape and do not include species richness.

If the species richness of natural habitats is taken as a baseline, another fundamental question arises: What is natural? Hunter (1996) is not sure whether the activities of Native Americans should be included in a definition of "natural" or not. And Haila (1997) points out that processes induced by humans are not fundamentally different from natural processes and that the term "natural" does not offers a clear benchmark (compare with Angermeier and Karr 1994).

Irrespective of these problems remain baselines a powerful and convincing tool if one prerequisite is fulfilled: the relationship between naturalness and species richness is, for the forest under consideration, clarified beforehand. Global biodiversity is declining rapidly (Wilson 1997) and tools for assessing its status and trends are urgently needed. Quantitative baselines derived from natural forests are, under the assumption that a detectable pattern between naturalness and species richness exists, promising candidates for these tools.

5. Conclusions

One target of contemporary forestry is to make forests as natural as possible. Accounting to this target, baselines of species richness developed from natural forests are appropriate tools for the evaluation of forest management measures. However, many factors affect plant species richness in forests and there is not one general relationship between naturalness and species richness. Depending on specific conditions vascular plant species richness may decline or increase with increasing naturalness. Therefore the target for nature conservation should be in some forests a species richness lower than the baseline and in others a species richness higher than the baseline.

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A Monitoring Network of Cork Oak Decline in Sardinia, Italy to Establish Control Strategies

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Abstract

A monitoring network of the declining cork oak stands in Sardinia (Italy) has been created. It consists of 43 stations spread out over the whole region in the various forest situations where the cork oak grows. The climatic, sylvicultural and phytosanitary profile for each station has been outlined and the decline index established. This parameter will be useful to follow the development of decline and to compare the declining areas with different characteristics, so as: i) to evaluate the incidence of the many potential adverse factors involved in the plant's decay, and so ii) to define the most suitable measures for re-establishing conditions of sustainable development in the Sardinian cork oak stands.

Keywords: decline, cork oak; monitoring network; Sardinia.

Introduction

The health of cork oak forests in Sardinia has progressively worsened in recent decades. Quite large areas are frequently encountered in all the main cork oak regions where young and adult plants are in poor condition, decaying or even dead (Marras et al. 1995; Franceschini et al. 1999; Sechi et al. 2002) (Figure 1).

This situation is very worrying because of the high ecological, economic and social importance of the Sardinian cork forests. Therefore, adequate measures must be taken to protect the plant's productivity and ensure their survival. However, it is very difficult to outline prevention strategies valid for all the various forest situations where the cork oak grows. This because of the multiplicity and variability of the potential adverse factors involved in stands with environmental and sylvicultural conditions often very different (Sechi et al. 2002).

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Figure 1. Mortality in cork oak decline area.

Thus, possible strategies should be defined only after the climate, soil type, vegetation and management system of each declining site have been analysed. The comparison among the sites based on the decay incidence would allow to ascertain which abiotic or biotic factors influence the plant's health in each stand situation.

With this aim, a research programme is in progress since 1998 in order to create a monitoring network of the decaying cork oak stands throughout the whole region (Franceschini et al. 2000, 2002). In this paper the characteristics of the monitoring network established until now are reported.

Materials and methods

The declining cork oak stands have been selected by systematic surveys in all the main Sardinian cork forest areas. A historic profile of these stands was outlined, describing the climatic trend, the forest management and any damage caused by adverse climatic conditions, fires, human activities, and insects or pathogenic fungi attacks. In addition, a circular sampling area of 20 m in radius (1256 m^2) was established in each stand to detect the following aspects:

- environmental: depending on the slope and exposition, the altitude, the nature of the substrate, and the average data for temperature and rainfall, both historical (1961–1990) and in the 1997–2002 period;
- silvicultural: all plants with a diameter of more than 5 cm were labelled, and their trunk diameter at 1.3 m was recorded. The de-barking height and the cork thickness were also measured for the cork oak trees in production.

  The density of plants of each forestry species, their total density and the percentage of consociation were computed. Moreover, according to the presence and composition of

Table 1. Classes of cork oak decline.

Crown transparency classes	Symptom intensity classes on branches and trunk 0	1	2	3
0	0	0	1	2
1	1	1	2	2
2	2	2	3	3
3	3	3	3	3

bushy undergrowth and the type of forest management, each cork oak stand was placed in one of the following three types: grazed and not shrubbed, grazed and shrubbed, not grazed and shrubbed. Finally, the natural regeneration of the forestry species was analysed, using four 20 m × 40 cm transects based on the cardinal points. The basal diameter and the height of each plantlet were recorded (Ruiu et al. 1996);

- phytosanitary: the crown transparency rate and the presence of cankers, bark necrosis, exudates and epicormic sprouts on branches and trunk of all the labelled plants were examined. Crown transparency was expressed as a percentage of defoliation, using 4 classes: 0 = 0–10%, 1 = 11–25%, 2 = 26–60%, 3 > 60%. The other symptoms were evaluated as a whole, using 4 classes of intensity: 0 = absent, 1 = scarce, 2 = medium, 3 = high. By combining these two groups of classes, as reported in table 1, the following decline class of each plant was defined: 0 = healthy plant, 1 = slight decline, 2 = medium, 3 = serious. The dead plants were considered in another class of value 4.

Finally, on the basis of the number of plants in these classes, the Decline Index (DI) of the stand was calculated, using the formula: $DI = \Sigma(C \times F)/N$, where C = value (0÷4) of each class, F = frequency of the same class in the stand, N = number of all the plants examined.

Results and conclusions

Up to now 43 declining cork oak stands spread out all over the region have been monitored (Figure 2). They reflect the different forest situations were the cork oak grows and show the following characteristics:

Environmental. The stands are found at three different ranges of altitude (up to 300 m asl, from 301 to 600 m, above 601 m). Most of them (58%) fall within the intermediate level (Fig. 3a). The stands on slopes were divided into the following groups, taking in account also the prevailing winds: East (including stands with E, NE and N exposure), West (W and NW), South (S, SE and SW) (Fig. 3b). More than half the stands has a granitic substrate; most of the rest is basaltic and schistose, and only in few cases the substrate is trachitic or sedimentary (Fig. 3c). The average temperature in the stands ranged from 1.3 °C to 27.6 °C. Average annual rainfall varied from 440 to 741 mm. The measurements were collected in the period 1997–2002.

Silvicultural. The stands were subdivided as follows:

a) 5 groups according to the plant's density, from < 400 to > 1000 plants/ha (Figure 4). Most stands have between 601 and 800 plants/ha;
b) 3 groups as to the consociation rate of cork oak with other forest species: ≤ 50%, from 51 to 80%, and more than 80%. In most areas cork oak plants made up more than 80% of the forest species (Figure 5);

Figure 2. Monitoring network of cork oak declining stands (○) in Sardinia.

c) 3 groups for grazing and undergrowth. The most frequent were grazed and shrubbed, followed by not grazed and shrubbed (Figure 6).

Phytosanitary. The stands were divided in 4 groups according to DI values: 0–1, 1.1–2, 2.1–3, 3.1–4 (Figure 7). The great part of the stands was included in the first two groups, few stands in the third one, and none in the last group.

By means of the DI parameter, it will be possible: i) to follow from year to year the evolution of cork oak stand decay; ii) to compare the declining areas in order to ascertain the role played by the adverse factors; iii) to evaluate the efficacy of the measures employed to prevent further spread of the decline.

In conclusion, this monitoring network is the essential instrument for studying a problem as complex as cork oak decline. The network will be available for regional organizations involved in the protection and conservation of Sardinian forests. They will implement and manage it by yearly surveys of the climatic and phytosanitary parameters at each station, and also changes in the sylvicultural conditions. In this way it will be possible to ascertain the real

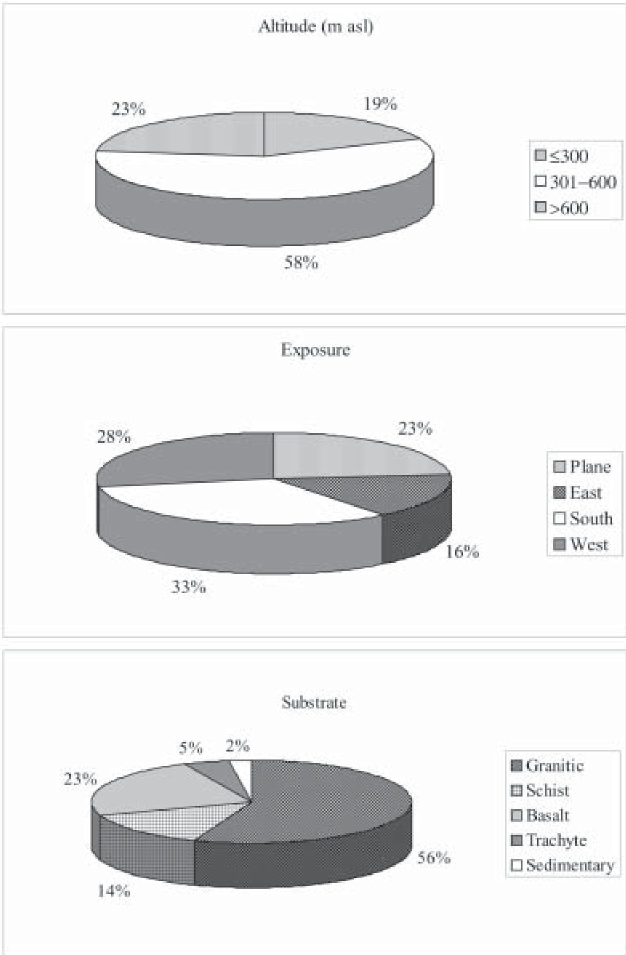

Figure 3. Percentage distribution of declining cork oak stands for: a) altitude, b) exposure, and c) substrate.

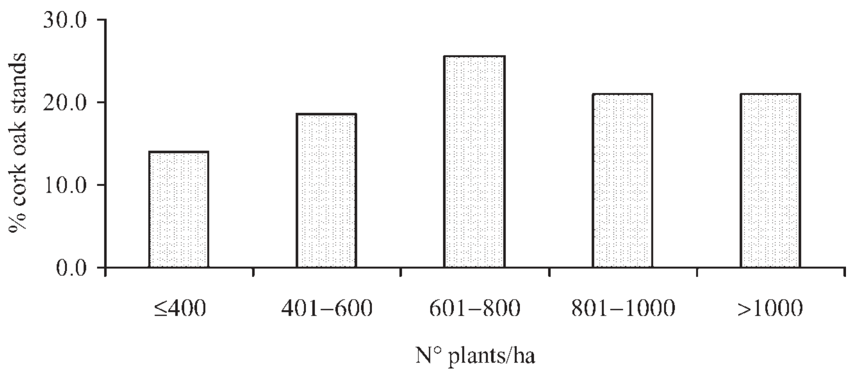

Figure 4. Distribution of cork oak stands according to the plant density.

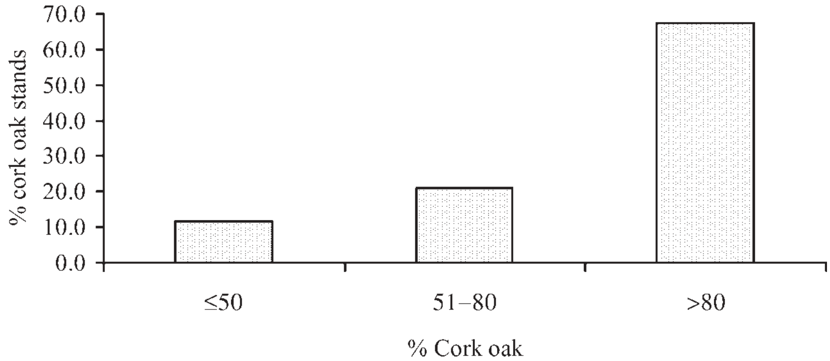

Figure 5. Distribution of cork oak stands for consociation rate of cork oak with other forest species.

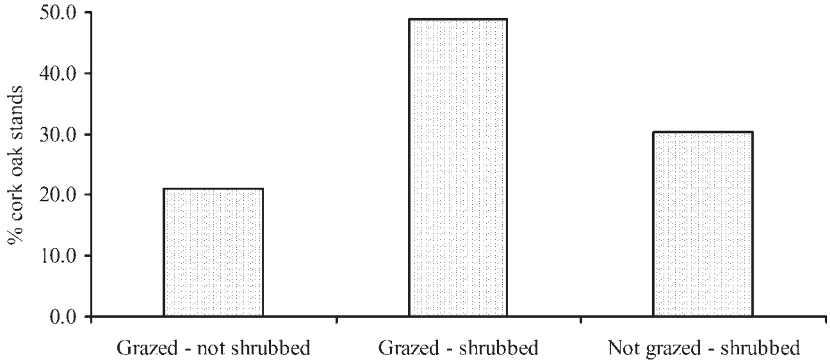

Figure 6. Distribution of cork oak stands for undergrowth and grazing typology.

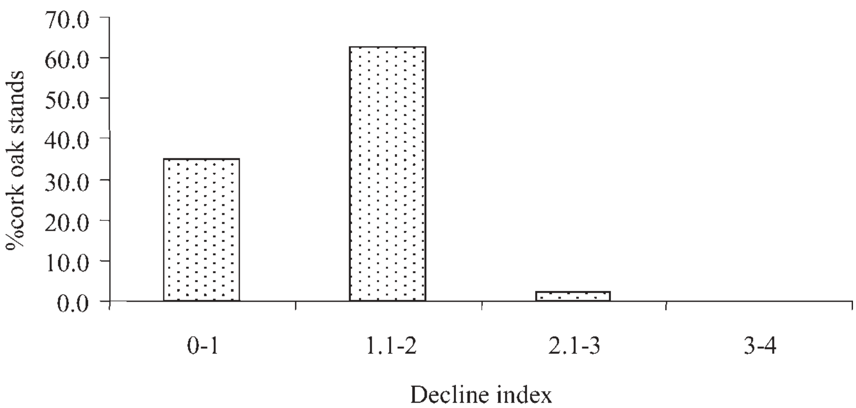

Figure 7. Distribution of cork oak stands according to the Decline Index.

extent of declining cork oak stands in the whole region, and to record the variations in the incidence of decline in the different stations. Moreover, by comparing all data relative to the environmental, phytosanitary and sylvicultural conditions of the stations it will be possible to identify the main causes of decline in each of them, and thus to outline the most suitable control strategies. The results will provide useful information for the economic and political decisions to be made to ensure the development of new sustainable management models for Sardinian cork oak stands.

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The Forest Monitoring Programme of ICP Forests – A Contribution to Biodiversity Monitoring

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Abstract

The UNECE International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) was established in 1985. In cooperation with the European Union it assesses inter alia deposition, stand and site characteristics and ground vegetation on around 700 plots. Abundance of the ground vegetation species could be explained by tree species, soil and site condition, climatic factors, and deposition. Other results support current views which accept acidification as a factor that negatively affects ground vegetation diversity.

In the context of high level political programmes and resolutions ICP Forests has launched the ForestBIOTA project for more specific contributions to forest biodiversity monitoring in the near future.

Keywords: ICP Forests; ground vegetation; deposition; biodiversity; forest monitoring.

1. Background and Introduction

The International Co-operative Programme on the Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests) was established in 1985 under the UNECE Convention on Long-range Transboundary Air Pollution. From 1986 onwards ICP Forests established in close co-operation with the European Commission a large forest biomonitoring network with the objective to provide a periodic overview on the spatial and temporal variation in forest condition and to contribute to a better understanding of the relationship between forest condition and stress factors in particular air pollution.

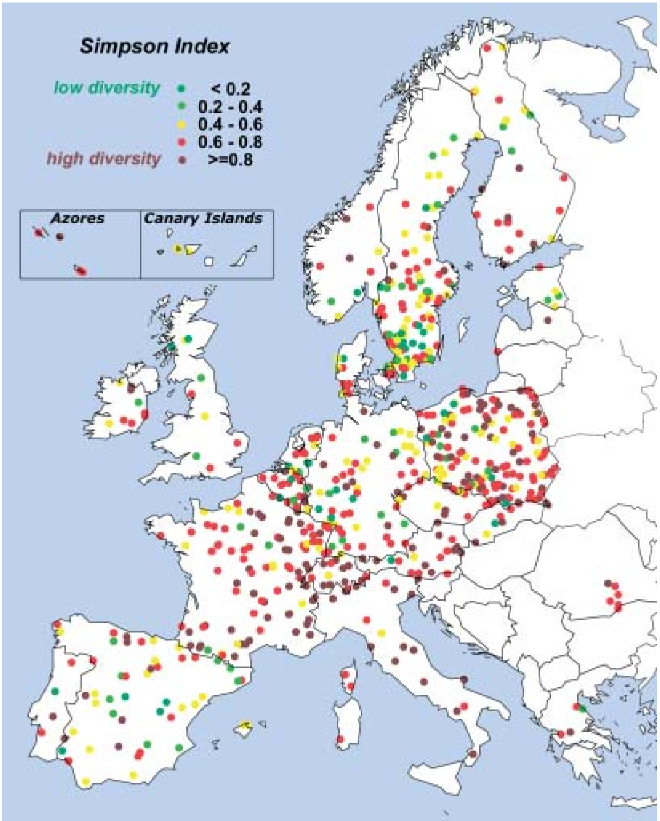

Figure 1. Plant diversity at 674 unfenced Intensive Monitoring plots, assessed for the first time in 1998–1999, mosses and lichens excluded. A repetition of the ground vegetation survey is routinely foreseen every five years. Results of the first repetition are expected in 2005.

To follow these main objectives, a systematic large scale monitoring network (Level I) and an Intensive Forest Monitoring Programme (Level II) have been set up. The strength of the Level I network is its representativity and the vast extent of its approximately 6000 permanent plots, arranged in a 16 × 16 km grid, throughout Europe. For intensive monitoring, more than 860 Level II plots have been selected in the most important forest ecosystems of the participating countries. A larger number of key factors are measured on these plots, including information on tree crown condition, foliar chemistry, soil and soil solution chemistry, atmospheric deposition, tree growth, ground vegetation, meteorology; the data collected enable case studies to be conducted for the most common combinations of tree species and sites in Europe.

The data also provide the basis of the programme's contribution to other relevant policy items such as biodiversity in forests. In specific, the programme's data on stand and site characteristics as well as on ground vegetation are of relevance in respect to biodiversity monitoring (see Figure 1). Of special interest is the possibility to explain changes in one of these parameters with stress factors also measured by the ICP Forests programme such as deposition, meteorological events or biotic agents.

Table 1. Percentage explained variance of the species abundances that could be ascribed to the four main groups of variables based on 194 plots.

Variable group	Explained variance
Actual soil situation	7.6%
Temperature, precipitation	5.6%
Tree species	4.1%
Deposition	3.3%
Total	20.6%

2. First results

Relationships between plant diversity and species numbers of the ground vegetation on one hand and environmental factors on the other hand were evaluated within multivariate correspondence analysis for approximately 200 plots for which combined datasets were available, including soil and tree species information, climatic data, and throughfall deposition (De Vries et al. 2002; Fischer et al. 2002).

Part of the variation in the abundance of the various species occurring in the ground vegetation could be explained by ‘tree species’, ‘soil parameters’ such as concentrations of potassium and calcium, pH, and N/C as well as P/C ratios, ‘climate’, mainly in terms of precipitation and temperature and ‘deposition’ (see Table 1). Soils with high pH, high base saturation and high availability of base cations, as well as southern climates and oak forests seem to determine high plant diversity. The impact of nitrogen deposition on plant diversity was small but significant. It is assumed that this influence will be even more pronounced when studying changes in plant diversity based on repeated surveys that will become available soon. The data sets of the Level II plots do presently not include direct information on light regimes and water availability within the stands. This is a clear item for improvement within the recently launched ForestBIOTA project (see Section 3.3).

Relationships between the occurrence probability of individual species and environmental factors were investigated for 332 different species. This was done by relating the species occurrence to more than 10 000 possible combinations of measured Level II data. Also these results show a predominant influence of soil chemistry, in particular pH, on the occurrence of single species and confirm the above presented findings. An example for 36 selected species against soil pH is given in Figure 2. Results show that most species occur on alkaline conditions whereas on acid sites only a few specially adapted species will predominate. This is in line with current views which accept acidification as a factor that negatively affects ground vegetation diversity. Species with a high modelled probability of occurrence on intermediate and acid soils are well in line with knowledge published in ecological textbooks. Astonishingly, some of the species with high modelled occurrence on alkaline soils are more intermediate species in the sense of e.g. Ellenberg indicators (e.g. *Melittis melissophyllum).* This might on one hand be due to the fact that Ellenberg indicators are originally confined to Central Europe. On the other hand Level II data do not contain information on all relevant ecological influences. In principal, however, the combined data provide a unique opportunity for the modelling of relations that were until now only based on empirical knowledge. In addition, dynamic models will in the future allow simulating the impact of changed environmental conditions on species composition.

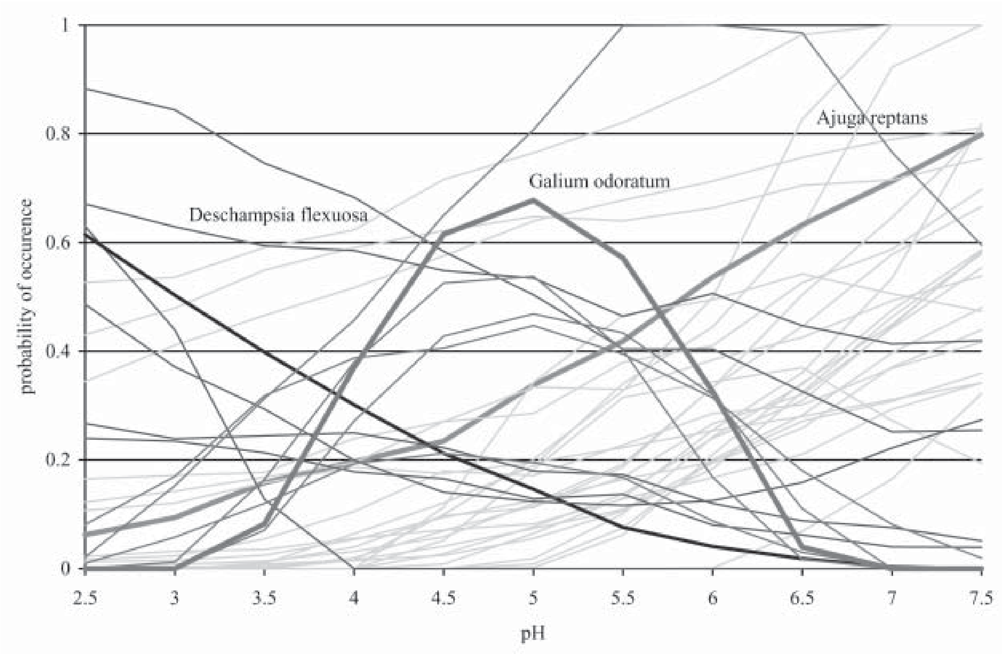

Figure 2. Response curves showing the probability of occurrence of 36 species at different pH. The species can be grouped into three species classes with an optimum at low, intermediate, and high pH; assessed in the period from 1998 to 1999 for 366 Intensive Monitoring plots. Species that prevail at intermediate soils are: *Galium odoratum, Melica uniflora, Anemone nemorosa, Veronica officinalis, Hedera helix, Carex sylvatica.* Species that prevail at acid soils (low pH) are: *Deschampsia flexuosa, Calluna vulgaris, Calamagrostis villosa, Vaccinium myrtillus, Vacciniu vitis-iaea, Picea abies Sorbus aucuparia*. Species that prevail at alkaline soils (high pH) are: *Ajuga reptans, Viola alba, Melittis melissophyllum, Dactylis glomerata, Sorbus domestica, Cardamine bulbifera, Silene italica, Digitalis lutea, Festuca heterophylla, Daphne laureola, Cruciata glabra, Ruscus aculeatus, Carex flacca, Stachys officinalis, Rubus caesius, Poa nemoralis, Carpinus betulus, Mercurialis perennis, Solidago virgaurea, Rosa arvensis, Luzula forsteri, Rubus idaeus, Prunus spinosa, Rubus ulmifolius, Arum maculatum.*

3. Future forest biodiversity monitoring contributions of the ICP Forests in a wider context – the need for concerted action

The need for preserving and monitoring biodiversity has been formulated on high political levels world wide. For the European region the Environment for Europe Ministerial conference in Kiew (ECE/CEP/94/Rev.1) as well as the European Union in its 6th environmental action programme (Decision No 1600/2002/EC of 22 July 2002) formulated the ambitious target "to halt the loss of biodiversity until 2010". At their 4th Ministerial Conference in Vienna, April 2003, the forestry ministers of Europe and the European Community declared the aim to "further maintain, conserve, restore and, as appropriate, enhance forest biological diversity" (Vienna Resolution 4).

Within a feasibility study, ICP Forests evaluated its possibilities for future contributions to forest biodiversity monitoring in the context of other monitoring systems like national forest inventories and remote sensing techniques (Fischer 2002).

Table 2. Ground vegetation information assessed in national forest inventories in selected European countries (for Austria: Schieler and Hauk 2001; for Germany: Bundesministerium der Justiz 2000; all others: European Commission 1997).

	Comprehensive species lists	Predefined groups or single species	incl. any inform. on mosses	incl. any inform. on lichens
Austria		50 types	2 types	
Belgium/F		yes		
Czech Republic				
Denmark				
Finland	few species	yes		2 types + few species
Germany		14 types	1 type	1 type
Ireland				
The Netherlands		N species	N species	
Norway		27 types		1 type
Poland		7 types	?	?
Spain				
Sweden	267 species and groups		22 species and groups	
Switzerland	at forest edge	closure of berries + GV	no	4×4 km (once)
U.K.				

3.1 National forest inventories

National forest inventories (NFI) are conducted in most European countries. Specifically in Northern Europe they have been carried out repeatedly. They mostly have a comparatively high spatial resolution and constitute a main source of forest information. The constraints are that – probably with the exception of Scandinavian countries (Stokland et al. 2003) - the methods are hardly harmonized across country borders and information is traditionally related to forest production. In the field of ground vegetation, for example, the assessed information mainly follows national systems for site type classification. Comprehensive species information is hardly available (see Table 2). Recent NFIs are measuring however an increasing number of biodiversity related parameters beyond vegetation assessments (e.g. Austria, Finland, Germany, Slovenia, Switzerland). ICP Forests cooperates with the recently launched ENFIN project (European National Forest Inventory Network) aiming towards a harmonisation of existing inventories.

3.2 Remote sensing data

Remote sensing data are of particular interest for landscape level assessments. Land cover maps and forest fragmentation information are typical fields of application. Technical possibilities are quickly developing and a number of information sources are existing today (e.g. CORINE and IMAGE 2000 data). For stand scale information which is relevant for the ICP Forests plots, however, the possibilities are limited. The FAO concludes that “a land cover map is not a forest assessment, and […] a return to field inventories to supply policy relevant national information seems to be appearing” (Holmgren and Persson 2003).

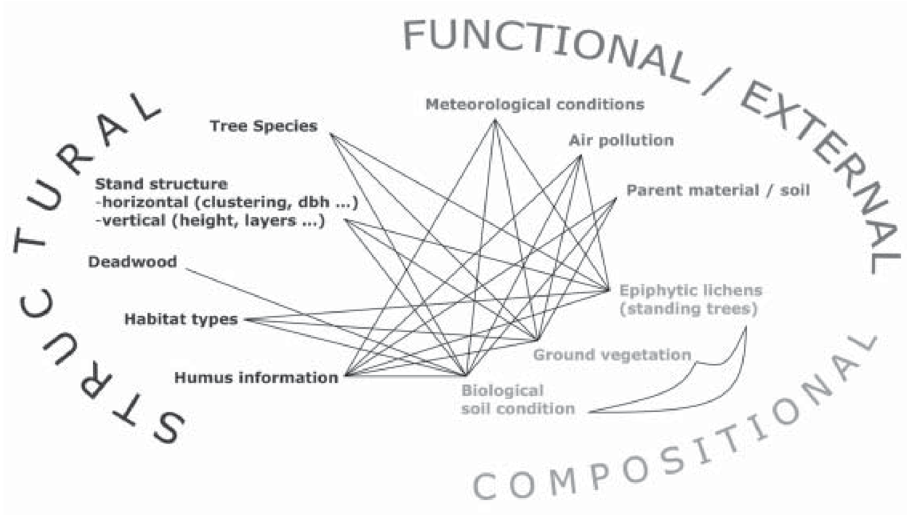

Figure 3. Main parameters and their possible relations on intensive monitoring sites of the ForestBIOTA project.

3.3 The forest monitoring programme of the EU and ICP Forests and ForestBIOTA

The strengths of the programme are its well established transnational monitoring and reporting infrastructure based on a common legal basis. Its present value for representative (on Level I based) biodiversity information is however limited (Packalen and Maltamo 2001) as in many cases only main tree species are assessed. Substantial efforts would be necessary to implement additional assessments on the large scale. For the development of methods and indicators as well as for the test wise implementation of harmonized monitoring methods the intensive monitoring plots offer however a unique platform. Under the ICP Forests Working Group on Biodiversity assessments the newly launched ForestBIOTA (Forest Biodiversity Test Phase Assessments) project will utilize this (ICP Forests 2003).

ForestBIOTA is a joint action of 14 European countries to be carried out on 123 existing plots. It aims at the further development of forest condition monitoring activities by conducting a monitoring test phase under Art 6(2) of the Forest Focus regulation. Its objectives are

1. the test wise development and implementation of additional assessments
2. correlative studies for compositional, structural and functional key factors of forest biodiversity based on existing Intensive Monitoring (Level II) plots (see Figure 3).
3. recommendations for forest biodiversity indicators that can be applied in the context of existing national forest inventories (collaboration with ENFIN – European Forest Inventory Network).

Harmonized methodologies for additional assessments in the fields of (a) stand structure, (b) deadwood, (c) forest classification, (d) epiphytic lichens, (e) extended ground vegetation surveys were already developed in 2003. Together with the existing Level II data base and infrastructure they are a main feature of the project (see Figure 3) which is scheduled until the end of 2005.

4. Outlook

The wealth of the ICP Forests data is an important source for studies aiming at detecting plant diversity changes and their possible causes. In addition the data can be used to further develop indicators for biodiversity. In order to gain highest benefits of its data, ICP Forests is interested in a close co-operation with partners in the field of biodiversity monitoring. ICP Forests is therefore member of the International Working Group for Biodiversity Indicators and Monitoring (IWG BioMin), convened at the Environmental Agency and will in this context also contribute to indicator development and monitoring under the 6th Environmental Action Programme of the EU. The close co-operation with the new EU Regulation Forest Focus is another important milestone for ICP Forests.

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A Methodological Approach for the Improvement of Biodiversity Monitoring and Management

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Abstract

This paper addresses the methodological approach assisting in providing credible estimates on the status of biodiversity and the ways of implementing biodiversity preservation and conservation actions in the field. In addition, it conveys experiences gained in developing Slovenia's biodiversity monitoring framework.

Designing a concept for biodiversity monitoring incorporates several working phases. The most important are the determination of biodiversity indicators, data definition and premises concerning the ways of data collecting. To gain the majority of opinion, all the phases should be developed in close collaboration among the initiators, the designers of a monitoring system and the stakeholders.

However, monitoring itself is not a guarantee of sustaining biodiversity. Therefore, natural resource planning, which addresses biodiversity issues, must be introduced as soon as possible.

Keywords: capacity of monitoring, data definition, the assessment process, planning for biodiversity, Slovenia.

1. Introduction

Although the notion of biological diversity has been known for a considerable length of time, it was not earlier than the 1990s (especially after the Conference in Rio) that biodiversity was internationally recognised as a factor significantly affecting sustainable development (United Nations 1992). Such recognition has had profound consequences for work in environmental sciences, land management and policy, which, since then, have been addressing the most intricate questions bound to biodiversity. Consequently, we now have a clearer picture of the ecological role of biodiversity (Noss 1990), indicators best describing its components (Cohen

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and Burigel 1997; European Environment Agency 2003; Ministerial Conference... 2002; Larsson 2001), possibilities for its inventorying (Hansen and Burk 2000; Bachmann et al. 1998) and of the means for implementing actions in the field (Szaro et al. 1999; Kovac 2002).

However, despite the newly evolved knowledge and clearly expressed incentives for biodiversity monitoring at all scales, the changes made in day-to-day monitoring practices are still rather small. Although it is difficult to say what reasons (besides financial) hinder its faster development and implementation, it is quite obvious that the paucity of recommendations and poor organisational knowledge, necessary in designing monitoring schemes, are serious obstacles. To evade the problems of that kind, which may – if not properly addressed – severely jeopardise the entire biodiversity monitoring program, this paper addresses only the selected topics related to design and implementation of monitoring systems. As such, issues considered include the organisation of a monitoring system, the assessment process and the means of implementing biodiversity actions in day-to-day management. In turn, the paper also brings to light experiences gained in developing Slovenia's biodiversity monitoring framework.

2. Factors to consider in biodiversity monitoring and management

2.1 Fundamentals

In systems theory the control system is considered a unity of processes and control structure. However, because the direct monitoring of natural processes is impossible, they are monitored in two ways: by analysing the relationships between the actions and outcomes and by analysing such measurable data and indicators that reflect the processes operating at different spatial scales. The same idea has also been applied to operational biodiversity-monitoring. Nevertheless, except for internal organisation, which always appears to be a critical factor of monitoring efficiency, little attention has been paid to the general principles necessary in designing monitoring systems or to the assessment process in the broader context.

2.2 Internal organisation and general principles

Four sources of information generally fulfil the demands of biodiversity planning and management: image data, field data, cartographic data and textual/numeric information. Since all the data is of different kinds and types, the biodiversity monitoring system is to be based on GIS technology – the best for integrating four dimensions of the monitoring systems; namely: location, scale, attribute, and time.

In addition to organisation, four more factors help improve monitoring efficiency (Lund 1986):

- Adaptability to a wide range of ecological conditions;
- Usability at different spatial levels such as eco-region, landscape, ecosystem and site;
- Compatibility, exchangeability and flexibility to fulfil the information needs of various users;
- User-friendliness.

As assumed, the greatest attention should be paid to the first two factors, both related to the shaping of suitable sampling designs.

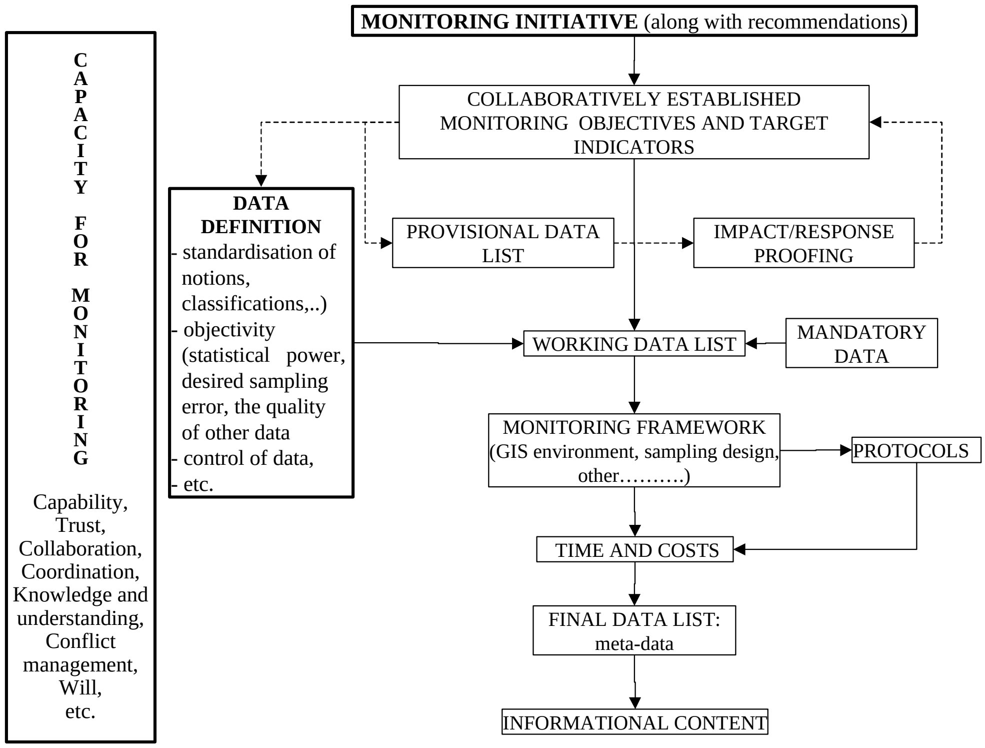

Figure 1. The scheme of the assessment process in the broader context (adapted from Hocevar 1996).

2.3 Assessment process

The main tool of assuring quality data is the assessment process (Hocevar 1996). Unlike some years ago, when it was considered just the algorithm for checking informational needs, the modern assessment process undertakes critical judgements of a much broader social environment. Therefore, as shown in Figure 1, the process should inherently include three rather independent activities: strengthening the capacity for biodiversity monitoring, data definition and the assessment process in the narrow sense.

Fostering the capacity for monitoring is the essential condition of biodiversity monitoring as a whole. Also quite significant is its role in the assessment process, where it assists:

- In bringing together scientists, stakeholders and data-users of different professions, backgrounds and views;
- In establishing a collaborative and trustworthy working environment;
- In increasing the legitimacy and fairness of the assessment procedure;
- In assuring valuable and trustworthy data, etc.

Next to monitoring capacity is data defining, which deals with the determination of all kinds of standards, statistical features as well as with the methods of the data control (Lund 1986). In the case of biodiversity monitoring, many of these issues are unfortunately still open. In the field of criteria and indicators, for instance, there is a growing interest for the recommended set of mandatory parameters to be assessed at all possible spatial scales, regardless of the type of land use. The lists proposed by the European Environment Agency (2003) and Ministerial Conference... (2002) seem to be a good starting point. While the first still needs further refinements in regard to definitions and protocols, the second is considerably well defined and can be recommended for testing.

Another difficulty is bound to the attributes indicating qualitative features. These of attributes are considered “suitability of habitat conditions”, “suitability of the management regime” and many others, that may be of interest in the future. Because such indicators are hardly provided on an objective basis, they should not be recommended unless they are unambiguously understood, defined and considered compound variables, which only can be derived through well-experienced models.

A similar issue concerns shortcuts to monitoring. These are considered for those sensitive species that quickly react to external impacts. Thus, if monitoring addresses them, it is not only possible to infer their viability, but also to say something about the severity of impacts as well as about the viability of other species (e.g. McLaren 1998). Such lists, quite reliable in the case of floral and animal species, would also help improve ongoing discussions about what more biodiversity monitoring should address.

Representativeness, sampling error and statistical power are statistical notions related to the selection of samples, accuracy of data and the likelihood of error. Because biodiversity monitoring is seldom organised independently and because its data will most likely be used at an international scale, knowing how accurate the desired information should be is recommended. The answer to that question is crucial, because many countries are entering biodiversity monitoring. While some of the publicly known attempts pursue the basic statistical principles (Bundesamt für Umwelt...1999; Schieck et al. 2003), some of them do not take account of them at all.

Since it assures error-free data (in accordance with the standards), the control is perhaps the essential part of monitoring. The basic elements the control should set are: the list of parameters to be controlled, timeframes, detailed instructions on the control procedure, and the assignment of responsibility (Lund 1986).

2.4 Considerations on the implementation of biodiversity preservation actions in the field

In bridging the gaps between the ecological potentials of the environment and the rapidly increasing environmental crisis, environmental sciences have developed many planning concepts. Despite the richness of trials, it is presently recognised that biodiversity issues can be best managed with natural resource strategic planning (Committee...1999; Kovac 2002; Groves et al. 2002). Such planning should be conducted within the eco-regional planning areas of different sizes that best reflect the conditions of selected land units and make ecological planning possible. In the European context of planning and management, at least four planning levels are recommendable:

- European large-landscape level for providing strategies needed in shaping the environmental policy and in stewarding the most important European vegetation types toward desired conditions. In this view, the proposal developed in the BEAR project (Larsson 2001) is a good starting point.
- National large-landscape level for providing strategies needed in shaping national environmental policy, sustainable development and large land complexes (e.g. forest types);
- National landscape-level for providing strategies necessary for the forest and land stewardship;
- Ecosystem-level for implementing actions in the field.

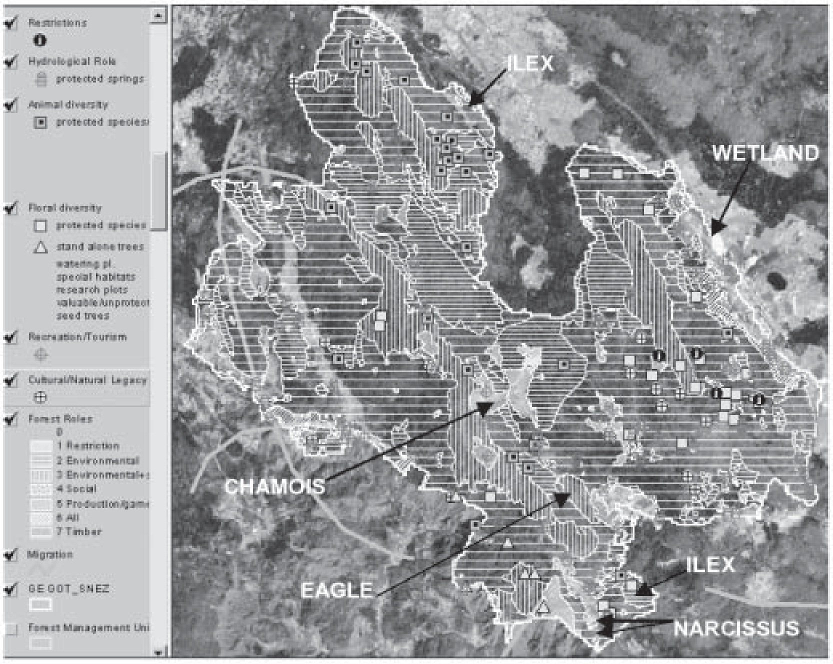

Figure 2. Established forest roles in one of the regions in Kocevska-SLO (Kovac 2002). The figure shows the distribution of environmental, social and productional forest role, all of them supporting sustainable forest development. The instructions for management with such sites are given in Regional Forest Plans and Forest Management Unit Plans.

As far as management approaches are concerned there are several possibilities. Worthwhile mentioning are the concepts of Natura 2000 and of forest roles (Bachmann 1996; Kovac 2002). However, while the first is still in development the latter has been used in Central European forest practice at least for a decade. The mission of both approaches is the long-term preservation of lands with certain characteristics. Thus, once the criteria for selection are defined, such lands can be identified, mapped, planned and managed (Figure 2).

3. Biodiversity monitoring and management - experiences from Slovenia

3.1 Forestlands

3.1.1 The concept of multi-resource inventorying

For historical reasons, Slovenia conducts a rather complicated forestland inventorying, consisting of four independent activities (Figure 3):

- Stand inventorying of forest compartments (Regulation on...1998);
- Large-scale statistical inventorying (Kovac et al. 2000);
- Operational inventorying of Forest Management Units by means of a control method (Hocevar 1990, Regulation on...1998).
- Forest roles (Regulation on...1998).

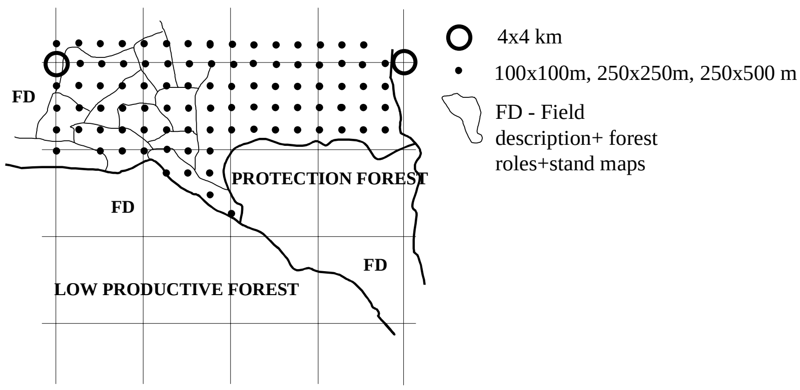

Figure 3. The concept of Slovenia's forestland inventorying (Kovac 2002).

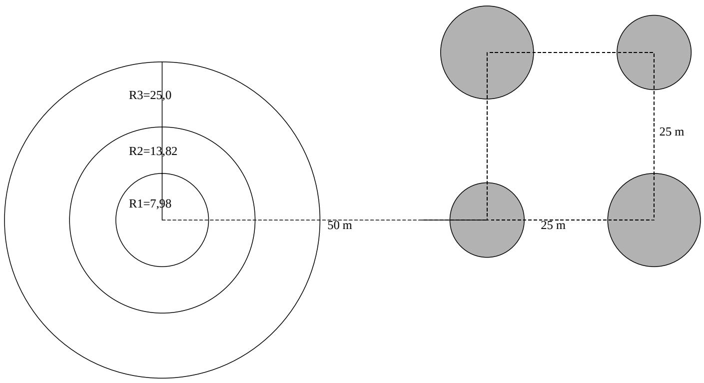

Figure 4. The scheme of the present large-scale inventory design (Kovac et al. 2000). Each sample tract consists of one (slope-rectified) permanent sample plot (left-sided concentric circles) and of four Prodan's M6 plots (right-sided shaded circles).

The longest tradition (since World War II) uses stand inventorying, which is based on the field inspection of all forestlands in the country. The inventorying is being conducted successively on 10% of forestlands annually.

Next to this is large-scale statistical inventorying, which began in 1985. The early inventory design was largely based on the International Co-operative Program methodological framework. However, due to the recognition that the system would not provide trustworthy information, the inventory design was modified in the easiest possible way (Figure 4), by introducing an additional invisibly marked permanent sample plot.

The third data set, the control method (originally developed by Schmid-Haas), was introduced for the first time in the late 1960s in only one of the fourteen forest enterprises. Because of its many advantages (Schmid-Haas 1983; Hocevar 1990) it became part of national inventorying in 1994. By the end of the year 2003, it was implemented on approximately 60% of forestlands (on the sampling grids 100 × 100m, 250 × 250 m and 250 × 500 m).

Table 1. Availability of information with regard to the Ministerial Conference... C4 and C5 criteria.

MCPFE C4 and C5 Criteria	Availability			
Indicator	Forest	OWL	Source	Data quality
41 Tree species composition	Y	N	LSI	SE
42 Regeneration	Y	-	FD, SM	Acceptable
43 Naturalness	Y	N	FD	Acceptable
44 Introduced tree species	Y	N	FD	Acceptable
45 Deadwood	Y	N	LSI	SE
46 Genetic resources	Y	-	FD	Acceptable
47 Landscape pattern	-	-	RS	SD
48 Threatened forest species	Y	N	FR	Acceptable
49 Protected forests	Y	Y	FR	Acceptable
51 Protective forests/S+W	Y	Y	FR	Acceptable
52 Protective forests/I+MNR	Y	Y	FR	Acceptable
Natura 2000 sites	Y	Y	FR	Acceptable

Legend: MCPFE=Ministerial Conference on the Protection of Forests in Europe; OWL=other wooded land; Protective forests-> S+W=soil and water; I+MNR = Infrastructure+Managed natural resources; Availability-> Y=yes; N=no; Source; LSI=large-scale inventory; FD=field description; SM=stand map; RS=remote sensing; FR=maps of forest roles; Data quality-> SE= defined by sampling error; SD=defined by standard deviation.

Forest roles (in some countries known as forest functions), the final dataset, were introduced along with the control method. The mapping of all of Slovenia's forestlands was completed in the year 2000 (Forest Service... 2000).

3.1.2 Possibilities for biodiversity assessment

The existing forestland monitoring has been designed in such a way as to provide the majority of information needed for forestland stewardship at all scales. Consequently, a large portion of it (Table 1) can be used for inferring the status of biodiversity.

As seen from the table, Slovenia is currently in position to provide all the indicators describing the diversity of forestlands that are required by the international community. On the other side, much greater problems are associated with the other wooded lands. Because such lands are, in most cases, not considered forests (in the view of Forest Act 1994) they remain unmanaged and somehow questionable from the biodiversity management point of view.

3.2 National biodiversity assessment

In the year 2001 the Ministry of Environment, Physical Planning and Energy of the Republic of Slovenia initiated a project that aimed to address national biodiversity monitoring. It's the main objectives were:

- To determine the limited number of biodiversity indicators to be assessed across the country regardless of the type of land use and within the boundaries of Natura 2000 sites;
- To design the monitoring framework;
- To assess the costs of monitoring.

The working group was formed of experts who have managed to cover the development of two separate sets of indicators, the development of the monitoring framework and project

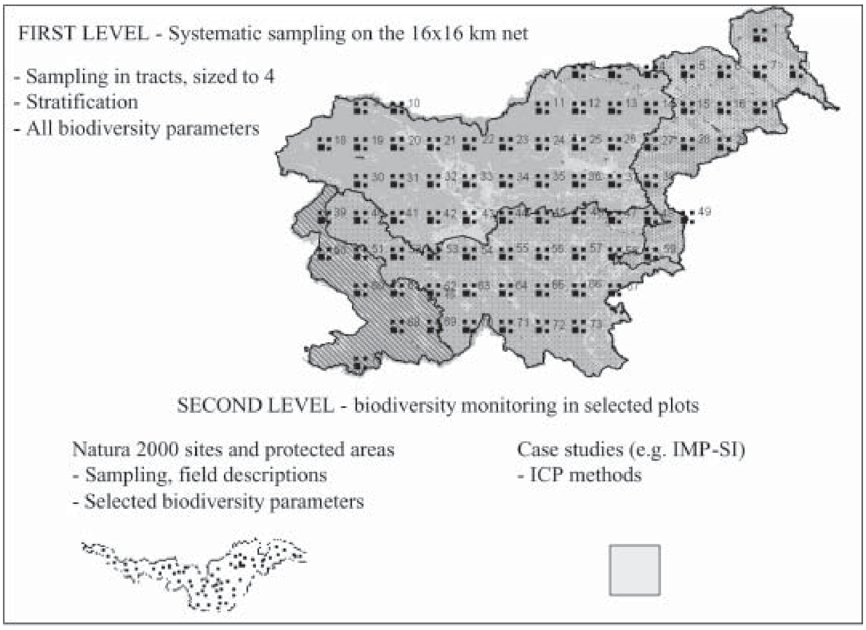

Figure 5. The framework of Slovenia's biodiversity monitoring. Legend: IMP-SI =Intensive Monitoring Program of Slovenia; ICP= International Co-operative Program

management. As the group has been rather small, it has not always succeeded in finding the answers to open questions. Such issues have been quite commonly resolved in collaboration with the experts outside the group.

The most important features of Slovenia's biodiversity monitoring are:

- The monitoring framework (Figure 5) connects two levels. While the first level will address the whole country, the second will deal with the selected plots such as the Natura 2000 network and other monitoring plots. All the monitoring designs developed for specific purposes (e.g. forests, vegetation, mammals, birds, etc.) are considered part of the national biodiversity monitoring.
- The developed list contains four groups of indicators to be assessed on the grid, namely: the richness of habitats, fragmentation, the richness of floral and animal species (alpha and beta level) and their population density.
- The initial density of the grid is set to 16×16 km (74 tracts) and can be (if necessary and possible) condensed at any time. Monitoring itself will cover all three biodiversity levels: landscapes, ecosystems and species. The initial number of sample units will not provide precise estimates, but will allow detection of trends. Unfortunately, this will not be possible in the case of animal species, which will be monitored on the 32×32 km grid.

4. Conclusions and recommendations

Managing for sustainable development, along with the issues related to biodiversity, is perhaps the greatest management challenge of modern times. Not only is the conclusion justified by the fact that the issues related to biodiversity are complex and difficult to investigate, but also by the fact that managing for sustainable development and biodiversity is subject to highly diverse interests. As assumed, strengthening the capacity for the land stewardship (along with biodiversity monitoring) is the essential condition that needs be met prior to a number of other activities. Besides capacity, which appears to be the crucial factor of biodiversity monitoring, much more should be done for the usability of biodiversity data itself. Despite the serious efforts that go into data collecting, day-to-day practice needs nothing else but simple, straightforward recommendations and directions concerning:

- The desired type of monitoring. The question that must be answered: do we need ecological process-oriented monitoring, which provides data on the ecology of a limited number of selected species, or do we need a monitoring system, which is capable of detecting eventual problems in the environment?
- Data credibility. The issue concerns the general acceptance of biodiversity data to be used for several purposes. For instance, if the data sets were used for international comparisons, it is quite natural that they would have to have comparable levels of precision.
- Time-scales. Not only is the time-scale crucial for detecting changes in the environment; it is also the crucial factor of monitoring design and contents. Slovenia, for instance, with a population of two million, lacks human resources in quite a few fields that concern biodiversity.
- Budgeting and so on.

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Multi-Source Forest Inventory Data for Biodiversity Monitoring and Planning at the Forest Landscape Level

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Abstract

Finnish National Forest Inventory data and in particular multisource thematic maps were used as a mean to test different approaches to measure landscape characteristics and complexity as well as biodiversity indicators. We used wall-to-wall output thematic maps from the Finnish multi-source National Forest Inventory (MS-NFI) to evaluate landscape level structure and composition of the Finnish forests. This work constitutes a first effort to estimate forest landscape structure, composition and habitat quality at a full national level. The use of MS-NFI maps with landscape metrics was proven as an efficient and novel method to monitor forest landscape structure and derive biodiversity indicators. In particular, the approach provides a useful tool for monitoring biodiversity when combined with georeferenced species information.

Keywords: Landscape metrics: habitat modelling; forest inventory; biodiversity indicators; GIS.

Introduction

Biodiversity aspects have become increasingly important in the field of environmental policy in Europe. In 1998, the European Commission presented a Communication on a European Biodiversity Strategy. The Sixth Environment Action Programme of the EC (2001a) defines the main objective to be achieved in 2010 related to biodiversity as: ‘to protect and where necessary restore the structure and functioning of natural systems and halt the loss of biodiversity both in the European Union and on a global scale’. Through the Biodiversity Action Plans, the European Community aims to integrate the protection of biodiversity in the areas of Conservation of Natural Resources, Agriculture, Fisheries, and Development and Economic Co-operation (EC 2001a,b). Then, the importance of biodiversity value in Europe is not the issue under discussion but the problem arises on *how to monitor biodiversity* given

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the particularities of the European landscape. Further more, we need to ask which may be the best indicators to use in order to evaluate the positive and negative effects imprinted in the human-induced European landscape.

Over centuries humans have created new habitats by opening and diversifying the European landscape (Angermeier and Karr 1994; Niemelä 2000). For instance, in boreal landscapes, the majority of current deciduous forest has been found on former agricultural lands (Mikusínski and Angelstam 1998). Due to the long human impact and forest ownership structures, which are dominated by small patches owned by private people, the protection of forest biodiversity by definition has to take into account some features that are distinctive to Europe and its landscape (Puumalainen et al. 2003).

In this work, using Finland as a case study, we focus on boreal forest landscapes and take under consideration the use of a valuable existing database – National Forest Inventory (NFI) – that is common to many European countries. We aim at showing the operationality that can be achieved to monitor biodiversity when good quality field data are combined with remote sensing data and spatial analysis. The method presented can turn into a powerful tool for monitoring biodiversity and consequently manage natural resources in countries with NFI data.

At the present in Finland, approximately 95% of the forest land is managed (Ministry of Environment 1999). Management applied to forest stands has substantially changed local forest properties, mostly in terms of tree species composition and the amount of coarse woody debris (Kouki 1994; Essen et al. 1997; Kouki and Niemelä 1997; Löfman and Kouki 2001). Also, regional characteristics, such as the spatial structure of forest landscapes, have been changed (Luque et al. 2004; Kouki and Löfman 1998). As a result of these changes, almost 700 forest-dwelling species are considered as threatened in Finland, mostly due to forestry practices during the twentieth century (Ministry of Environment 1992). Biodiversity issues have gained importance in forestry as a result of the increased awareness of forest landscape changes (Kouki 1994, Mielikäinen and Hynynen 2003), but still there is much to do before forest management meets reasonable goals for forest protection and renewal of biodiversity (Spence 2001).

In order to achieve efficient monitoring systems that focus on the understanding of changes and its linkage to ecological processes a thorough detailed-spatial knowledge of the landscape is needed. Efforts to quantify spatial heterogeneity of landscapes began in the 1980s (Romme 1982; Krummel et al. 1987, O'Neill et al. 1988) but have in recent years accelerated in such a way that there are at present hundreds of indicators that allow some sort of quantification of various aspects of spatial heterogeneity at the landscape level (Baker and Cai 1992; McGarigal and Marks 1994; McGarigal et al. 2002; Gustafson 1998).

We need detailed and reliable information about forest structure and methods to evaluate forest biodiversity over large spatial domains. In this way, the Finnish National Forest Inventory, and in particular the multisource thematic maps (MS-NFI), provide an excellent mean to test different approaches that can lead to a somehow different interpretation of landscape complexity based on landscape metrics using a multilayer level-approach. The challenge set up during this research was to focus on how to understand and measure spatial variations at the forest landscape level. We used wall-to-wall output maps, created by the Finnish national forest inventory (MS-NFI) since 1990, to derive indicators at the landscape level that can provide new insights into biodiversity monitoring.

Methodological considerations

The map format estimates derived from MS-NFI was used to describe the land cover type and its spatial variations. The overall aim of the landscape analysis of the project was to develop

Figure 1. The scale of work (whole country) for Finland by biogeographical regions. The analysis focused on forest volume and stand age derived from the NFI data. Each variable is analysed separately and for each forest category (i.e. spatial distribution for each pine, spruce, birch and other broad-leaved spp.).

quantitative methods to describe the structure of the forest landscape (together with other land use classes). Using MS-NFI thematic maps, landscape metrics were computed for the entire country by biogeographical regions (Figure 1) and for each theme. MS-NFI layers are based on remote sensing data, digital maps and field measurements (for more details see Tomppo 1991, 1996, 1997). Stand age and volumes by tree species were employed, among other relevant variables. Forest structure was based on mean volume of the growing stock by tree species (m^3/ha) in the groups Scots pine (*Pinus sylvestris*), Norway spruce (*Picea abies*), birch (*Betula spp*.) and other broad-leaved trees, mainly aspen (*Populus tremula*) and alder (*Alnus spp*).

In order to computationally handle the large data base at the country level (302 946 km^2 mainland Finland), the existing subdivision of biogeographical regions for Finland were used (Figure 1). The composition and structure of the forest landscape was computed per region and afterwards compared among regions. The analysis was performed using Apack 2.17 software (Mladenoff and DeZonia 2001). The source code was edited to correct some precision problems caused by the large image size and compiled in UNIX. The resolution used for all layers was 50 m (i.e. MS-NFI layers were shifted from the original 25 m pixel size). The purpose of the resolution shift was not only to optimise computational work but also to filter single pixels (i.e. salt and pepper) in the original data. Stand age and volume were the driving variables used for each of the four categories studied (i.e. pine, spruce, birch and other broad-leaved spp).

For the present study, *Patches* are defined as regions that are more-or-less homogeneous with respect to a measured variable. The method used is an *aggregation of like-valued*

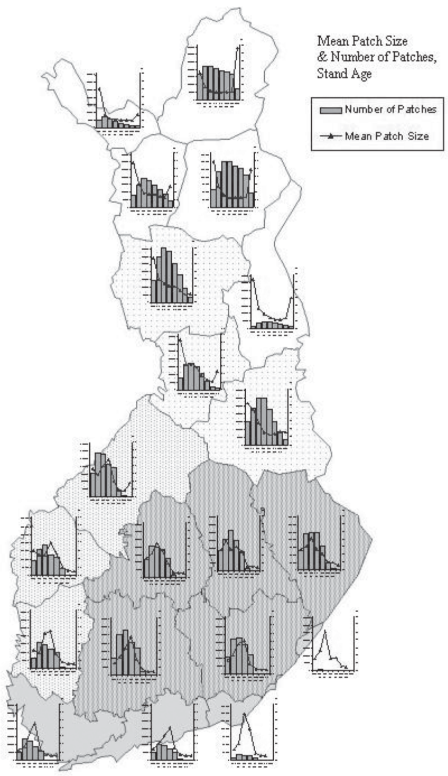

Figure 2. Mean Patch Size (MPS) and Number of Patches (NP) by stand age for each biogeographical region in Finland.

regions. In this way, all adjacent areas that have the same (or similar) value of the variable of interest are aggregated. The common instance is to cluster adjacent (touching) cells in a raster grid, when the cells have the same value. As explained before, the values used are tree stem volume by tree species and stand age derived from the MS-NFI data set, in addition site fertility (Tomppo 1992) was as well considered. Each variable was analysed separately and for each category, so in that sense a *multilayer approach* was used (i.e. a layer showing spatial distribution for the volume of each tree species (m^3/ha) in the groups' pine, spruce, birch and other broad-leaved sp). Thus, the estimates derived from the combination of remote sensing and field data provided by MS-NFI were used to describe spatial structure of habitat types and ultimately to assess habitat quality.

Results

Mean patch size and number of patches as indicators of forest structure

A summary of the number of patches and mean patch size by stand age can be seen in Figure 2. A first attempt of regionalization of landscape patterns at the country level was done (see shaded areas) based on shape and characteristics of the distribution for each individual

Figure 3. MPS by stand age for all forest categories.

region. It is striking to note the consistency in the relationship between number of patches and patch size, which shows consistent regional patterns from north to south in Finland. There is a clear north- south gradient denoting a transition zone from Perä-Pohjanmaa to Kainuu in which mean patch size increases for young ages (21–40 years old forest stands) and decreases for old forest stands (> 100 years old forests). Lapland is the only region where large forest patch sizes of more than 100 years old forest were recorded.

An interpolation based on the central value per region was performed in order to look at the aforementioned trends in patterns using age as a driving variable (Figure 3) and also stand volume on a separate analysis. The striking feature to note in the patch characteristics for the Finnish forest is the small patch size and therefore the high level of landscape dissection. It seems that 80 years old patches constitute a threshold for larger patches (> 2.0 ha), when in most cases the total patch area drops to 0.25 to 1.2 ha (Figures 2, 3). Also, in the 61 to 80 years old class, the proportion of bigger patches in relation to the total patch area decreases. In most regions the distribution of the number of patches in the different age group-classes resembles a Gaussian curve denoting a high number of patches in the intermediate classes (i.e. 21 to 40, 41 to 60 and 61 to 80 years old). In the central regions of the country, 41 to 60 years old class creates the largest number of patches but also the largest patch sizes as we move to the south of the country. In the northernmost regions the patch allocation is very different, showing a more even distribution over the different age categories. The highest proportion of big patches (> 25 ha) is found in the age class 1 to 20 years old (very young stands), and, quite surprisingly, in the age class of more than 140 years old (i.e., very old stands for Finnish forest) patches.

When looking at the patch size distribution based on volume, spruce forest show the largest patches in the north with values in the order of 11 ha, while towards the south the patches are smaller but of higher volume (0.75 ha to 0.95 ha). Still, the problem is that we need to be cautious with this approach, since for example 20 m^3/ha of birch can either be a young seedling stand or also part of a deciduous mix in a mature spruce forest. Also, borders that are not real are created, since much of the spruce and broad-leaved forest may be mixed forests that present then a larger patch size.

Still the approach allows us to understand some general trends for all four tree species categories (pine, spruce, birch, and other broad-leaved) studied. The biggest mean patch sizes are found in the volume classes between 1 to 20 m^3/ha (deciduous) and 1 to 40 m^3/ha (conifers). This might reflect forestry practices: clear-cut areas and the consequent seedling stands that are often quite large. It is very likely, though, that also mixed stands artificially increase the patch size substantially especially within 1–40 m^3/ha volume class.

The mean patch size of pine is bigger (ranging from 0.58 ha to 1.35 ha) than the mean patch size of spruce (0.25 ha to 0.93 ha) when looking at the 41 to 80 m^3/ha and 81 to 120 m^3/ha volume classes. For larger forest volume classes the situation is reverse. Pine has been favoured in forest regeneration during the past decades (except the very last), which could explain this difference in patch sizes. Also, pine is dominant in northern Finland, where stands in general are bigger than in the south. Geographically, the highest mean patch sizes in the volume class 41 to 80 m^3/ha are found in Pohjanmaa (pine) and Lapland (pine and spruce). The highest mean patch sizes of pine in the volume class 81 to 120 m^3/ha are found in Karelia and northernmost Lapland. For spruce, the distribution in this volume class is very even over the country, except for northern Lapland where spruce is scarce.

In the higher volume classes, the decrease in mean patch size is very significant for pine. The largest mean patch sizes are found in northernmost Lapland, Karelia and Perä-Pohjanmaa. Patch size distribution for Spruce is quite even throughout the country, excluding northern Lapland.

The highest mean patch sizes for birch are found in Pohjamaa (volume classes 21 to 40 m^3/ha, 41 to 60 m^3/ha, and 61 to 80 m^3/ha) and Kainuu (classes 61 to 80 m^3/ha and 81 to 120 m^3/ha) (see Figure 4 to have an idea of the distribution). The high mean patch size in Pohjanmaa and Kainuu reflects the high amount of peatlands in those regions. From the southwest coast up to the eastern central Finland there is a belt where the mean patch size of birch is significantly lower than in the surrounding areas. The belt of quite low mean patch sizes could be the consequence of the active measures to remove birch in the past. A larger mean patch size of birch can be found in eastern Finland, where the slash-and-burn agriculture continued up until the 19th century.

In northern Lapland, the mean patch sizes are low, except in the smallest volume class (1 to 20 m^3/ha). There are vast areas of fell birch there, but they are likely to be so small in volume that their effect on the mean patch size is only visible in the smallest volume categories.

Compared to pine and spruce, the mean patch size of birch shows a geographically more heterogeneous pattern and greater variation. In the smallest volume class (1 to 20 m^3/ha), broad-leaved species show a high mean patch size concentration in central Finland. This matches the locations of herb rich forest concentrations in southern Häme and northern Savo. The patches in this volume class are most likely broad-leaved mixture in mature stands. Except for the volume class 1 to 20 m^3/ha, the mean patch size of other broad-leaved species is the lowest among all four tree species studied in every volume class. Compared to birch, the geographic variation is smaller. A clear trend can be found as patch size decreases towards the north, which is due to the climatic requirements of the broad-leaved species.

Forest landscape complexity

In the overall, the Finnish forest was characterized by intermediate values of complexity - between 1.50 and 1.60 (Figure 5) in terms of fractal dimension. Despite the narrow interval of

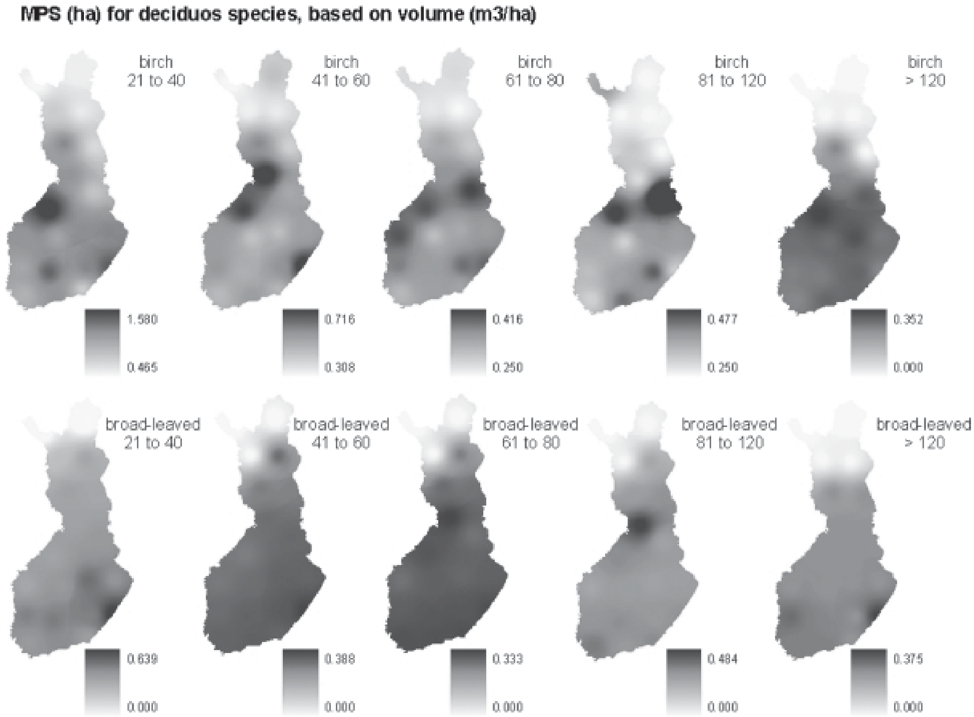

Figure 4. MPS based on volume (m^3/ha) of birch (top) and other broad-leaved species (bottom).

variation, fractals still show a more complex forest landscape in Perä-Pohjanmaa contrasting with lower values (i.e. less complexity) to the south, in particular the south-east. Complexity was found low for forest patches of the dominant categories of spruce and pine, contrary to birch and broad-leaved forest species that showed higher complexity values for stands with low volumes of the species. Towards the west of the country, higher values of forest dissection (i.e. many small forest patches) are evident.

When looking at both indicators – perimeter-area relation and edge density – the contrast is noticeable between the more complex and continuous landscape to the north (in particular the Lapland's region) and the more dissected and managed situation in the rest of Finland (particularly to the south). The Finnish forest in this sense shows a landscape that is well interspersed but highly dissected as denoted by the very low values of contagion for all regions (Figure 6). On the whole, the Finnish forest landscape lacks heterogeneity as is showed by the low values of dominance and diversity and a great deal of homogeneity in the edge distribution (Figure 6).

Habitat suitability models

One example of *application from the landscape level analysis* developed is the production of site quality models. These models when produced at the country level allow to learn about the

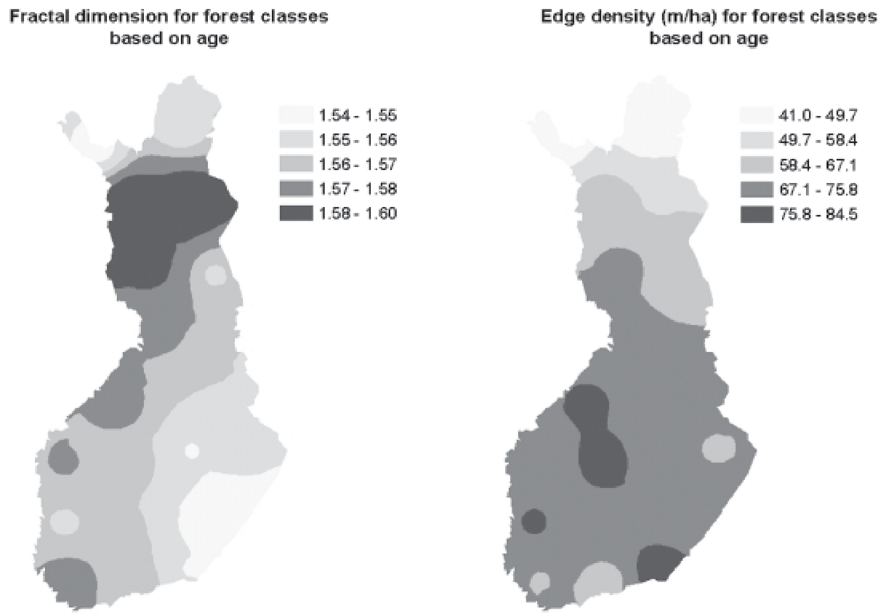

Figure 5. Fractal dimension and edge density for the forest landscape.

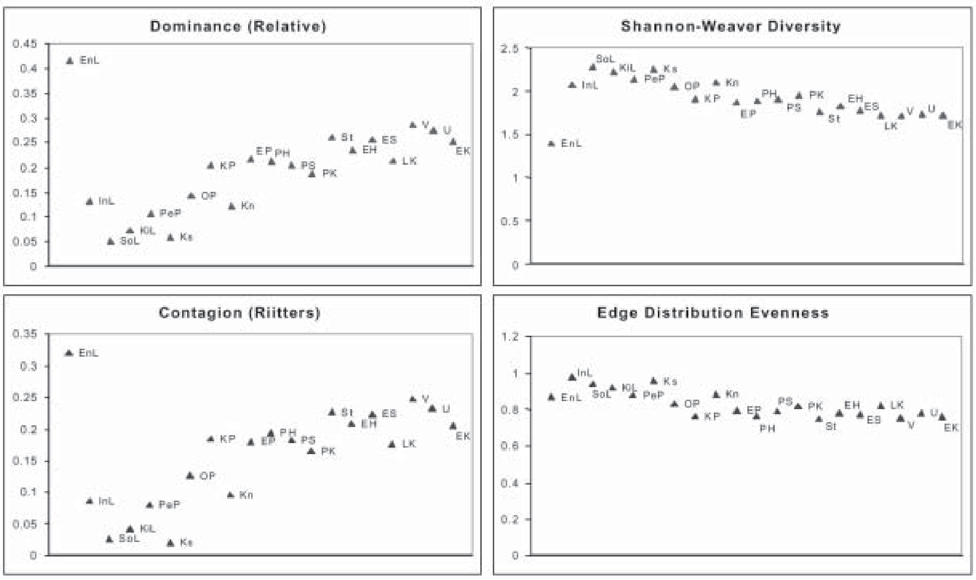

Figure 6. Contagion index and other indices derived from information theory by regions for all age classes (i.e. 11 categories) (refer to the region map (Figure 1) for location of the regions in Finland).

Table 1. Sets of criteria and constraints used for the quality sites maps.

	MS-NFI Thematic Map	Constraint for north	Constraint for south
Spruce model	Dominant tree species	Spruce	Spruce
	Stand age	61–80 years > 80 years	61–80 years > 80 years
	Stand volume	> 20 m^3/ha	> 40 m^3/ha
	Potential productivity	> 2 m^3/ha/a	
Mixed model	Spruce model output	Output for north	Output for south
	Volume of deciduous species (birch and other broad-leaved)	> 20 m^3/ha	> 20 m^3/ha

extent and distribution of sites at the regional level that may have potential biodiversity value for the Finnish forest. The spatial models were built in order to test the response from the indicators developed at the landscape level and to look at potential useful patterns that can be used for management applications at the regional level. Therefore, the results cannot be considered as conclusive. The example presented here is just a test, based on the statistics obtained from the country level results in relation to landscape metrics and MS-NFI data. Moreover, the variables considered are rather simple and therefore the sets of criteria selected for spruce and mixed forests are not meant as a comprehensive evaluation of the quality of Finnish forests.

The sets of criteria selected can be seen in Table 1. In addition, roads, agricultural areas and populated areas were each extracted separately from MS-NFI and a distance buffer was computed to test the effect of human pressures that may affect the continuity of quality sites. The buffers were computed as follows:

- distance from roads: 50 m
- distance from agricultural areas: 100 m
- distance from populated areas: 300 m

Several approaches were considered based on these different constraints and buffers. An example of the logic used can be seen in the flowchart that presents the steps followed for southern Finland considering the aforementioned buffers (Figure 7). The proportion of high quality sites of land area for the spruce model ranges from 12% in the south to 5% in the north (Figure 8). For mixed forest, the proportion in the south is approximately 5% and 1.5% in the north. The proportions seem reasonable considering the lack of old-growth forests in Finland, dominance of pine in the north and scarcity of old deciduous trees.

The amount of high quality sites, according to the set of criteria considered, was found larger in the age class group of 61 to 80 years old forest than in the age class > 80 years old. Buffers denoting human pressures have more importance in the south than in the north. In the south, buffering almost halves the amount of high quality sites.

Given the impact that buffers have in the south, it will be important to consider the effect of these types of pressures in the quality of natural habitats. The continuous surfaces of the proportion of high quality sites (biogeographical level regions interpolated over the country) show that for the age class of 61–80 years old, central southern Finland is clearly the area of

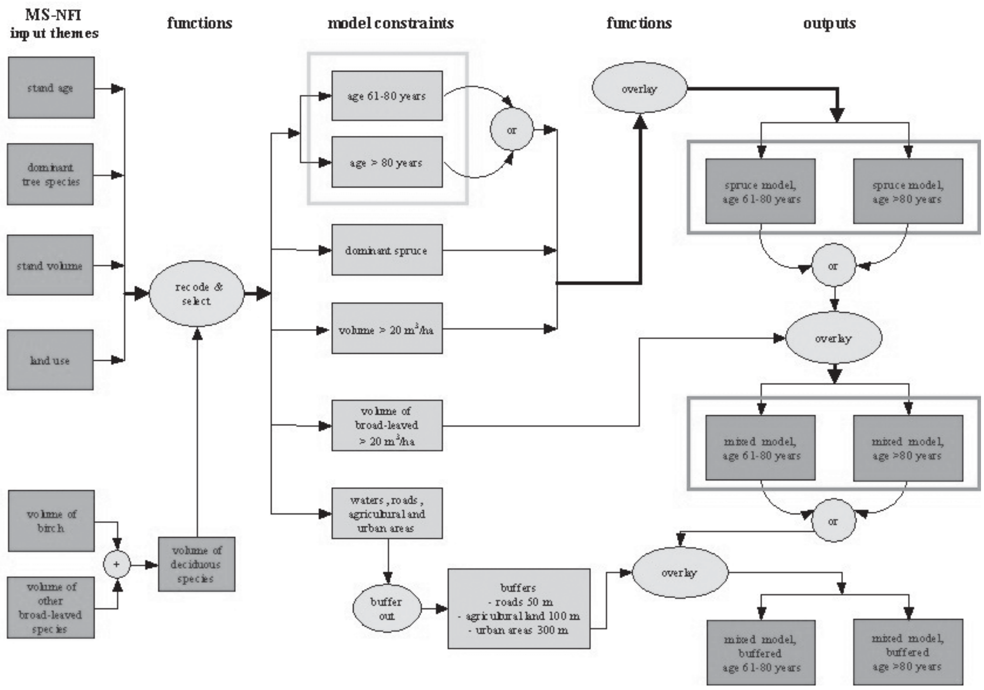

Figure 7. Flowchart of the variables, functions and constraints used in the habitat models for southern Finland.

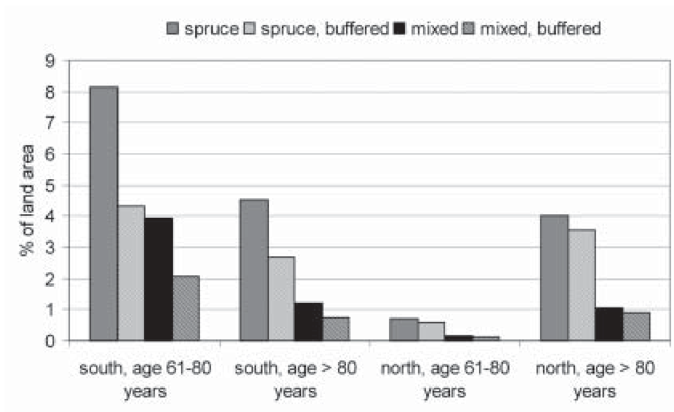

Figure 8. Proportion of high quality sites (% of land area) in southern and northern Finland. Using two forest age criteria (61–80 and >80 years), with and without buffers (roads 50 m, agricultural areas 100 m, populated areas 300 m).

high proportion of good quality sites, considering both spruce and mixed forest (Figure 9), while for old-growth forest (age class > 80 years old) it does not seem to be a clear delimitation of a region. The one region that shows a great proportion of high quality sites in all four cases (spruce, mixed, and two age criteria) is Pohjois-Savo. This region has a

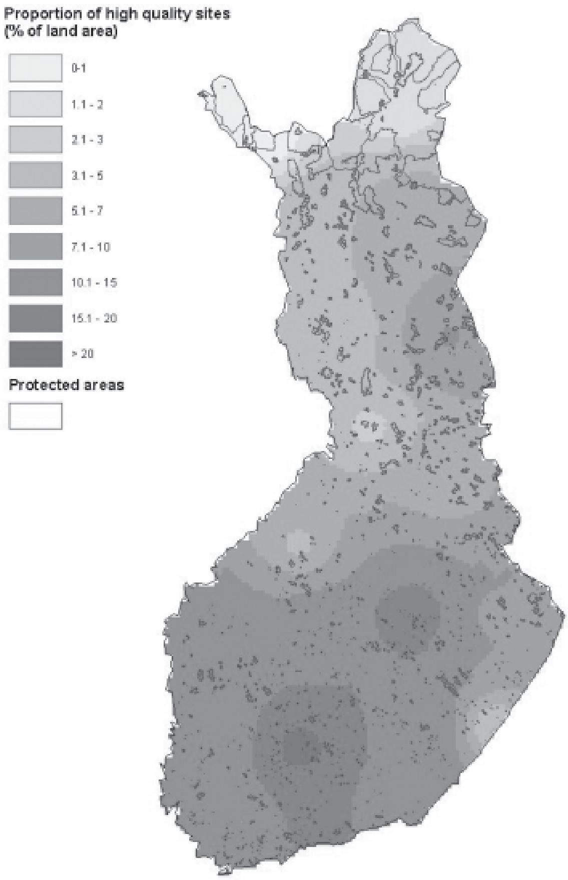

Figure 9. Suitable areas according to dominant spruce and deciduous 'mature' forest in relation to protected areas in Finland. Towards the south the lack of representativeness of protected areas in more suitable regions is evident due to the small fragment size of protected areas.

concentration of herb-rich sites due to a rare rock type in the bedrock. The fact that the model picked up this region denotes a big achievement in the method, even considering the few variables and constraints used in the exercise. In the same way, some other concentrations of herb-rich forests – e.g. Kuusamo and Etelä-Häme – were evident in the site quality model output. These regions showed relatively high concentrations of high quality sites. On the other hand, no apparent effect of the previous slash-and-burn practices can be seen in the amount and distribution of the mixed sites.

Relation between protected areas and high quality sites

In northern Finland the proportion of high quality sites in protected areas is in most cases higher than in southern Finland (Figure 9), despite the reverse situation if all land area, not just protected, is considered. Furthermore, in southern Finland the 61 to 80 years old forests make up the greater proportion of high quality sites in the protected areas, whereas in northern Finland the age class > 80 years old forest is the dominant category. Therefore, it seems that the lack of old forests in southern Finland stands also in the forest structure within protected areas.

It must also be noted that buffers do not affect much the amount of high quality sites within protected areas. In northern Finland, the only category where the amount of buffered sites is less than 90% of the unbuffered sites is the privately protected areas. This was to some extent expected, because of the size of these areas that is often quite small. In southern Finland, the amount of buffered sites is between 80 to 90% of the unbuffered sites, the exceptions being herb-rich forests (mostly very small patches) and areas protected by the Finnish Forest and Park Service (Metsähallitus).

Some protected areas, such as old forest conservation areas and areas in the old forest conservation programme, have a significant proportion of high quality sites (approximately 15–35%) (Figure 9). However, a considerable amount of high quality sites of more than 80 years old forest, in particular in southern Finland, is located outside any protection status. Here it is important to note that the variables included in the models are rather simple and are not meant to provide a comprehensive evaluation of the quality of Finnish forests. It is important also to keep in mind that with this simple type of site quality model it is not possible to evaluate properly the quality of the protected areas, since, for example, old pine forests are not considered in the model. The importance of pine forests is likely to be greatest in northern Finland. Still, the result is an important factor to consider when evaluating possible future expansion of existing protected areas.

Conclusions

Results obtained within the framework of this research provided new insights into the more complex underlying processes, mostly throughout the understanding and quantification of the structure of the landscape (see also Luque et al. 2004; Luque et al. 2002). The landscape patterns obtained showed a landscape transformed into a mosaic of managed forest stands in which species composition is very homogeneous and age distribution of stands quite even. The small patch size and therefore the high level of landscape dissection are the dominant characteristics of the Finnish landscape. In the overall, the forested landscape presents high levels of patchiness at both, regional and country level and seems to be a consequence of not only the current forest management but also the past history of forestry, natural and human-induced disturbances as well as natural variation of soil and site fertility. In the same way, the landscape level patterns were quite distinctive from north to south and among regions. In the overall the trends found revealed different management history and different driving environmental factors.

Contrary to the assertions presented in the Tema Nord report (Stokland et al. 2003: 101) NFI data can be used as an excellent data base for landscape level analysis, in particular to accurately monitor large regions. The use of thematic maps derived from MS-NFI together with landscape metrics and georeferenced species data has proven as an efficient and operational method for monitoring biodiversity, in particular at the forest landscape level.

The habitat quality models developed in the present work were rather simple and just used as a mean to test the method. Results from this work provided important insights in terms of spatial variation in forest structure for boreal managed systems and also denoted the need for additional parameters and information that we aim to improve and develop. At the present, a more detailed evaluation of protected areas in southern Finland is under way, and different models are under analysis (www.metla.fi/hanke/3241/prohab/). The approach been developed is more close to an ‘Habitat index’ value in terms of habitat quality as a surrogate for biodiversity value. Particular importance is focus on the integration of other variables provided by NFI's such as dead wood, stand damages, management history, key biotopes and additional vegetation information from permanent plots in order to add value to NFI for biodiversity studies.

Data integration from species to forest structure information is crucial for the development of diagnostic models that incorporate our up-to-date understanding of all relevant processes leading to spatial and temporal changes. Studies are needed in countries with high quality historical data such as Finland with MS-NFI to allow predictions of future changes and subsequent effective management. It is expected that management based on high quality data such as the one derived from NFI's in Europe, in tandem with a good interaction among end users and policy makers, will help to meet reasonable goals for protection and subsequently an appropriate renewal of biodiversity.

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First Results of the ICP Forests Biodiversity Test-Phase in Italy

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Abstract

In the frame of the ICP Forests biodiversity test-phase and anticipating the new tasks required by Regulation (EC) n. 2152/2003 *Forest Focus*, a first biodiversity survey has been carried out in some selected Level II EU/ICP Forests permanent plots belonging to the Italian National Network for Forest Ecosystems Monitoring *CONECOFOR*, during summer 2003. The key parameters, assessed at stand level in 12 plots from Alps to Sicily, include (1) vegetation, (2) epiphytic lichens, (3) stand structure, (4) deadwood, (5) insects communities, (6) naturalness and (7) landscape biodiversity.

Keywords: biodiversity; vegetation; lichens; deadwood; naturalness.

1. Introduction

After the formal approval of the UN Convention on Biological Diversity (UNEP 1992), many international and national bodies have made efforts developing reliable approaches for the assessment of forest biodiversity; several meetings have taken place for this purpose (e.g. Puumalainen et al. 2002). In this framework, the first projects testing specific methods in the field have started very recently. In the frame of the ICP Forests biodiversity test-phase (UN/ECE and EC 2003) and anticipating the new tasks required by Regulation (EC) n. 2152/2003 *Forest Focus*, a first biodiversity survey has been carried out in some selected Level II EU/ICP Forests permanent plots belonging to the Italian National Network for Forest Ecosystems Monitoring *CONECOFOR* (Petriccione and Pompei 2002), during summer 2003.

The key parameters, assessed in 4–12 plots, include (1) vegetation, (2) epiphytic lichens, (3) stand structure, (4) deadwood, (5) insects communities, (6) naturalness and (7) landscape biodiversity.

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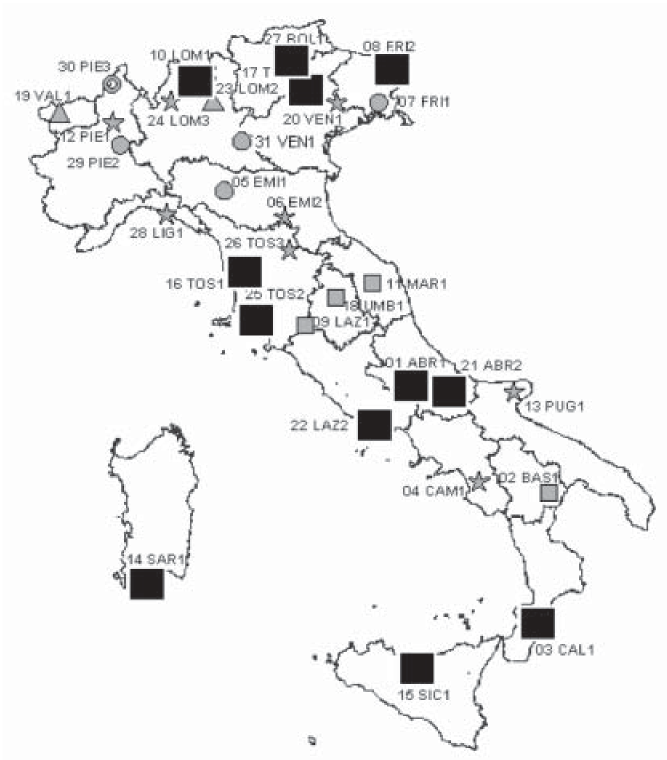

Figure 1. Location of selected plots for biodiversity assessment (black squares) of the Italian National Network for Forest Ecosystems Monitoring CONECOFOR.

2. Data and methods

Selection of plots for the test-phase is based on the Italian National Network for Forest Ecosystems Monitoring *CONECOFOR* (Petriccione and Pompei 2002), operative since 1995 and currently including 31 Level II permanent plots in the framework of the UN-ECE International Co-operative Programme on Assessment and Monitoring of Air Pollution Effects on Forests (ICP Forests); 13 plots are also classed as “bio-monitoring sites”, in the framework of the UN-ECE International Co-operative Programme on Integrated Monitoring of Air Pollution Effects on Ecosystems (ICP IM). CONECOFOR Programme is co-ordinated at national level by the Ministry for Agriculture and Forestry Policy (National Forest Service), following EU Regulations on the protection of forests against atmospheric pollution. The Programme also operates within the framework of the UN-ECE Geneva Convention on Long-Range Trans-boundary Air Pollution *(LRTAP*) and of the Resolutions of the Ministerial Conferences on the protection of forests in Europe.

Selected plots (Figure 1, Table 1) are distributed overall the national territory and include the main forest types, classified according to the BEAR Forest Types for Biodiversity Assessment: *Picea abies* Mountain woodland on Central and Eastern Alps (4 plots), *Fagus sylvatica* Mountain forest on Central and Southern Apennines (2 plots), *Quercus ilex* Mediterranean woodland on Thyrrenian coast and Sardinia (4 plots), and *Quercus cerris* eu-

Table 1. Selected plots for biodiversity assessment (ICP Forests codes, ICP IM codes, National CONECOFOR codes, Altitude in m a.s.l., Average Annual Temperature, Average Annual Precipitation, BEAR Forest Types for Biodiversity Assessment).

ICPFor	ICP IM	*National*	Official name	Alt. (m)	T (°C)	P (mm)	Tree age	Dominant tree species	FTBA
01	IT05	*ABR1*	Selva Piana	1500	10.0	1300	110	*Fagus sylvatica*	1N.3b
03	IT06	*CAL1*	Piano Limina	1100	10.0	1500	110	*Fagus sylvatica*	1N.3b
08	——	*FRI2*	Tarvisio	820	6.0	1500	70	*Picea abies*	3N.1
10	IT10	*LOM1*	Val Masino	1190	8.0	1300	50	*Picea abies*	3N.1
17	IT03	*TRE1*	Passo Lavazè	1775	5.0	800	110	*Picea abies*	3N.1
27	IT01	*BOL1*	Renon	1740	4.0	970	110	*Picea abies*	3N.1
15	——	*SIC1*	Ficuzza	940	13.0	800	30	*Quercus cerris*	1N.7
21	——	*ABR2*	Rosello	960	8.5	1000	130	*Abies alba, Q. cerris*	1N.7
14	——	*SAR1*	Marganai	700	14.0	900	110	*Quercus ilex*	2N
16	IT12	*TOS1*	Colognole	150	15.0	900	30	*Quercus ilex*	2N
22	——	*LAZ2*	Monte Circeo	190	15.5	900	30	*Quercus ilex*	2N
25	——	*TOS2*	Cala Violina	30	15.0	650	40	*Quercus ilex*	2N

mesotrophic woodland on Central Apennines and Sicily (2 plots). They include also the highest biodiversity forest plant communities (except for the Alps) and some of the sites with available facilities and active research groups.

All biodiversity surveys are performed at stand level, on the basis of the 2500 m^2 analysis area established in each permanent plot, where also *routine* analyses are carried out since 1995 (crown condition, chemical content of leaves and soil, tree growth, atmospheric deposition and air chemistry, micro-climate, etc.).

Methods for field surveys follow that adopted by the ICP Forests Working Group on Biodiversity in 2003 (harmonized methodologies are available for assessments in the fields of epiphytic lichens, stand structure, deadwood, forest stratification and ground vegetation at plot level, Fischer and Neville 2003, ICP Forests 2003).

3. Results

Seven key biodiversity indicators have been assessed in 4-12 permanent plots (Table 2). It is foreseen to complete the survey in all (12) selected plot until 2004. Preliminary results are reported in the this paper, according to a first quick evaluation.

3.1 Vegetation

Phytosociological knowledge of plant communities and their synecological allocation are the reference basis of the CONECOFOR Programme: vegetation surveys are performed in all CONECOFOR plots (Petriccione 2002) and have been performed in most of them for 6-8 years, following two fundamental approaches: (1) phytosociological (plant community level) and (2) dynamical (population level). According to a syntaxonomical analysis, 18 plant

Table 2. Biodiversity indicators assessed for the selected CONECOFOR plots: vegetation (no. of vascular species, lichens (Index of Lichen Biodiversity), stand structure (omplexity index), deadwood (total amount in m3/ha), insects (total no. of Coleoptera and Diptera species), naturalness (average value), landscape (Biological Territorial Capacity in Mcal/m2/y) (n.a.: not assessed).

PLOT	vegetation	lichens	stand str.	deadwood	insects	naturalness	landscape
ABR1	48	n.a.	8.87	n.a.	n.a.	4.6	n.a.
CAL1	73	n.a.	7.59	6.4	169	n.a.	n.a.
FRI2	78	16	18.34	n.a.	n.a.	0.2	n.a.
LOM1	92	29	98.56	n.a.	n.a.	0.2	n.a.
TRE1	31	116	12.91	n.a.	n.a.	4.7	7.65
BOL1	54	n.a.	n.a.	n.a.	n.a.	2.8	6.26
SIC1	81	n.a.	39.75	25.0	140	n.a.	n.a.
ABR2	66	n.a.	n.a.	38.0	95	4.8	n.a.
SAR1	38	n.a.	57.88	12.0	135	n.a.	n.a.
TOS1	20	29	124.88	n.a.	n.a.	3.3	n.a.
LAZ2	54	n.a.	n.a.	n.a.	n.a.	4.0	n.a.
TOS2	29	63	n.a.	n.a.	n.a.	5.0	n.a.

Table 3. Vegetation data (cumulated for community and population level) for the selected plots for biodiversity assessment.

PLOT	Vegetation survey years	Syntaxon	Vascular species no.	Tree layer species no.	Vegetation main dynamical tendency
ABR1	1996–2003	*Polysticho-Fagetum*	48	1	regeneration
CAL1	1996–2003	*Aquifolio-Fagetum*	73	3	fluctuation (regeneration)
FRI2	1996–1999	*Veronico uticifoliae-Piceetum*	78	5	regeneration
LOM1	1996–2003	*Veronico uticifoliae-Piceetum*	92	14	regeneration (fluctuation)
TRE1	1996–1999	*Homogyno-Piceetum*	31	2	fluctuation
BOL1	2002	*Homogyno-Piceetum*	54	5	regression
SIC1	1996	*Quercetum gussonei*	81	1	regression
ABR2	2002	*Aceri lobelii-Fagetum abietetosum albae*	66	12	fluctuation
SAR1	1996–1999	*Viburno-Quercetum ilicis*	38	5	regression
TOS2	1999	*Viburno-Quercetum ilicis*	20	5	fluctuation
TOS1	1996–2003	*Orno-Quercetum ilicis*	54	15	regression
LAZ2	2002–2003	*Orno-Quercetum ilicis*	29	6	regeneration

communities are represented in the CONECOFOR permanent plots, grouped in three classes *(Querco-Fagetea, Vaccinio-Piceetea* and *Quercetea ilicis).*

Analysis of species richness at community and population level (Tables 2 and 3) shows that the total number of vascular species per plot varies between 20 and 92, the lowest values occurring in Mediterranean evergreen forests and primary spruce forests, whereas the highest values are in secondary spruce forests.

Table 4. Stand structure indicators for the selected plots for biodiversity assessment (no.: number of tree species; H: Shannon diversity; S: Simpson diversity; SD: diameters Standard Deviation; CV: diameters Variation Coefficient, VE: Vertical Evenness; A: Spèecies Profile Index; HC: Holdridge Complexity Index; n.a.: not assessed).

PLOT	SPECIES DIVERSITY			HORIZONTAL DIVERSITY		SIZE DIVERSITY		VERTICAL DIVERSITY		COM-PLEXITY
	no.	H	S	Cox	Pielou	SD	CV	VE	A	HC
ABR1	1	0.00	0.00	2.28	0.84	12.2	59.9	0.92	1.08	8.87
CAL1	2	0.17	0.05	1.01	0.90	19.3	56.8	0.99	1.06	7.59
FRI2	2	0.21	0.06	0.41	0.84	9.5	27.6	0.96	0.61	18.34
LOM1	9	2.01	0.66	n.a.	n.a.	13.6	77.8	0.98	2.33	98.56
TRE1	2	0.08	0.02	0.41	0.56	14.5	36.9	0.94	0.78	12.91
SIC1	3	0.22	0.06	1.09	0.86	3.9	20.4	0.80	0.73	39.75
SAR1	5	1.13	0.37	n.a.	n.a.	9.4	64.1	0.98	1.68	57.88
TOS1	13	2.15	0.70	n.a.	n.a.	7.3	71.3	0.77	2.21	124.88

Analysis of the main dynamical tendencies (Table 3) shows that fluctuation is the commonest ongoing process (occurring mostly in beech and primary spruce forests). Regeneration is also widespread, following the recent general abandonment of wood exploitation and coppicing, whereas regression and degeneration have been identified only in a few plots. Kind of dynamical tendency does not seem linked with species richness values. The first vegetation changes seen during the first 6–8 years of investigation are slight and of very low significance. The temporal variation, however, is generally positive, with a fair increase in the number of species. Further assessment is required to evaluate the ongoing trends.

3.2 Epiphytic lichens

Lichen survey has been performed in 5 selected plots (Table 2), according to the harmonized ICP Forests (2003) methodology, on 12 sampling trees selected randomly by tree numbers from the existing data base (taken in account pre-stratified groups). Lichen richness and frequency are observed and the Index of Lichen Biodiversity (ILB, Asta et al. 2001) is calculated. ILB provides information on long-term effects of air pollution, eutrophication and other anthropogenic factors on lichens; it is also a measure of biodiversity and it can be used representing the lichen species richness of a survey area.

A total number or ca. 100 species has been observed, 9 of them very rare and with high conservation value (e.g. *Calicium viride, Hypotrachyna laevigata, Mycomicrothelia confusa)*: among the assessed plots (Table 2), species richness varies between 4 (secondary spruce forests) and 26 (primary spruce forests); a similar range occurs for IBL values, from 16 to 116.

3.3 Stand structure

Stand structure survey has been performed in 8 selected plots (Table 2), according to the harmonized ICP Forests (2003) methodology. A total number of 9 indices of structural diversity (grouped in horizontal, vertical, size and complexity types) have been calculated (Table 4): high values occur in a secondary spruce forest (LOM1) and a Mediterranean

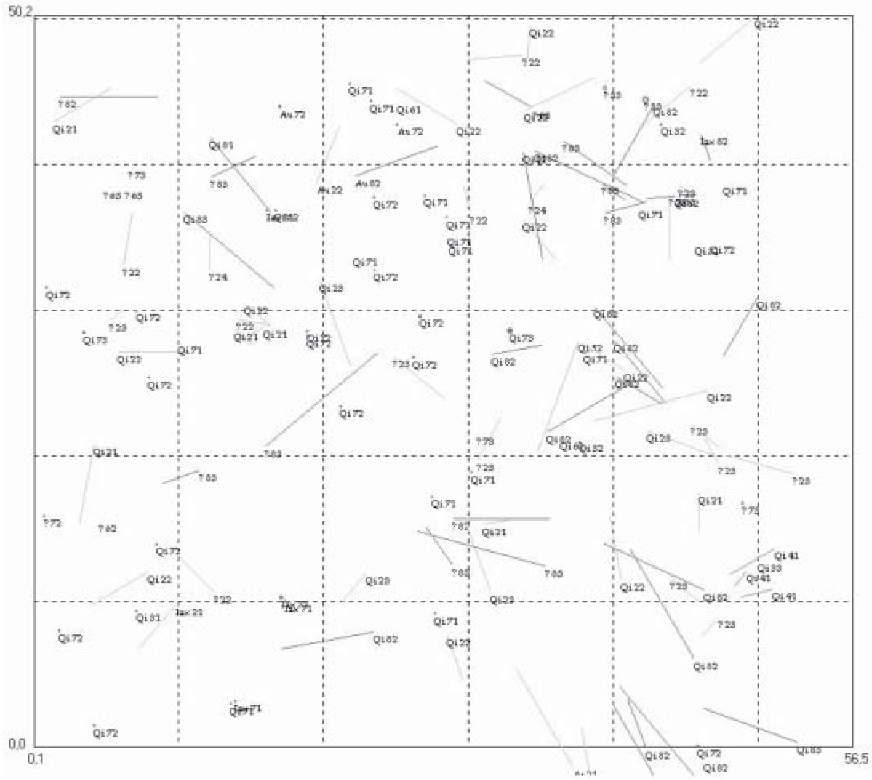

Figure 2. Deadwood graphs for CONECOFOR plot SAR1. Qualitative characteristics of dead trees are expressed by a code number as follows (i.e.): 82 = DWP (Dead Wood Position) + RDW (Rate of Decaying Dead Wood) (Qi = *Quercus ilex*; Au = *Arbutus unedo*; Iax = *Ilex aquifolium*; ? = others).

evergreen oak forest (TOS1), with high tree species richness due to local ecological factors; generally, all indices are statistically correlated.

3.4 Deadwood

Deadwood survey has been performed in 4 selected plots (Table 2), according to the harmonized ICP Forests (2003) methodology: localisation, volume and decay state (Mason 2002) of dead downed trees, lying coarse and fine wood pieces, stumps and standing deadwood have been calculated. Data processing and graphs (Figure 2) have been obtained using a specific program developed by National Centre for Study and Conservation of Forest Biodiversity (Verona, Italy).

Table 5 shows the obtained results for 2 plots, as an example of methods and data processing; these plots have relatively low volumes of large-sized deadwood, if compared to other areas with higher naturalness (Paterken 1996, Mason 2002), whereas the decay state of deadwood is enough advanced to allow the presence of micro-habitat with high ecological value. Generally (Table 2), the coppice forests (plots SIC1 and ABR2) show higher amount of deadwood than the high forests (plots CAL1 and SAR1).

Table 5. Volume and qualitative characteristics of dead wood in the CONECOFOR permanent plots CAL1 and SAR1 (RDW= Rate of Decaying Dead Wood).

PLOT	Volume (m^3) dbh < 10cm	%	Volume (m^3) dbh ≥ 10cm	%	Total volume (m^3)	RDW < 3	%
CAL1	0.10	53.6	1.40	46.4	1.5	0.98	65.2
SAR1	1.25	39.5	1.92	60.5	3.17	1.897	59.9

PLOT	RDW ≥3	%	Standing wood (m^3)	%	Fallen wood (m^3)	%
CAL1	0.53	34.8	1.14	75.9	0.36	24.1
SAR1	1.268	40.1	2.05	64.9	1.11	35.1

3.5 Insects communities

Insect survey has been performed in 4 selected plots (Table 2), by direct collecting methods, pitfall and Malaise traps. A total number of ca. 4–500 species has been recorded: 150 morpho-species of *Coleoptera,* belonging to 39 families, and roughly 250 morpho-species of *Diptera,* belonging to 54 families. A new species of genus *Pseudogonia (Diptera: Tachinidae)* has been observed and described *ex novo* in SAR1 plot (Cerretti, 2004) and three species of *Tachinidae (Diptera)* have been recorded in Italy for the first time (Cerretti, in prep.).

3.6 Naturalness

The level of naturalness, defined like the degree of self-functioning of the natural processes and the intensity of human interventions on the function and structure of ecosystems, is a very important criterion for maintenance, conservation and enhancement of biological diversity in forest ecosystems (MCPFE 2002). In a wide context, naturalness can be considered equivalent to the concept of environmental quality (Ploeg and Vlijm 1978, Margules and Usher 1981, Greco and Petriccione 1991) and measured by specific indicators based on vegetation composition and structure (Petriccione 1994).

A new methodology for a rapid assessment of naturalness at stand level has been tested in 9 selected plots (Table 2), by direct observation in the field: (1) comparing real and potential vegetation types and micro-communities, (2) assuming like reference the nearest comparable undisturbed stand, (3) evaluating the native origin of species composition and (4) calculating values of six specific indices (Petriccione 1992, 1994, vegetation naturalness, chorotypes coherence, native species, species richness and diversity, evenness and dominance). Preliminary results (Table 6) point out that forests in fluctuation stage show the highest naturalness values (4.7–5.0, plots ABR1, TRE1, TOS2), whereas lower values occur in the case of forests in regression stage (2.8–3.3, plots BOL1 and TOS1). Secondary forests resulting by native species plantation are placed in a very peculiar situation, with the lowest naturalness level (0,2 in plots FRI2 and LOM1), slowly increasing by a small natural regeneration process in the course.

Table 6. Naturalness level (from 0.0 to 5.0), based on simple average of indices values (according to 5 classes: 1: 0–20%; 2: 21–40%; 3: 41–60%; 4: 61–80%; 5: 81–100%).

PLOT	vegetation naturalness	chorotypes coherence	native species	species richness	species diversity	evenness (dominance)	NATURAL-NESS
ABR1	5	5	5	4	5	4	4.6
FRI2	—— secondary community (plantation by native species) ——						0.2
LOM1	—— secondary community (plantation by native species) ——						0.2
TRE1	3	5	5	5	5	5	4.7
BOL1	2	4	5	2	2	2	2.8
ABR2	4	5	5	5	5	5	4.8
TOS1	3	5	5	2	2	3	3.3
LAZ2	3	5	5	4	3	4	4.0
TOS2	5	5	5	5	5	5	5.0

3.7 Landscape biodiversity

Landscape biodiversity has been performed in 2 selected plots (Table 2), following the methodology proposed by Ingegnoli (1999, 2002). Parameters for calculation of Biological Territorial Capacity (BTC), based on main landscape characteristics and vegetation structure and function, have been surveyed in both the analysis area and in a wider range. Preliminary data on land quality (63.43–74.86%), plant biomass (517–759 m^3/ha) and BTC (6.26–7.65 Mcal/m^2/y) have been calculated, so to put the investigated spruce high forests in an intermediate position (generally, BTC values for the same forest types range from 2.0 to 11.0, Ingegnoli 1999).

4. Conclusions

First results of the 2003 biodiversity test-phase show that: (1) in very short time and relatively not expensive costs is possible obtaining valuable indications on biodiversity status of forest communities; (2) the ICP Forests (2003) harmonized methods (vegetation, lichens, stand structure and deadwood) are reliable and effective; (3) the new tested parameters (naturalness, landscape diversity) have shown a good performance, expressing a high synthesis capability; (4) the surveyed forest communities have shown very high values for nature conservation (Community interest or priority habitats and species occur on 8 of 12 plots, according to the Habitat Directive (EEC) n. 92/43); (5) finally, qualitative results of surveys are very important to increase the basic scientific knowledges (in a few months of survey on lichens and insects, 1 species new for science and 20 new or very rare for Italy have been discovered).

The obtained results will be the basis for: (1) a more detailed survey that will be performed in 2004, in the framework of the Pan-European project *ForestBIOTA* (a joint project by 20 European Countries, based on 110 EU/ICP Forests permanent plots, collecting data on four main biodiversity indicators in standardised way, Fischer and Neville 2003); (2) an integrated and combined evaluation of biodiversity of forest ecosystems in Italy, considering his relationships with climate, ozone, deposition, soil and leaves chemistry, in the framework of the national Integrated and Combined strategy 2001–2005 (Ferretti 2002).

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Monitoring Biodiversity at a Wide Land Scale to Support Sustainable Planning and Policy: The Proposal of a Key Indicator Based on Vegetation Cover Data Deriving from Maps

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Abstract

Increasing biodiversity beyond the site scale is now required. Planning tools have to be able to monitor biodiversity level at the land scale as to decide how and where to intervene. Nowadays there are some key biodiversity indicators suggested by the main international associations. These indicators measure the presence or absence of an area under national or state park control. They measure *a de jure* situation and this is considered a limit. This paper presents a simple methodology to evaluate the biodiversity level, starting from the vegetation cover land classification (pasture lands, rural lands, different types of forests…) where forest represents one of the best performance in biodiversity. Forest cover is considered a good indicator for biodiversity or for biodiversity start up. The methodology, fitted for planning issues first, has been tested on the plain area of Lombardia where now the regional government, with the scientific support given by a scientific committee, is planning ten new large forests. One of the strengths of this planning methodology is its easy application at the land scale, thanks to the use of data coming from GIS-map where land covers and destinations have been signed for the whole plain area. This indicator is a useful tool for policy and planning issues and aims due to the availability of data and its easy application. The results, at the beginning and, surely, to improve, encourage the applications and give a better outputs than other key indicators like, for instance, the OECD indicators for biodiversity monitoring: in the paper it will be offered a comparison between the two monitoring indicators at the wide scale.

Keywords: Biodiversity; indicator; forest; land cover; planning, decision making, GIS, botanic types.

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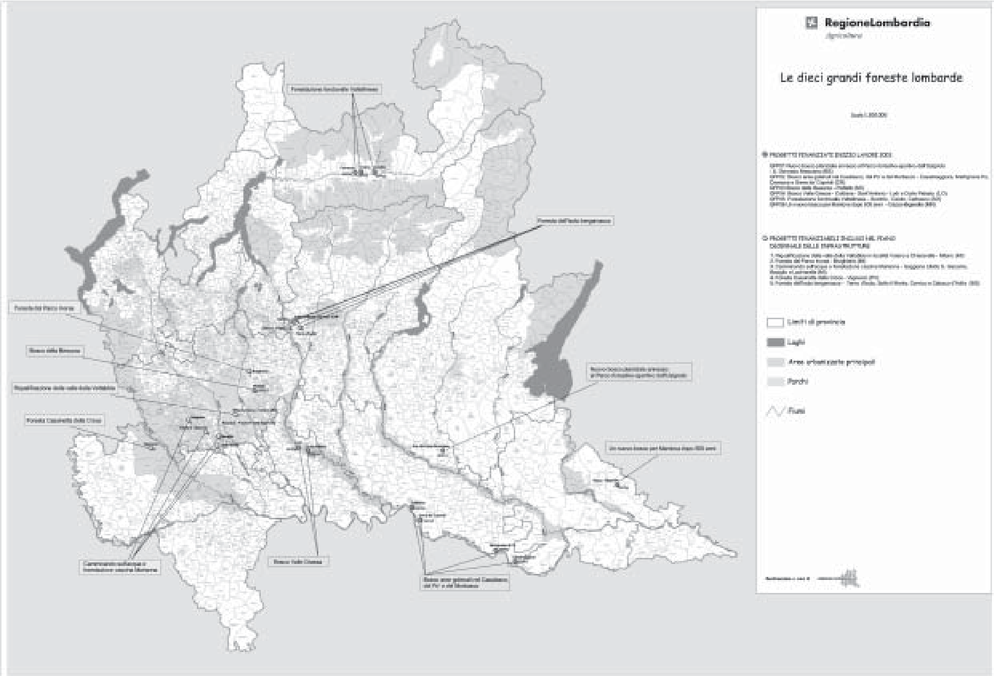

Figure 1. Map presenting the new forest project in Lombardia.

Towards new forms of environmental and rural land planning

Until the 1990s the Padan Plain had been characterized by a continuous demolition of the forests reserves both for the exploitation of agro-forestal space and for the changeover in the agricultural work organizational patterns: the landscape resulted dramatically simplified and biodiversity suffered a remarkable impoverishment. With the communitarian regulations a new process in wood plantations, vineyards and bushes as well as rural land re-naturalization was undertaken by the farmers.

Landscape ecology and the recent land planning orientations derived from the European Convention for the Landscape point out the need for land quotas to be devoted to ecosystems with various naturality degrees in order to create ecological hubs and webs for landscape re-qualification.

The General Management of Lombardia Region, moving from such considerations, assumed as strategic to promote some thousands of hectares of new green plain systems, in order to start a land qualification process. aimed at satisfying actual life and landscape quality needs.

The strategy, in order to realize this goal, has been to promote a general project known as the “10 great forests for the plain areas”: more than two years have been necessary, beginning from 2000, in order to define in concrete terms project criteria, all possible funding resources, publicity as well as consensus and feedbacks. In figure 1 are shown the new forest areas.

Assessing biodiversity to improve planning strategies: starting from a botanic classification

Since the preservation and the development of biodiversity has been taken on as a priority aim of the “Ten great forests” project, the inclusion of new forest land covers has required a study on a methodology in order to estimate the present biodiversity level in the Lombardia plain area. This methodology has been set up in order to become a further helping tool for taking decisions about environmental and land planning.

Concretely, an index – LBI, Land Biodiversity Index – has been elaborated which may assign a biodiversity value (or better, an aptitude to biodiversity) to the territory, starting from land covers types (vegetation). The map of such index, referred to the land scale, should lead to obtain a more organic and complete scenario of the biodiversity level, putting in evidence the situations of greater and smaller weakness.

Researchers do not intend to consider the elaboration of LBI index as an alternative towards the different ecological indicators and indexes, already suggested and accepted in the literature, which are regularly applied in many cases; its most suitable use here proposed is that of monitoring and previously evaluating the biodiversity in an intentionally large land scale. In a following research phase on LBI, it will be possible to make other deeper analyses. At the beginning of the research it was not available a monitoring biodiversity tool, more precise and reliable than outputs from indicators based on the reading of the extent of land area protected (for instance, the OECD core indicator, that will be seen in comparison). The set up monitoring tool will be used together with other tools by the decision-makers in order to plan the future forest projects all over the region.

The methodology used in this application is to define a biodiversity level, starting from existing and available environmental (vegetation) data at the wide land scale, without further collection of starting data, in order to reach a first likely complete and reliable outline, which may lead to consider the environmental planning directions. The final purpose is to give the possibility of better deciding where new forest areas should be realized in order to obtain better results in terms of efficacy and efficiency, above all from the point of view of increasing biodiversity (at the land scale), that is a goal of a sustainable way of planning.

Starting hypotheses for the setting up of the monitoring tool of the land biodiversity potential level.

In the literature, there are many definitions of biodiversity, starting from the one contained in the agreement on biological diversity signed in Rio de Janeiro on June 5, 1992. Just in such agreement it is encouraged a constant research of monitoring tools for the biological diversity components having some importance for its preservation and lasting use (art. 7; http://www.europa.eu.int/scadplus/leg/it/s15006.htm). Vegetation, pointed out on a land scale through the different land covers which characterize it, is one of the elements (certainly not the only one) that could both define the biodiversity level and give a useful direction for the operations of environmental and urbanistic planning. In this paper we also agree with the cultural position considering the biodiversity not an environmental theme exclusively for specialists, but as an application theme to every one dealing with enviromental planning and protection issues. Biodiversity is considered one of the “qualities contributing to the formation of the existence value of any environmental resource pertaining to the living world” (Boggia et al. 2002). This last reading key is very important for the present paper’s placing in relation to its purposes. In this perspective, the realization of new plain forest lands, keeping different environmental aims, such as the preservation and the improvement of biodiversity, also and preliminarily becomes a matter of sustainable planning aimed to improve the environment, taking care of one of the key characteristics important to quality. We must consider such contribution from this point of view: this is our first (cultural) hypothesis.

The starting point of the method is based on the geographic information data used here. They come from a geoprocessing application made from land covers, mapped on the environmental cartography of Agricultural and Forest Lands Destination and Use (DUSAF), set up by the Agricultural and Forest Development Regional Office (ERSAF) of Lombardia Region. The map is vectorial and is available on scale 1: 10 000 (updating: between 2000 and 2002) and include information on vegetation types. Such cartography, despite some inaccuracies, has been taken on just so, considering that, on the whole, it is a reliable product. This is our second hypothesis: data from DUSAF land cover map are well done for biodiversity monitoring, especially because they have similar formats all over the area considered in this study.

The third and last starting hypothesis on which the research is based, consists in the assumption of a logic consequence according to which plants biological variety also involves a certain correlation with animal biological variety. The methodology here proposed is just based on the monitoring of plants variety depending on their use and available all over the land, in spite of other kinds of flora, fauna and vegetation data, just known in specialist and deep research. At the moment, such research data are still few, broken up and carried out with different methodologies, in order to be usefully considered in a paper aimed to provide a unitary and complete outline to the scale of Lombardia Region plain.

We can say that this hypothesis, if in one way can seem excessive, in the other it has already been shown how a landscape impoverishment, and so a smaller and less various land cover, matches with a loss of vegetal and animal species wealth; the result is an impoverishment of the variety content in that land.

Therefore, it is necessary to make the planner aware to the importance, in his resolutions, of the biological variety as a tool to raise the environment value.

Identification of biodiversity levels according to land covers

The conceptual point of monitoring/assessment methodology especially lies in the previous information about the estimated biodiversity level associable to every land cover and/or its use: it is a basic and delicate step.

Analysis criteria have been chosen in order to respect the principle of analytic easiness. Five key characteristics of plants land cover have been found out, important to determine both the biological variety level of an area and the conditions for a long keeping; through them, every single cover considered in the cartography has been provided with scores.

The five key criteria to estimate the biodiversity of the different land covers are:

1. vertical structure
2. horizontal structure
3. temporal structure
4. plants composition
5. dynamic state.

Such criteria are essentially based on three conditions:

- the more diversified the way vegetation covers the land, the more chances it has to develop ecologically different patches, right for being colonized by different individuals and communities; the first three parameters related to vertical, horizontal and temporal structures follow such condition;
- the more diversified the flora forming a kind of plants cover, the more the opportunities for biodiversity; for example, a maize field, apart from being formed by man-selected varieties, presents an extremely repeated and even ground, with plants of the same size;

they blossom, fructify and are all used at the same time, so creating a clear ground evenness; the flora structure criterion is based on such conditions;
- the more permanent is a vegetal cover, the more chances of long keeping and specializing has biodiversity, creating particularly valuable ecosystem, very rare in a land such as Padana plain, where the main crops are those mostly characterized by yearly repeated rhythms; therefore, dynamic state criterion is based on such assumption.

For each criterion the analysis of cover types has been made using the following elements, estimated according to a geometric scale:

- vertical structure:
  1. monoplane vegetation, generally herbaceous plants;
  2. tendentially biplane vegetation, herbaceous and ligneous plants;
  3. multi-stratified vegetation with herbs, shrubs and trees, in different environmental conditions (air, water, land);
- horizontal structure:
  1. constant vegetation, mainly cultivated, with a planned and repeated space distribution;
  2. mosaic of spontaneous and grafting vegetation; spontaneous and uneven herbs vegetation on the land cover;
  3. mainly spontaneous ligneous and herbaceous vegetation;
- temporal structure:
  1. vegetation with a short cycle, less than one year;
  2. mixed vegetation with a short or pluriannual cycle;
  3. tendentially permanent vegetation, also spontaneously repeated;
- flora structure:
  1. vegetation formed by man-selected species;
  2. mixed vegetation with man-selected species and spontaneous species;
  3. spontaneous species vegetation;
- dynamic state:
  1. artificial man-preserved vegetation;
  2. mainly spontaneous herbaceous and/or ligneous vegetation – growing or dynamically stopped – in the first growth periods;
  3. spontaneous, mainly ligneous vegetation.

The application of these criteria to the DUSAF cartography land covers types is shown in Table 1. The cover types have been unified in four main groups related to four biodiversity levels and just so used in the following phases. It's right not to forget that also this appraisal procedure, like other ones, is clearly influenced by the scientific training, the land knowledge level and the land personal idea of the researcher using the method.

The calculation of "Land Biodiversity Index" (LBI)

The methodology here proposed takes into account the extensions and level of different vegetation land covers to obtain a final score. It has been formed a land grid by many unitary areas of 100 hectares. To every area it has been assigned a value, associated to a numerical score depending both on each different type of land cover area and on its different importance in generating a high quality level of biodiversity (see last paragraph). In this way it's possible to prepare a biodiversity score matrix organized according to the four different biodiversity levels defined before. Thanks to a final aggregation phase, through weighted average, it has

Table 1. The score matrix considering the different land covers for defining biodiversity levels.

Land cover types according to DUSAF cover land map legenda		vertical structure	horizontal structure	temporal structure	flora structure	dynamic state	Total	Biodiversity level
S4	Covered cultivations of horticultural and gardening species	1	1	1	1	1	5	Level 1
S1	Crops	1	1	1	1	1	5	
S6	No urban kitchen gardens	1	1	1	1	1	5	
S3	Open field cultivations of horticultural and gardening species	1	1	2	1	1	6	
S1c	Crops with few tree lines	1	1	1	2	1	6	
S2	Wooded crops	1	2	2	1	1	7	
S7	Rice fields	4	1	1	1	1	8	
L7	Poplar plantations	2	1	2	2	1	8	
S41	Nurseries of agrarian, forestal and ornamental trees	2	1	2	2	1	8	
S31	Open field nurseries of agrarian, forestal and ornamental trees	2	1	2	2	1	8	
P2s	Plane meadows with crops	1	1	2	2	2	8	
L8	Other agrarian tree plantations	2	2	2	2	1	9	Level 2
S1a	Crops with many tree lines	2	2	2	2	1	9	
P2a	Plane meadows with tree lines	2	1	2	2	2	9	
L1	Fruit orchards	2	1	4	2	1	10	
P2	Plane grasslands	1	1	4	2	2	10	
R2q	Quarries	2	2	2	2	2	10	
R3q	Waste places	2	2	2	2	2	10	
L2	Vineyards	2	1	4	2	1	10	
F2	Discontinuous tree lines and hedges	2	2	2	2	2	10	
L1v	Fruit orchards with vineyards	2	1	4	2	1	10	
L2f	Vineyards with fruit orchards	2	1	4	2	1	10	
P2p	Plane meadows under pastures	1	1	4	2	2	10	
P4	Meadows and pastures	1	1	4	2	2	10	
B7	Recent tree plantations	2	1	4	2	2	11	
P4a	Grasslands with isolated trees	2	1	4	2	2	11	
P1	Water meadows	4	1	4	2	1	12	
F1	Continuous tree lines and hedges	2	2	4	2	2	12	
N5g	Vegetation of artificial hight enbankments	2	4	2	2	2	12	
R5	Sandy and gravelly areas and beaches	1	4	2	4	2	13	Level 3
A3	River beds and artificial river courses	1	4	2	4	2	13	
N5	River bed vegetation	2	4	2	4	2	14	
N8	Shrub vegetation	2	4	2	4	2	14	
R1	Detritus heaps and lithoid outcrops without vegetation	1	4	4	4	2	15	
N3/N4	Detritus heaps and rocky vegetation	1	4	4	4	2	15	
B4	Conifer woods	2	4	4	2	4	16	
A2	Lakes, basins and permanent ponds	2	4	4	4	2	16	
N8b	Shrub vegetation with spots of vegetation evolving towards natural woods	2	4	2	4	4	16	

Table 1. continued.

Land cover types according to DUSAF cover land map legenda		vertical structure	horizontal structure	temporal structure	flora structure	dynamic state	Total	Biodiversity level
N1/N2	Mire and bog vegetation	4	4	4	4	2	18	Level 4
B5	Mixed woods of conifer and broad-leaved trees	4	4	4	2	4	18	
A2x	Artificial lakes, basins and permanent ponds	2	4	4	4	4	18	
B1u	Riparian shrub and wood vegetation	4	4	4	4	2	18	
B1	Wood of broad-leaved trees	4	4	4	4	4	20	

been possible to obtain an index (LBI) taking into account the various extension quotas pertaining to each bdv level (lev. 1→weight 0,25; lev. 2→weight 0,5; lev. 3→weight 0,75; lev. 4→weight 1). For the present simulation the urban areas, the infrastructures, the rubbish dumps and the quarries have been considered with a weight equal to zero, that means, for the moment, to exclude them from the assessing procedure.

The application of LBI in the Lombardia plain area: the comparison with a OECD biodiversity core indicator

The result of the LBI methodology application in the Lombardia plain area is in Figure 2. According to LBI index the biodiversity level is tendentially low, rather fragmented and however not uniform along the studied area. It is possible to recognize some areas of linear shape with a high biodiversity level: they act as benchmarks. They coincide with the main river corridors in Lombardia (Ticino, Adda, Oglio and, partially, Po rivers). It can also be noticed how the biodiversity level gets weaker passing from the territory in the west side to those in the east side: these ones seem to be the most critical areas.

According to the significance of LBI index, the comparison with the meaningful core indicator proposed by OECD to measure the level of biodiversity at the land scale is interesting. OECD indicator is good for environmental planning issues considering biodiversity a global issue. The OECD core indicator intends to score the importance of the areas protected. The output scenario deriving from map in figure 3 is definitely different from the one coming from LBI map. First of all the difference is due to the definition of OECD indicator that takes into account a de iure than a de facto situation. In fact, according to OECD core indicator, the areas highlighted in Figure 3 are characterized by a high level of biodiversity, mainly because they belong to a park; but, according to LBI index, the same areas do not maintain such environmental performance. This outlines how it is not enough to belong to a park in order to be characterized by a high level of biodiversity. It is already meaningful enough to notice how the availability of a better environmental monitoring index

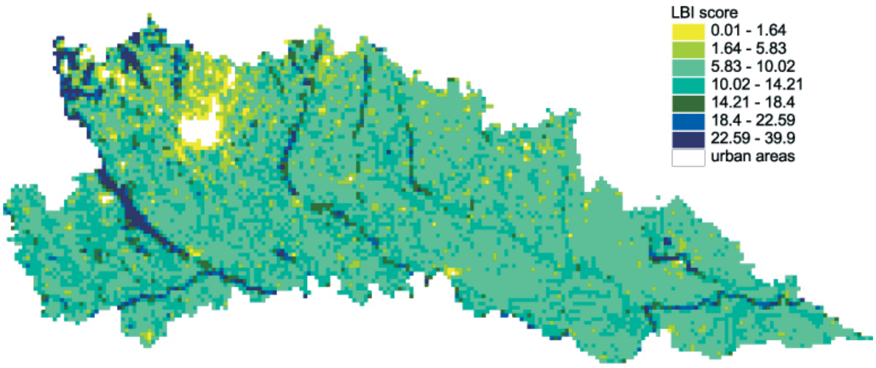

Figure 2. Map presenting LBI index in Lombardia plain area (Blue: highest biodiversity level. Yellow: lowest biodiversity level. White: urban areas)

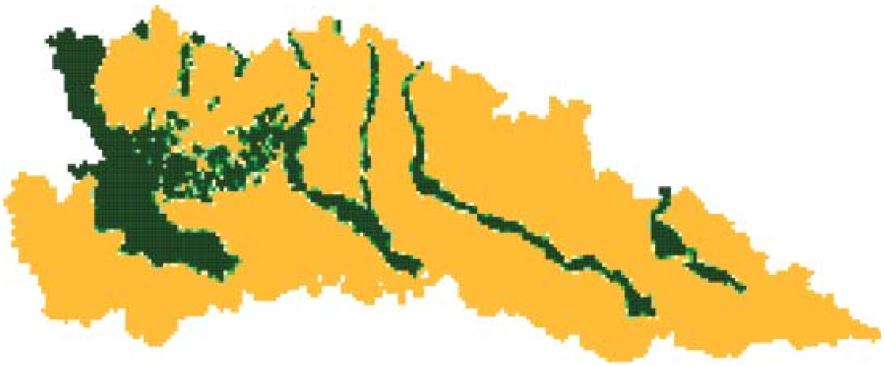

Figure 3. Map presenting biodiversity OECD core indicator in Lombardia plain area (Dark green: park areas)

could empower the decision making process for planning. For instance, the areas placed in the south of Milan urbanization do not seem to have such an important role in biodiversity level when measured by LBI and therefore they could be object of new improving enviromental interventions. Following the outcomes coming from map in figure 3, the same areas present a biodiversity level higher, not suggesting new interventions or planning issues. This opens to different planning strategies.

Conclusions, perspectives and areas of application

The methodology of assessing of the state of biodiversity at the wide land level, based on LBI index, can usefully be placed in the areas of environmental planning and of decision making disciplines rather than in the ecological ones, giving a better land biodiversity scenario according to the best sustainable practices. The strongest points of the proposed LBI methodology are the flexibility and the simple way of data calculation, the quick upgrading, the cheap procedure, the possibility to easily obtain a useful and transferable map to the land scale: everybody having a vector map and a GIS could obtain a similar result. In perspective, the basic hypothesis done for LBI calculation could be integrated with new conditions and variables important to biodiversity, but always calculable onto the whole territory under study.

In fact, there are other factors, equally important in order to define the level of biodiversity, that are not considered yet in the present paper. They will be included in a following part of the research. They are, for instance: the number of the botanic species, the importance of every single species, the presence of specific habitats, the validation of the found vegetation types directly on the open field, the shapes of the land covers, the green infrastructures. It is useful to remember that such experience comes from a forestry project in the plain area. This project needed just an environmental diagnosis tool to better address each project hypothesis, in order to obtain results compatible to the sustainable development hypothesis. The research has introduced the LBI index to get a better final outcome, in detail and contents, beginning from a geographic database usable homogeneously all over the land.

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Session 7: Human Influence on Biodiversity

Towards Biodiversity Assessment for Boreal Forests in the Pechora River Basin (Russian Federation)

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Abstract

A method for biodiversity assessment and natural resources management in the Pechora River Basin is developed, to evaluate the sustainability of land-use. Data were collected on the abiotic and the biotic system, and models are being developed on forest production, regeneration and changes in biodiversity. Species diversity differs per taxon, but also alpha and beta diversity differs, and management has large impact on the species diversity. We show that high species diversity is often associated with harvesting, due to invasion of plant and bird species in open vegetation. In other ecosystems we observe a high species diversity for disturbed areas (e.g. gardens, with ruderal species and neophytes), and low diversity for species-poor ecosystems such as peatlands or even some pristine forests.

Species diversity was found to be a poor approximation of biodiversity in the sense of ecosystem value, since different succession stages and disturbance regimes result sometimes in high species numbers, compared to undisturbed ecosystems. We propose therefore a different approach, based on a number of indicators at both landscape (ecosystem) and stand level. At landscape or ecosystem level we propose ecosystem rarity, landscape pattern, naturalness, and representativeness as indicators. At stand level we propose species diversity, species rarity and dead wood.

Introduction

The Pechora River Basin (Komi Republic and Nenets Autonomous District) is situated in the Russian Federation, just west of the Ural Mountains (Figure 1). The territory is larger than Germany and is covered with tundra in the north, and boreal forests (far northern, northern and middle taiga subzones) in the south. Parts of these forests have been harvested, but still

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Figure 1. Pechora River Basin, Russia.

large areas can be considered as pristine areas. Within this territory lies the Komi Virgin Forest Reserve, which was included in the UNESCO World's heritage sites in 1995 and forms with 3.28 million ha the largest protected taiga forest in the world (Anon. 1999; Anon. 2001).

The Pechora River is, with a length of 1809 km, one and half time as long and with a catchment basin of 288 000 km^2, twice as large as the river Rhine. The river itself is however almost in its natural state, with only one bridge crossing the river and no major river improvement works established (Van Eerden 2000). Only one railway line connects the northern industrial town of Vorkuta with the southern part of the Komi Republic, and the Russian hinterland, no roads are present in the north outside the few urban areas.

Forestry and mineral exploitation (oil, gas, minerals) are important economical activities in Komi. Several processing industries related to these are present in the region, in particular Neusiedler-Syktyvkar, one of the largest pulp and paper factory of Europe. Small scale farming activities, hunting, fishing and haymaking take place, concentrated around existing settlements and villages. Production is mainly for subsistence, since the infrastructure is very limited.

Over the past 80 years vast areas of mainly pristine forest have been harvested, with a steady increase from the forties onwards up to the 1980s of the past century, when 26 million m^3 were harvested annually (Figure 2). Low prices for timber have lead to a decrease in wood demand from this region, production being only some 5.5 million m^3 per annum at present (Kozubow and Taskaev 2000; Angelstam et al. 1995). However, with more strict conservation policies being implemented in Western Europe, it is to be expected that demand will increase, leading to more harvesting, and eventually also increased pressure on pristine forests or valuable secondary forests. In addition, the following problems are encountered in forestry:

- large scale clear-cuts in primary forest;
- unsatisfactory regeneration after clear-cut, leading to commercially uninteresting stands as secondary forests;
- loss of biodiversity;
- small share of commercial stems and large losses of commercial stems at harvest;
- limited rural development

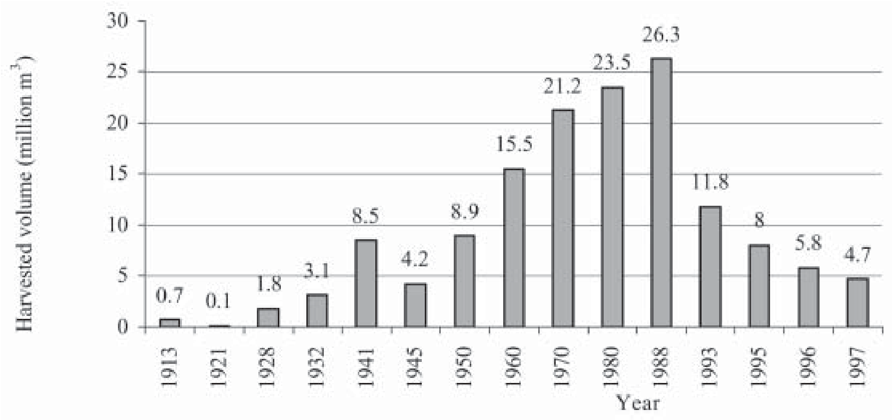

Figure 2. Harvested timber volume in the Komi Republic (Kozubow and Taskaev 2000).

A method for biodiversity assessment and natural resources management in the Pechora River Basin is necessary if a sustainable land-use is to be accomplished. Such a method is at present being developed in the framework of the PRISM project. The PRISM (Pechora River Integrated System Management) project focuses on sustainable management of natural resources. The results of the project should give indications for more sustainable forestry management in Russia, but also help in understanding natural processes in forests in Western Europe.

A first step is to collect and compile spatial data on the abiotic (soils, hydrology) and the biotic systems (flora, fauna). This information should be stored in a digitized form and be made available to planners and decision-makers. Second, models are required to develop and evaluate different development scenarios. Important input data for such models are forest structure, forestry production and biodiversity. Models currently developed in the PRISM project are the Pechora Basin Hydrological Model, the ForGra forestry model (Jorritsma et al. 1999) and a biodiversity model. Based on these models the evaluation of different strategies for forest management is possible as well as predictions on forest production, regeneration and changes in biodiversity.

This paper presents criteria and indicators and proposes a biodiversity model for boreal forest.

What is biodiversity?

The definition on biodiversity is central in any assessment method. A common indicator is the species number per area; often the number of vascular plant species. However, it is well known (e.g. Newton and Kapos 2002) that species number only is not a good indicator, since relatively high values are encountered in disturbed situations, and relatively low species numbers are found in pristine ecosystems.

'Biodiversity' is a contraction of biological diversity. Diversity is a concept which refers to the range of variation or differences among some set of entities; biological diversity thus refers to variety within the living world. It has become a widespread practice to define biodiversity in terms of genes, species and ecosystems, corresponding to the three fundamental and hierarchically related levels of biological organization (WCMC 1992; http://ceres.ca.gov/biodiv/Biodiversity/biodiv_def2.html).

Box 1. Principles for choosing indicators (CBD 1999, 2003).

On individual indicators:

Policy relevant and meaningful
Indicators should send a clear message and provide information at a level appropriate for policy and management decision making by assessing changes in the status of biodiversity (or pressures, responses, use or capacity), related to baselines and agreed policy targets if possible.

Biodiversity relevant
Indicators should address key properties of biodiversity or related issues as state, pressures, responses, use or capacity.

Scientifically sound
Indicators must be based on clearly defined, verifiable and scientifically acceptable data, which are collected using standard methods with known accuracy and precision, or based on traditional knowledge that has been validated in an appropriate way.

Broad acceptance
The power of an indicator depends on its broad acceptance. Involvement of the policy makers, and major stakeholders and experts in the development of an indicator is crucial.

Affordable monitoring
Indicators should be measurable in an accurate and affordable way and part of a sustainable monitoring system, using determinable baselines and targets for the assessment of improvements and declines.

Affordable modelling
Information on cause-effect relationships should be achievable and quantifiable, in order to link pressures, state and response indicators. These relation models enable scenario analyses and are the basis of the ecosystem approach.

Sensitive
Indicators should be sensitive to show trends and, where possible, permit distinction between human-induced and natural changes. Indicators should thus be able to detect changes in systems in time frames and on the scales that are relevant to the decisions, but also be robust so that measuring errors do not affect the interpretation. It is important to detect changes before it is too late to correct the problems being detected.

- Genetic diversity: the heritable variation within and between populations of organisms.
- Species diversity: the number of species in a site or habitat (is also called species richness)
- Ecosystem diversity: the diversity of ecosystems. Since there is no unique definition and classification of ecosystems at the global level, it is difficult to assess ecosystem diversity other than on a local or regional basis and then only largely in terms of vegetation.

Under the Convention on Biodiversity (CBD) countries are obliged to monitor biodiversity. Monitoring is important to detect ecosystem changes, or effects of e.g. specific restoration measures or air pollution.

The UNDP formulated a number of criteria, which should be met by the indicators (see Box 1, CBD 1999, 2003).

Indicators should be appropriate for use at a local scale level, but it should be possible to aggregate data to larger scale levels (FAO 2003). Also the CBD has emphasised the need to adopt the ecosystem approach in indicator development (Newton and Kapos 2002).

Table 1. Used indicators to determine conservation value of areas. Source: De Groot (1992), modified after Usher (1986) and Spellerberg (1992).

Criteria	Relative importance*	Ranking	Relative importance**	Ranking
Diversity (of species and/or habitat /only species)	12.2	1	18.1	1
Rarity (of species and/ or habitat)	11.3	2	9.2	4
Representativeness	10.2	3	2.5	11
Area size needs/minimum critical ecosystem size	9.9	4	1.3	13
Naturalness/heritage value	8.9	5	8.1	6
Scientific value	8.4	6	-	-
Ecological fragility/species vulnerability	8.3	7	2.5	11
Uniqueness/endemicity	8.0	8	-	-
Threat of human inference	8.0	9	11.2	3
Wildlife reservoir potential	7.4	10	-	-
Potential value	5.0	11	3.3	10
Management factors	4.8	12	0.7	15
Position in ecological geographical unit	4.7	13	4.0	8
Replaceability	3.8	14	13.1	2
Amenity value/aesthetic qualities	2.8	15	-	-
Record history	2.0	16	0.8	14
Education value	1.5	17	-	-
Availability	0.7	18	-	-
Special environmental conditions	-	-	0.7	15
Maturity	-	-	9.0	5
Completeness	-	-	4.5	7
Protection function for abiotic factors	-	-	0.7	15
Synecological importance	-	-	4.0	8

* The importance of values calculated are based on Margules and Usher (1986) using the Delphi-method
** Weighting in 20 analysed assessment methods for areas (biotopes), which were within impact regulation in Germany

Many aspects are in general considered important for biodiversity, or conservation value. Table 1 presents indicators generally used for biodiversity. Different combinations and weighting of criteria can be compounded to a ‘total value’ for conservation. The approach with different criteria and their associated weighting includes a large element of subjectivity, which might lead to widely differing evaluation results between different evaluators (Spellerberg 1992).

The PRISM biodiversity assessment method aims to describe and quantify impacts of certain scenarios. The indicators should be sensitive to impacts of forestry, changes in hydrology and land use, and pollution and fragmentation.

Method

The biodiversity assessment of our method is based on the number of species per ecosystem type, since this type of estimate is easily measurable and well understood (Noss 1997). Although it is difficult to cover all species, in particular in large areas such as the Pechora River Basin, using a relevé basis gives a reasonable approximation. The species-richness approach used here seems justified, considering the scale of the area, and the fact that a large number of relevés is available.

In 2002 and 2003 surveys were performed in the Pechora Basin, both in nearly pristine areas and areas where land use (mainly forestry, mining activities, fisheries, infrastructure) had a large impact on the ecosystem. A multidisciplinary expedition team collected data on different aspects of the ecosystem. Various abiotic parameters were described in a multidisciplinary approach: soil type, hydrology, geomorphology, and humus profile. Also flora and fauna were surveyed: composition of lichens, mosses, vascular plants, mammals, birds, fishes, insects and herpetofauna. Plots of 400 m^2 were sampled in different land units, for abiotic conditions, flora and insect composition. Transects of several kilometers were sampled in different land units to define composition of birds and herpetofauna.

Mammals were classified according to their size. Elk, brown bear and wolf were considered large mammals, red fox, stoat, red squirrel and arctic hare medium sized, and muskrat and water vole were considered small mammals.

In key sites seven ecological transects were laid out, their length varying up 2 up to 5 km from the river through the floodplain into the upland territory. Transects were selected on the basis of satellite images. At the selected sites in different land units along the transects, detailed information was collected on plant communities and diversity of vascular plant species at plots of 400 m^2 (in total 275 releves).

The structure of vegetation communities was described according to the different layers identified – tree stand, shrub and tree undergrowth, grass, dwarf-shrub and moss cover. In forest communities a number of parameters were described – composition of stand, crown density, height, trunk diameter, age, regeneration. The species composition and abundance of vascular plants, lichens and bryophytes were noted.

In the same locations also data on soil type, mosses, lichens and soil invertebrate diversity was collected. Lichen (3000 samples) were taken from different types of substratum – the bark of trees and shrubs, stumps, fallen trees, dead trees, soil, stones, treated timber. Mosses (500 samples) were taken mainly from the soil and dead wood. Insects were collected using generally accepted quantitative and qualitative methods of recording: excavation and manual sorting of soil samples with a volume of 0.0625 m^2, catching by Berber's soil traps and window traps, and about 3000 entomological soil-litter samples were collected. Species composition of lichens, mosses and invertebrates were stated more precisely after determination of samples in the laboratory.

In all studied areas direct observation of mammals, their tracks and droppings were recorded to obtain an impression on mammal species composition and distribution in different habitats. Mammal presence were studied along transects by foot, by car and by boat. For investigation of bird species composition so-called "territory mapping method" was used in accordance with generally accepted methods in zoogeographical research. Also here the survey routes were done on foot, by car and by boat. During surveys singing male, spotted birds (mainly territory holders) and families were counted.

Results

The α-diversity for vascular plants (i.e. the total number of species within a local habitat) was highest in the land units related to the fluvial area, i.e. meadows, willow stands and to some extent birch forests (Figure 3).

The highest macro-lichen diversity was found in the forests with and accumulated total of 123 species and 93 species in spruce forests. In willow stands 74 species were recorded, in swamps 50 species, and only 27 species on meadows.

Although not tested statistically, there seemed to be a tendency for higher vascular plant species number per relevé in undisturbed forests relative to forests affected by forestry (Table

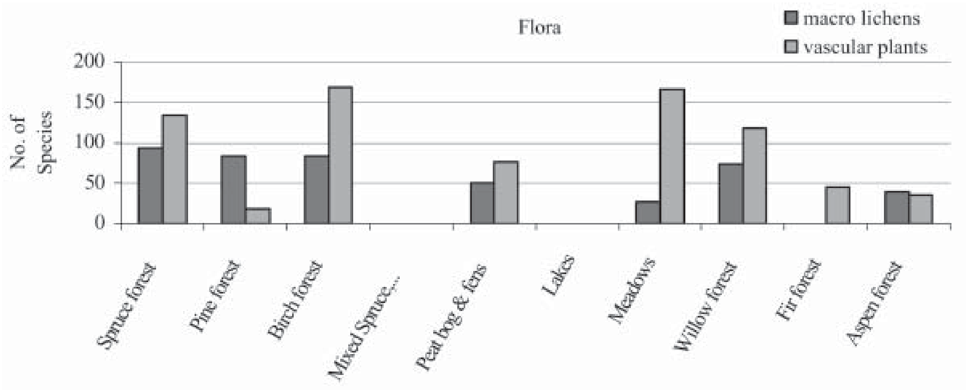

Figure 3. Number of vascular plant species and macro-lichens (total number, accumulated) as observed in 2002 fieldwork (Leummens et al. 2002).

2). For example, the average number of vascular plants, mosses and trees for relevees classified as birch and aspen forest is 48 in an undisturbed situation (n=9). Birch is mainly found as climax vegetation along streams and rivers or as pioneer vegetation after clearcut or other disturbance. In pioneer vegetations species diversity decreases to respectively 32 (n=8), 33 (n=2) and 28 (n=7). After disturbance or human impacts, e.g. floodplain meadows, farming or horticulture, the biodiversity increases, i.e. the absolute number and diversity of species might be larger (Table 2).

Two amphibian and one reptile species were observed in the study area: Common frog (*Rana temporaria*), Moorfrog (*Rana arvalis*), and Common lizard (*Lacerta vivipara*). Most observations were done on amphibians in the River valley, in particular in meadows and riverine grasslands (Figure 4).

The highest bird species diversity was observed in mixed forests, which were formed by spruce and birch (42 species) followed by spruce forest and floodplains (which includes a variety of riverine habitats such as sandbanks, gravel banks, channels and floodplains). In general, conifer forests seemed to have the highest diversity (Figure 5).

The largest number of mammals was observed in the pine forest (7 species, equally distributed over different groups), although differences were limited. The upland forests had in general higher species diversity than e.g. peat bogs, willow forest (along the river) and floodplains (Figure 6)

Discussion

Species diversity approach

Species diversity as a biodiversity indicator is usually applied in the sense that high species diversity is regarded as better and maximum species-richness is the most important management goal (Attiwell 1994, in Lindenmayer 1999). However, in our study we observed that high species richness for some species groups is associated with harvesting activities due to invasion of plant and bird species in open vegetation, also observed by e.g. Wohlgemuth et al. (2002). In other ecosystems we found high ‘biodiversity values’ for disturbed areas (e.g.

Table 2. Species diversity per relevé, (Van der Sluis in Leummens et al 2002b) ; n=148. - = no observations, n.a. = not applicable.

		Average # of vascular plant species			
		clearcut	selective cutting	otherwise disturbed	not disturbed
Upland Territory	Spruce forest	13.0	9.0	41.0	20.8
	Pine forest	10.0	-	10.0	-
	Birch & Aspen forest	19.0	13.0	16.6	34.4
	Mixed forest	-	-	9.0	20.1
	Sphagnum bog	n.a.	n.a.	17.0	9.8
	Fen	n.a.	n.a.	10.0	n.a.
River Valley	Meadow	n.a.	n.a.	36.4	n.a.
	Sandbank	n.a.	n.a.	n.a.	39.0
	Willow stand	n.a.	n.a.	n.a.	30.0

		Average # plants, trees, mosses			
		clearcut	selective cutting	otherwise disturbed	not disturbed
Upland Territory	Spruce forest	16.0	18.4	59.0	23.9
	Pine forest	24.0	-	18.5	-
	Birch & Aspen forest	32.3	33.0	28.1	48.2
	Mixed forest	-	-	23.0	36.4
	Sphagnum bog	n.a.	n.a.	31.0	20.0
	Fen	n.a.	n.a.	n.a.	16.5
River Valley	Meadow	n.a.	n.a.	41.8	n.a.
	Sandbank	n.a.	n.a.	n.a.	40.5
	Willow stand	n.a.	n.a.	n.a.	35.4

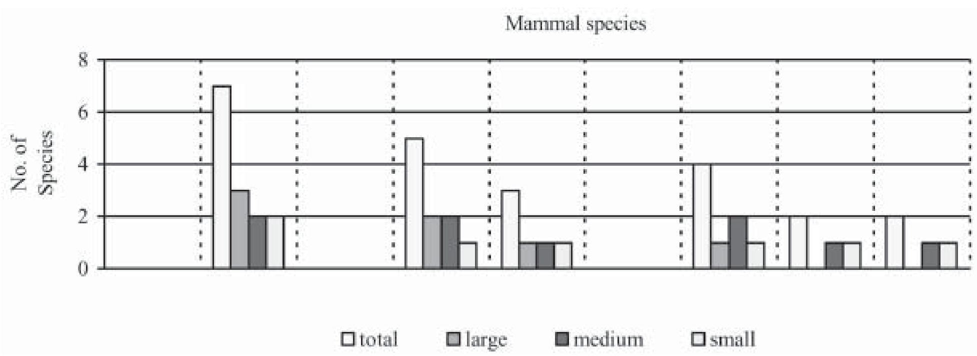

Figure 4. Number of mammals (grouped according to size) as observed in 2002 fieldwork (Leummens et al. 2002).

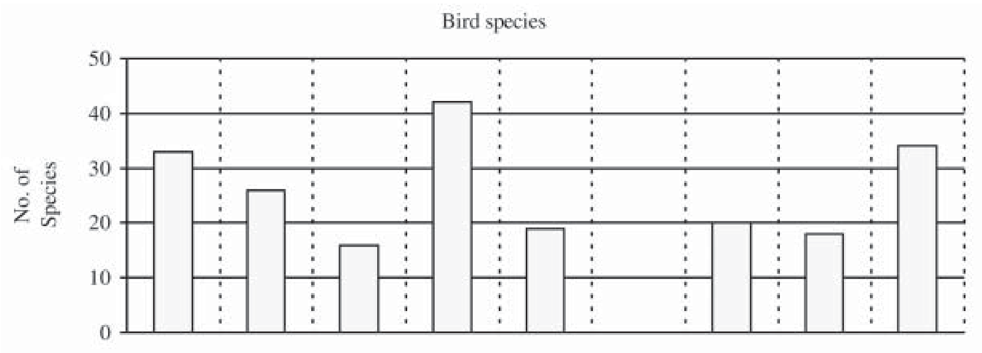

Figure 5. Bird species as observed in 2002 fieldwork (Leummens et al. 2002).

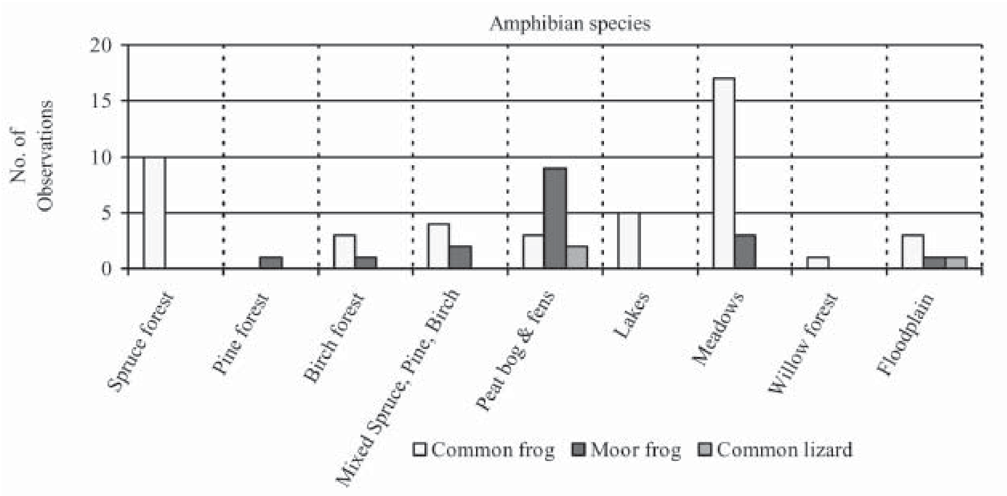

Figure 6. Observations herpetofauna, as observed in 2002 fieldwork (Leummens et al. 2002).

gardens, with ruderal species and neophytes), and low biodiversity for species-poor ecosystems such as peatlands or even some pristine forests. Species that depend on intact forest ecosystems may well disappear in such a dynamic situation, and in particular rare species might be absent despite the high species diversity (Lindenmayer 1999, Wohlgemuth et al. 2002). Species richness assessed at a scale exceeding local stand level might be reduced in these situations. It also shows that species richness is very much dependent on scale and time. Species diversity is therefore a poor approximation of biodiversity.

Finally, species diversity may mask important changes in community assemblages.

But biodiversity is not just the sum of species numbers for all taxa, we also find differences between species groups: the important habitats for lichens might differ totally from the areas with high species diversity for birds or mammals. This was also observed in other research (e.g. Jonsson and Jonsell 1999).

For vascular plants and amphibians we find a high species diversity in meadows, whereas these are rather poor for mammals and birds (Figures 3–6). So here we also see that species diversity as such is no good indicator in a diverse forest landscape with natural forest stands.

Another approach would be to use special indexes to describe species diversity. The simplest index is Simpson's diversity index, which considers both abundance (biomass), and species richness. Also Shannon-Weaver's diversity index is widely used (Huston 1994). This index uses both abundance and number of species present. Higher values are obtained in communities with many species, evenly distributed. Both methods have the disadvantage that they are more difficult to interpret. Besides, very specific and valuable aspects of biodiversity would still be lacking in this approach.

An alternative approach for biodiversity assessment

Since species diversity does not reflect very well biodiversity, it is considered important to use a wider definition of ecosystem biodiversity. To improve our assessment we have selected other indicators to be included in the biodiversity model. There is much overlap between different indicators. In addition, indicators are operational at different levels, e.g. at stand level or landscape level (Table 1).

For the PRISM project the following selection of the most relevant indicators is proposed for 'biological diversity':

- At landscape/ecosystem level
  - ecosystem rarity (e.g. number of rare or endemic species)
  - landscape pattern (minimum critical ecosystem size)
  - naturalness
  - representativeness
- At stand level, or local level
  - species diversity
  - species rarity (e.g. number of Red List, rare, protected or endemic species)
  - dead wood

Indicators at landscape level

Ecosystem rarity

Rarity or uniqueness of an ecosystem or species is an important component for biodiversity. Ecosystem uniqueness can be assessed by the mean of the endemism of various taxonomic groups. Another measure is the share of an ecosystem type in the total surface area.

Only one endemic (plant) species occurs in the area, this is therefore not of much use. There are rare ecotopes present like Mountain tundra on the Ural mountains, or specific abies forest types, which would be valued higher due to a small share in the study area.

Landscape Pattern (area size; minimum critical ecosystem size)

Each natural community or ecosystem requires a minimum amount of space, to maintain its diversity and to function properly. The size of an area therefore is of critical importance for its functioning as protected area (McArthur & Wilson 1967). Reserves that are too small can never support the full range of species that might be considered as part of the ecosystem. Besides, if the area is limited or if the carrying capacity is low, populations are too small to be sustainable (Groot Bruinderink et al. 2003).

Studies in Sweden and Finland show that species diversity might increase with the age of the forest. However, in some cases where forest fragments were less than 20 ha in size it was argued that the absence of these species might be due to fragmentation, since in similar areas in more intact landscapes specific indicator species like tree-toed woodpecker (*Picoides tridactylus*) or grey-headed woodpeckers (*Picus canus*) are present (Uliczka & Angelstam 2000).

Most common indicator for fragmentation or landscape pattern are landscape matrices, or indices, e.g. calculated by Fragstats (McGarigal & Marks 1995). However, these indices are of no value, as long as there is no proper relationship with specific species and species requirements.

Important indicator for large-scaled, intact forest ecosystems in Pechora (and therefore for the scale of the landscape) may be the Brown bear or Elk (*Alces alces*), or characteristic birds species such as Capercaillie (*Tetrao urogallus*), tree-toed woodpecker or White-tailed Eagle (*Haliaeetus albicilla*)

Naturalness

Naturalness of a site can be narrowly linked to species diversity. In general species numbers tend to increase after disturbance of virgin ecosystems, due to a different light regime and an increase in available (disturbed) habitat. We might see therefore e.g. an increase in vascular plant species, bird, insect and invertebrate species. On the other hand, some species groups clearly show a preference for undisturbed situations, in particular dead wood fauna, cryptogamic species and large mammals (Wohlgemuth et al. 2002).

The naturalness of an area depends on the degree of human presence, either in terms of physical, chemical or biological disturbance (De Groot 1992). The degree of naturalness defines the intensity of human interventions. In the Ministerial Conference on the Protection of Forests in Europe (www.mcpfe.org) naturalness has been described in three classes:

- undisturbed by man
- semi natural
- plantations

In the Pechora Basin the former two classes are prevalent. Undisturbed forests were classified in a number of international projects, but most of the forests have been cut during the past century. Plantations are rare, due to the extensive forests still presents and the relatively high costs for planting trees.

Representativeness

Representativeness does refer to the fact that a reserve should contain biota which represent the range of variation found within some land class or region (Usher 1986). The concept might have been introduced under the Man and Biosphere program, where the aim of the biosphere reserves was to represent the range of global biotic provinces.

The Yugid Va is a MAB reserve which contains forest and mountain biota, as well as undisturbed peatlands and tundra. However, for the Pechora Delta more specific biota might be selected still as specific representative area.

Indicators at stand level or local level

- Species diversity as discussed before under methods.
- Species rarity. The number of rare or endemic species is a measure of rarity. Measures of presence of rare species are the Red lists species, species that are under threat, at various levels.

Endemism might be a specific form of rarity, for species restricted to particular areas with a prescribed extent. Endemics, as mentioned, are few, but in the study area red list species occur: at least 24 plant species, 29 birds, 1 reptile and 6 mammals.

Rarity can also be based on range-size as well as density (http://www.nhm.ac.uk/ science/ projects/worldmap/ index.html).

Dead wood

In many assessments and evaluations dead wood is seen as an important indicator for biodiversity. Many species are dependent on dead wood, and its presence means therefore additional diversity in the ecosystem. Dead wood might also indicate more extensive management practices or no management at all, with associated higher biodiversity. There are many different approaches in assessment of dead wood (see e.g. Ståhl & Lämås 1996).

In the Pechora Basin dead wood is present, and very obvious, because transformation and decaying processes are very slow, due to climatic conditions. That may result in burned trees of 70 years old that are still standing, and sometimes up to 10 fallen trees per sampled area.

Biodiversity algorithm

The indicators listed above can be combined and integrated in one measure of biodiversity. Based on the available field data we can define for every relevé or sampled area and for every taxon an integrated measure for biodiversity. We propose the following algorithm for the Biodiversity Value (B_{tax}) for each taxon:

$$B_{tax} = \frac{DI + R + N + MA + Re + DW}{6}$$

in which:
DI = species diversity (e.g. α-diversity or Shannon's diversity index)
R = rarity (number of Red-List species, protected species, endemism)
N = naturalness (i.e. rate of disturbance)
MA = meeting requirements for Minimum Area size (fauna)
Re = Representativeness
DW = Dead wood

The B_{tax} can be compiled for different ecosystems on the basis of values for all relevees. For all the taxa we can then come to an assessment of biodiversity for different ecosystems, which can be scaled, e.g. from 1 to10, to make them comparable.

An integrated measure for different taxa can also be compiled to define hotspots for biodiversity within the Pechora River Basin.

Conclusions

Species diversity is a poor approximation of biodiversity in the sense of ecosystem value. In particular comparing situations of climax vegetation with disturbed situations, the high species number in the latter would indicate higher biodiversity, which is not realistic. More appropriate indicators of biodiversity should then be selected to come to a sustainability-related assessment of biodiversity. In this paper we propose a number of indicators that are considered important in the framework of the PRISM project, these indicators are based on other studies and biodiversity assessments.

The indicators described here are relevant for Pechora, however, it should be tested still with data for the entire region.

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Agroforestry Systems of Mt Etna, Italy: Biodiversity Analysis at Landscape, Stand and Specific Level

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Abstract

Due to its geomorphological, social and historical characteristics, Sicily shows widespread agroforestry systems (Cullotta et al. 2000). In their possible combinations, silvopastoral, agrisilvicultural, agrosilvopastoral and others (Nair 1985), these systems are the traditional ones of the hilly and piedmont mixed arboriculture where, especially due to geomorphological roughness, intensive monoculture systems are not suitable.

On the piedmont belt of Mt Etna, between the lower crop fields and the upper forest belt, these combined systems are well represented and characterise the environment. In order to typify their structure and composition, a biodiversity analysis was carried out at different level (landscape, stand and specific). Biodiversity indicators were applied on agroforestry systems and then compared to the features of agricultural and forestry ones, sampled along an altitudinal transect.

At landscape scale indicators of composition (number of patches, patch density, mean patch size and median patch size) and shape (edge density, mean shape index and mean patch fractal dimension) were applied. At stand level the biodiversity was analysed through structural indicators (number of plants, arboreous, shrubby, viny and herbaceous cover, trees clustering, vertical vegetation layering), type, frequency and extension of microsites and species richness indicators (number of arboreous, shrubby and herbaceous species, % of native/non native species).

According to their high contribution to the biodiversity, it would be indispensable to preserve agroforestry systems, for the conservation of environmental resources and the maintenance of traditional low-input cultural systems and of a unique landscape.

Keywords: patch diversity; community diversity; species diversity; management system; multipurpose system; Sicily.

Introduction

An agroforestry system represents an interface between agriculture and forestry. It counts mixed land use practices and it combines productive, social, economic and environmental functions. In so doing, the simplest agroforestry system is always more complex, ecologically (structurally and functionally) and economically, than a monocropping system (Lundgren et Raintree 1982).

The agroforestry concept was developed in tropical regions (Nair 1993) but, with different structures and reasons, both environmental and socio-economic, also in temperate zones agroforestry systems are historically present in their different combinations (Gordon and Newman 1997; Gordon et al. 1997; Rule et al. 1994).

In Mediterranean regions are evident too: vast areas of olive (*Olea europaea*) plantations and vineyards are intercropped, a practice to date back to Roman times or before (Lelle and Gold 1994); the *Dehesa* of the Iberian peninsula is still the largest agroforestry system in Europe today; a wide range of grazing in forests and mixed fruit orchards can be still observed (Dupraz and Newman 1997). However, Mediterranean agroforestry systems are simpler, structurally and functionally, and present a lower richness of species than tropical ones, despite their larger extension. They are returning to an earlier and/or less advanced form, but they may be revaluated (Paris and Cannata 1991; Bertolotto et al. 1995).

Nevertheless, intensification, specialization and mechanization of agriculture ultimately became key factors in the elimination of trees from cultivated fields (Dupraz and Newman 1997) and humane disturbance can be seen as the major threat to local biodiversity, such as modern agriculture. The increasingly intensified forms of land use have had a wide range of effects on biotic and landscape diversity in terms of ecological oversimplification that results in a decline in biodiversity (Swift et al. 1997). Agroforestry, such as extensive cultural system, can be seen also as an integrated approaches to biodiversity conservation on farms in nature reserves (Dobson et al. 1997).

In Italy, as well as in all the Mediterranean Basin, old-traditional agroforestry systems are widespread. In particular, this is the case of extensive policultures of the traditional fruit trees system of Italy, the oldest one of Europe, characterised by a large use of local genetic (specific and intra-specific) resources (Barbera et al. in press).

Due to its geomorphological, social and historical characteristics, Sicily, located in the centre of Mediterranean regions, shows still widespread agroforestry systems (Cullotta et al. 2000) in all of their possible combinations (silvopastoral, agrisilvicultural, agrosilvopastoral, others) *sensu* Nair (1985). On the piedmont belt of Mt Etna (Sicily) these combined systems are well represented.

In order to typify them on their structural and compositional characters a biodiversity analysis was carried out at different level (specific, stand and landscape), as well as the term of biodiversity implies vary aspects of the complexity of the ecosystems: from the number of species both plants and animals to the variability of the landscape.

An analysis of some of these systems, sampled along an altitudinal transect, made it possible to show how agroforestry systems present higher heterogeneity comparing to specialized crops at landscape, stand and specific level that is reflected in a more complex and rich ecosystem structure and in a poli-functionality (productive, protective, naturalistic, recreative, faunistic, landscape) of land use types.

Materials and methods

The study area is located in the North-West of Mt Etna (Italy) and it is situated between 900 and 1150 meters above sea level (Figure 1). It is part of the Mt Etna Regional Park. The geo-

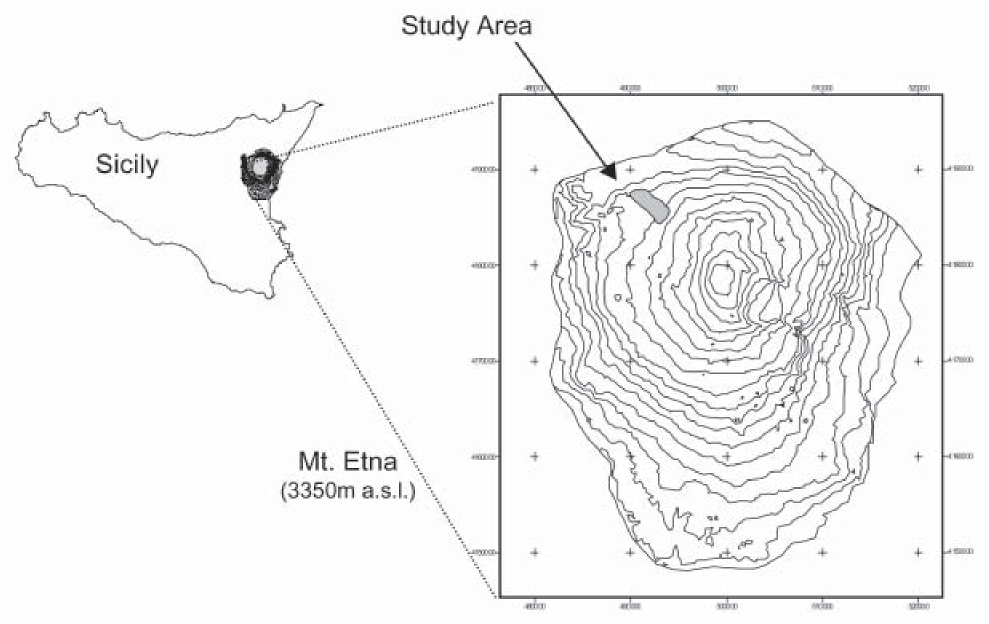

Figure1. Localization of the study area on the Mt. Etna.

morfology is characterized by alternating flat and sloping zones in which dry stone terraces have been built for agricultural purposes. They are typical of the traditional agricultural land use of Mt Etna, together with dry stone walls. The lithological matrix is constituted by basalt and soils are fertile volcanic andisols. The bio-climatic context is supra-mediterranean with temperate dry summer and humid winter (Cullotta 2003).

The area was almost completely covered by forest in the past, but natural ecosystems have been partially replaced by different cultural systems, altered and enriched by the introduction of many non-native and native woody species. Currently, remnant patches of forest, various traditional cultural systems (fragmented farms, mainly for family consumption production), specialized farms with intensive cultivations (hazelnut, apple, olive, vineyard, etc.), pastoral systems (structurally and functionally *dehesas* like), and agrisilvicultural systems are present.

In order to delimit the various cultural systems and to study biodiversity at landscape level, a preliminary phase of black-and-white orthophotos interpretation dated 2000 was carried out. An altitudinal transect was bordered, including lower crop fields and the upper forest belt. The transect extends 423 hectares and it is constituted by a total number of 163 patches. Among 40 different types of agricultural systems, 28 of them have been randomly selected as sampling patches for the evaluation of the main vegetation structures and the specific composition of cultural types and for the study of biodiversity at stand and specific level. All together the 40 types of agricultural systems have been classified according to 8 *cultural systems*: monoculture (M.), policulture (P.), cultivated agroforestry (C.A.), uncultivated agroforestry (U.A), forestry (F.), silvopastoral (S.P.), agropastoral (A.P.), agrosilvopastoral (A.S.P.) (tab.1). Cultural systems have been identified according to structural, compositional and functional characteristics of main land use types.

In order to apply biodiversity indicators data were collected from sampling patches during 2003 using a sampling table derived from Cullotta et al. (2000). The sampling table reports information about specific cultural systems, cultural types (vineyard, olive, oak wood, etc.), topographical and dimensional patch features (slope, exposure, altitude, perimeter and area of patch), structural complexity (species, number of plants, forest and cultivated trees distribution, shrubs, uncultivated herbs and regeneration clustering, vegetation layering), frequency, extension and/or distribution of specific microsites (rocks outcrop, heaps of stones, heaps of branches, dead wood, litter, dry stone walls, dry stone terraces, fence for animals, pastoral infrastructures, farms) and woody and herbaceous component functions (productive, protective, naturalistic, recreative, faunistic, landscape) (tab.1). Moreover, some of these data (perimeter and area of patches) have been collected indirectly using the GIS ArcView 3.2. Data have been allowed to achieve not only a land use map performed by the Gis in which different themes were distingued, but also to calculate different biodiversity indicators.

Cultural systems have been analysed at different level (specific, stand and landscape level) and different indicators were applied according to the analysis level. Moreover, biodiversity indicators were applied on agroforestry systems and then compared to the features of agricultural and forestry ones.

At landscape level, indicators of composition and shape were applied (Rutledge 2003), in particular to quantify patch complexity which can be important for different ecological processes (Forman 1995). Different indexes have been used to quantify fragmentation and patch complexity for such of these type of composition and shape landscape indicators: number of patches, patch density, mean patch size and median patch size for the first one, edge density, mean shape index and mean patch fractal dimension for the second one.

At stand level the biodiversity was analysed through compositional and structural indicators (number of plants, trees, shrubby, viny and herbaceous cover, trees clustering, vertical vegetation layering, frequency and extension of microsites) and species richness indicators (number of trees, shrubby and herbaceous species, % native/non native species). These indicators were measured on 28 randomly selected cultural type patches.

Results and discussion

Field and derived data collected allowed the realization of the land use map in which eight cultural systems and forty cultural types were distinguished.

The mapping and GIS calculations show that forestry systems, covering a surface area of 43.47% of the total area, are more extensive than pastoral, agroforestry and cultivated systems (Table 2). Forestry systems are scattered mostly in the northern part of the transect (upper altitude belt) and cultivated systems downward. This consideration underlines differences between natural vegetation and cultivated systems in which diversity is mostly due to ecological factors under human influences.

Main indicators of landscape composition and shape show how man influenced the fragmentation and the ecological interactions in cultural systems (Table 2). Indicators of fragmentation/composition indicate that a greater number of patches is present in cultivated systems (monoculture, policulture and cultivated agroforestry) than others.

Agropastoral systems show the lowest values whereas forestry ones have similar values as uncultivated and silvopastoral systems. Number of patches provides an incomplete picture on fragmentation giving absolute information, so the patch density [the number of patches within a given area (100 ha)] can be used easily to compare different cultural systems. High values in patch density of agricultural systems distinguish clearly monocultural systems (patch density 202.40) from each others. Moreover, the large range of patch density indicates great differences between cultural systems (lowest value in forestry systems: 10.88). Policultural

Table 1. Synthesis of main information collected by the field sampling table: cultural systems, cultural types, patch topographical and dimensional features, structural complexity and prevalent function of patches.

Cultural systems	Cultural types	Patch topographical features	Patch dimensional features	Structural complexity			System functions
Monoculture Policulture Cultivated Agroforestry Uncultivated Agroforestry Forestry Silvopastoral Agropastoral Agrosilvopastoral	(vineyard, olive, oak wood, etc.)	slope exposure altitude	perimeter of patch area of patch	number of species and plants mean cover vegetation layering	regeneration clustering plant distribution	Types, frequency, extension and/or distribution of specific microsites	(productive, protective, naturalistic, recreative, faunistic, landscape)

Table 2. Indicators of landscape composition and shape applied to cultural systems.

Cultural System	(1) Total Analyzed Area (ha)	Cultural System Area (ha)	% of total area	Number of Patches	Patch density	Mean Patch Size (ha)	Median Patch Size (ha)	(2) Sum of Perimeter of All Patches (m)	(2/1) Edge Density (m/ha)
Monoculture	423.0	13.8	3.3	28.0	202.4	0.5	0.3	8418.1	19.9
Policulture		15.1	3.6	21.0	139.1	0.7	0.4	8063.8	19.1
Cultivated Agroforestry		22.9	5.4	25.0	109.0	0.9	0.6	11426.6	27.0
Uncultivated Agroforestry		27.0	6.4	19.0	70.3	1.4	1.0	12185.3	28.8
Forestry		183.8	43.5	20.0	10.9	9.2	1.6	33146.2	78.4
Silvopastoral		44.1	10.4	19.0	43.1	2.3	2.2	14597.7	34.5
Agropastoral		77.6	18.3	14.0	18.1	5.5	1.5	17429.0	41.2
Agrosilvopastoral		38.6	9.1	17.0	44.1	2.3	1.1	13519.4	32.0

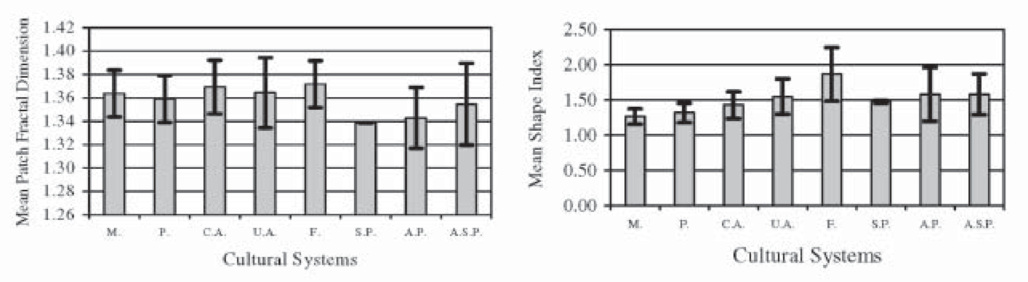

Figure 2. Mean patch fractal dimension and mean shape index of cultural systems: monoculture (M.), policulture (P.), cultivated agroforestry (C.A.), uncultivated agroforestry (U.A), forestry (F.), silvopastoral (S.P.), agropastoral (A.P.), agrosilvopastoral (A.S.P.).

and agroforestry (cultivated and uncultivated) show intermediate values between monoculture, forestry and pastoral systems.

However, fragmentation can be also represented by patch size. Mean patch size reflects up an inverse trend comparing to patch density. So that, highest and lowest values of mean patch size are respectively in forestry (9.19 ha) and in monocultural (0.49) systems. Moreover, mean and median patch size do not coincide as well as empirical and theoretical distribution do not overlap and high differences between these value indicate large range of size variability in each forestry and agropastoral systems.

According to patch density and mean patch size values, it seems that at landscape level cultivated and agroforestry systems show higher biodiversity and lower homogeneity than silvopastoral, agropastoral, agrosilvopastoral and forestry systems. Moreover, last systems show shape values higher than other systems that could reflect a higher level of edge species niches as well as perimeter concept is closely connected to specific biodiversity. In this point of view, higher values of patch density indicate larger probability to find plants and animals species (Del Favero 2001). So, the agricultural practices of land use types had a wide range of effects that results in an increase of biodiversity at landscape level.

Shape complexity is better explained by mean shape index and mean patch fractal dimension and higher values of these indexes, ranged between 1 and 2, indicate a higher patch shape complexity and irregularity of agroforestry and forestry systems than others (Figure 2). Again, agroforestry systems, together with forestry, can be considered an intermediate term between cultivated and uncultivated systems. Excluding silvopastoral, the other systems show high values of standard deviation of mean patch fractal dimension and of mean shape index.

Biodiversity analysis at stand level indicates that policulture, cultivated agroforestry and agrosilvopastoral systems display a high number and higher mean height of vegetation layers (Figure 3).

In this elaboration, viny layer was distinguished from each others because, although it is a woody species, it is planted on the border or in rows in cultivated fields or associated with trees dispersed within fields, it grows on trellises and its height is intermediate between dominated and shrubby layer.

Human action generated an increase in vegetation layers in these systems. On the other side, in forestry ecosystems past silvicultural treatments have been influenced the actual vertical structural diversity in terms of multilayering reduction, such as in pastoral practices, in monocultures and agropastoral systems. Consequently, the cover of prevailing vegetation

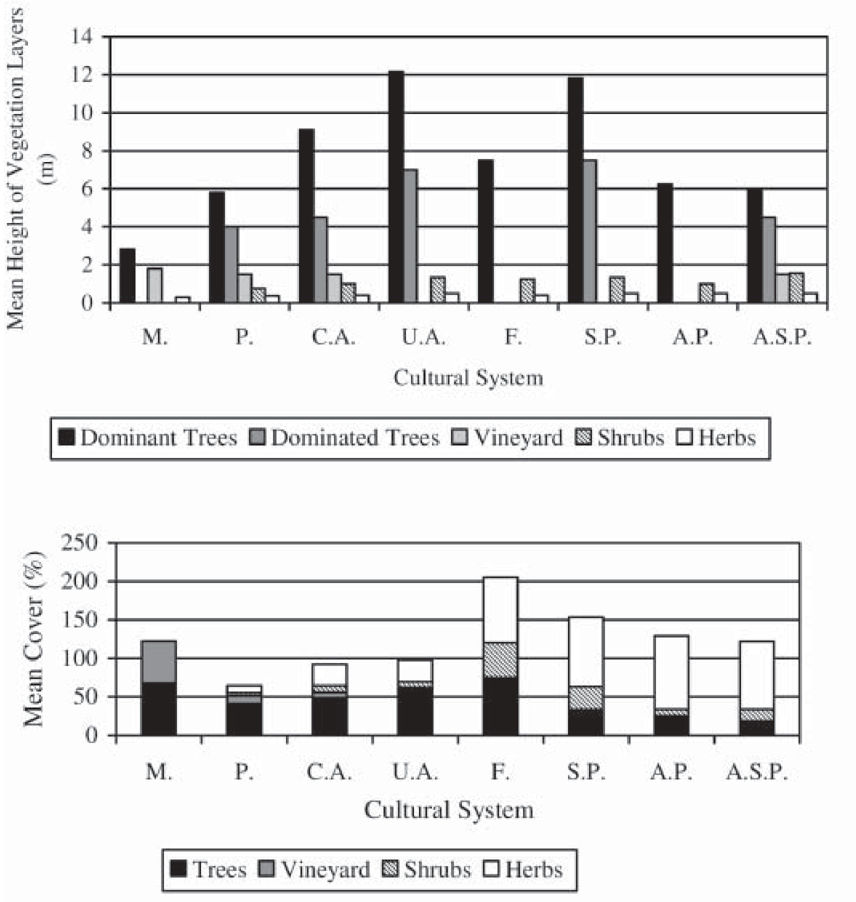

Figure 3. Mean height of vegetation layers and mean cover of cultural systems: monoculture (M.), policulture (P.), cultivated agroforestry (C.A.), uncultivated agroforestry (U.A), forestry (F.), silvopastoral (S.P.), agropastoral (A.P.), agrosilvopastoral (A.S.P.).

layer differs according to the system: trees in agriculture, forestry and agroforestry systems; herbaceous in silvopastoral ones (Figure 3). The shrubby layer increases its cover degree towards forestry and agroforestry systems, while it is completely absent in monocultures. Adding up the cover of different layers, percentage of 100% is exceeded in every systems, excluding policultures and some cultivated and uncultivated agroforestry systems. In monoculture mean cover exceed 100% due to the presence of planted dense tree species or vines: viny cover is comparable to the tree one.

As vegetation cover and vegetation layers, specific microsites influence biodiversity at stand level. Agroforestry and pastoral systems display a higher mean frequency of microsites (dry stone walls, rock outcrops, heaps of stones, etc.) than other systems and some microsites are peculiar of specific cultural systems: heaps of branches in agroforestry and agropastoral; rural infrastructures and fences for animals in pastorals; dead woods in agropastorals (Table 3). Every systems show rocks outcrop due to typical characteristics of Mt Etna rocky substrata.

In order to quantify microsite extension in each patch and to compare their values among systems, same type of microsites have been measured in terms of mean microsite perimeter/

Table 3. Mean frequence of microsites in cultural systems.

		Types of microsites									
		Rocks Outcrop	Heaps of Stones	Dry Stone Walls	Litter	Heap of branches	Dead Wood	Dry Stone Terraces	Country Houses	Pastoral infrastructures	Fence for Animals
Cultural system	Monoculture	0.08	-	0.42	-	-	-	-	-	-	-
	Policulture	0.22	-	0.88	-	-	-	0.22	0.44	-	-
	Cultivated agroforestry	0.10	0.10	0.49	-	2.44	-	0.20	0.39	-	-
	Uncultivated agroforestry	0.21	0.07	0.21	0.14	0.71	-	0.14	0.07	-	-
	Forestry	0.52	-	1.04	1.04	-	-	-	-	-	-
	Silvopastoral	0.09	0.05	0.14	0.14	-	-	0.14	-	0.05	0.05
	Agropastoral	0.21	0.21	0.42	-	0.21	1.25	0.21	0.21	0.62	-
	Agrosilvopastoral	0.12	0.12	0.17	-	-	-	-	0.04	-	-

mean patch perimeter ratio or mean microsite area/mean patch area ratio (Figure 4). Dry stone walls and dry stone terraces, quantified in terms of linear extension, are more widespread in policultures and agroforestry systems in which they exceeded patch perimeter of corresponding cultural systems. Their lowest and highest values have been measured in monocultures, probably due to intensification, specialization and mechanization of agriculture, and uncultivated agroforestry systems, respectively. Dry stone terraces are absent in forestry and agrosilvopastoral systems. Among litter, rocks outcrop, heaps of stones and heaps of branches, quantified in terms of area extension, only the first one exceeds 25% of corresponding cultural systems area in uncultivated agroforestry, forestry and silvopastoral systems. Both microsites and cultivation practices seem to have great influence in regeneration clustering. In fact, renovation and plant seedling of forest species is mainly distributed along patch edges and dry stone walls and plant seedlings is absent in monocultures. In this point of view, microsites may be seen as regeneration niches that provide microclimatic and microenvironmental benefits in regeneration mechanisms (Grubb 1977). Moreover, according to Whittaker and Woods (1981), in forest ecosystems microsites promote seedling germination and satisfy major needs of young forest species. Cultivation practices also influenced the number of plants in cultural systems; cultivated systems show a higher plant density, especially due to the presence of cultivated agricultural species (Figure 5).

Biodiversity analysis at specific level was restricted mainly to woody species (tree, shrub and vine) for the following reason: human impact on woody species was much more severe than on herbaceous native species and biodiversity variability may be appreciate easily. A number of 26 species have been recordered during field work among which 17 woody and shrubby species and only 9 herbaceous cultivated and uncultivated species. Moreover, native species amount to 50% of total number, such as non native (cultivated) species. Due to human impact, number of species (cultivated species) is clearly higher in policulture and agroforestry than others systems (Figure 5), while wild species predominate in pastorals and forestry. A general positive correlation exits between number of species and plant density in all cultural systems excluding monocultures.

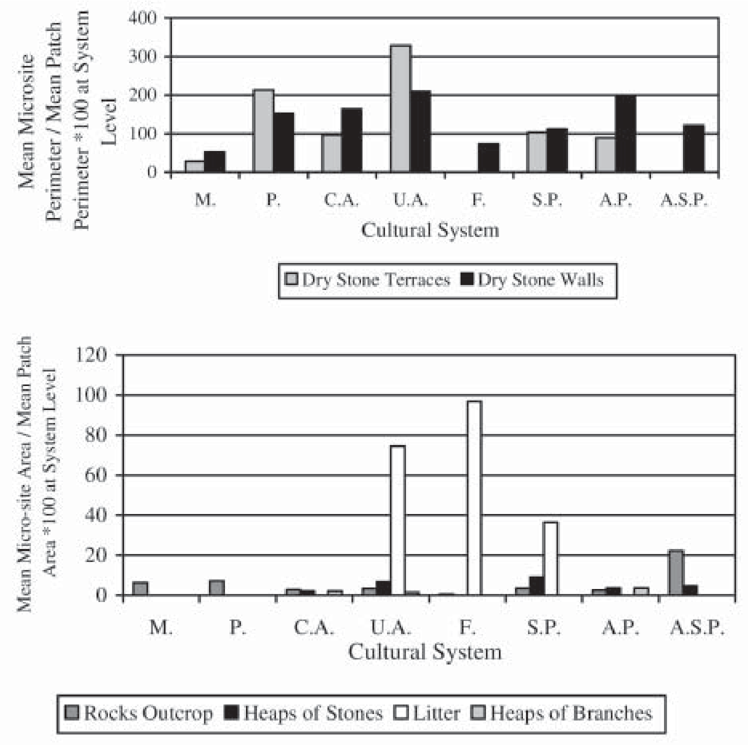

Figure 4. Mean microsite perimeter/mean patch perimeter ratio and mean microsite area/mean patch area ratio among cultural systems: monoculture (M.), policulture (P.), cultivated agroforestry (C.A.), uncultivated agroforestry (U.A), forestry (F.), silvopastoral (S.P.), agropastoral (A.P.), agrosilvopastoral (A.S.P.).

Species richness, that is assumed as the number of species in sampled patches, their evenness and the other examined indexes are important to quantify the oversemplification and the loss of biodiversity not only at specific level but also at stand and landscape one. According to Meyer and Tuner (1994) and Shift et al. (1997), in intensive crops the reduction of species evenness and richness, vegetation cover degree and multilayering, frequence of micro-sites and more generally biodiversity could be due to the modern agro-industrial forms of agricultural development that leads to a progressive simplification and standardization of cultural systems.

Vice versa, policulture and agroforestry systems contain a great number of species and a more complex vegetation structure reflecting positive influences on ecosystem self-regolation, resistance and resilience against ecological and anthropogenic disturbs.

In this point of view, agroforestry systems and their ecological functional group diversity [as defined by Hobbs et al. (1993) they are a set of species with similar impacts on ecosystem processes; for instance, arboreous, shrubby and herbaceous plants and vineyards may be classified as four different ecological functional groups that are related in different way to the sustainability and conservation of nature] may be considered a way to biodiversity conservation and to the protection of ecosystems (Huang et al. 2002).

Recent ecological studies suggest that even the only remnant trees may play an important role in conserving biodiversity within agricultural systems because they provide habitats and

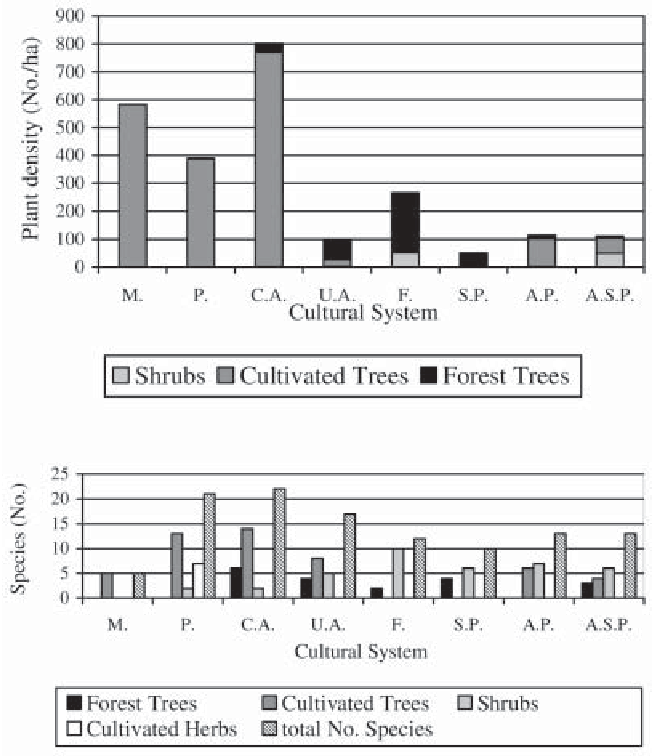

Figure 5. Species number and plant density in cultural systems: monoculture (M.), policulture (P.), cultivated agroforestry (C.A.), uncultivated agroforestry (U.A), forestry (F.), silvopastoral (S.P.), agropastoral (A.P.), agrosilvopastoral (A.S.P.).

resources that are scarse in the agricultural landscape (Harvey and Haber 1999). At the same way, remnant trees in pastoral systems, that are similar structurally and functionally to *dehesas*, contribute to maintain timber, forage, fruits and watershed protection for animals (Hietz-Seifert et al. 1996). Moreover, under tree canopy soil nutrient accumulation is guarantee due to decay of tree litter and animal accumulation (Gordon and Newman 1997).

These considerations are reflected in the multifunctionality of mixed systems. Data obtained in the analysed area clearly show the higher number of functions carried out by policulture and agroforestry systems in comparison with others.

Conclusions

The higher heterogeneity in patch shape and border of agroforestry and forestry systems is reflected in a more complex and rich ecosystem structure as well as higher number and extension of microsites (dry stone walls, dry stone terraces, etc.). These ecosystems elements are positively correlated with seedling and regeneration processes of wild species, increasing structural complexity and biodiversity level.

In agroforestry and policultural systems the rise of species number is firstly correlated to plant introduction by farmers, specially by cultivated tree species and varieties, and secondly to the irregularity and complexity of patch shape. An higher irregularity in patch shape is often reflected in the positioning of patch perimeter according to natural borders (natural escarpments, rocky lines, small brook lines, etc.). As a matter of fact, this is positively correlated to an high level of structural complexity and richness in microsites and to the sinergy due to the presence of both interior and edge species.

The multifunctionality (productive, protective, naturalistic, recreative, faunistic, landscape) of land use types increases with the complexity of stand structures and biodiversity level. Results exhibit as agroforestry/forestry systems and policultures carry out not only functions at stand level (e.g. productive) but also landscape, naturalistic, recreative and protective functions. In this way, particularly high is the importance of these systems specially in landscapes characterized by monocultural matrix and high anthropogenic level, where agroforestry and policultural systems become unique elements of a greater ecological complexity, elements of species conservation and resource exchange systems.

In view of these first results and considerations, a spreading out of the study area and the analysis of new units of similar cultural landscapes need, in order to better know these systems in a systemic way and to create as complete as possible a specific inventory. It is indispensable to preserve agroforestry and policultural systems, in view of their high contribution to the biodiversity and for the conservation of environmental resources and the maintenance of traditional low-input human made systems and of a unique Mediterranean cultural landscape.

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Effects of Management and Restoration on Forest Biodiversity: An Experimental Approach

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Abstract

To conserve biodiversity of boreal forests, many management and restoration actions have been undertaken during the last decades. However, research based knowledge about the impacts of the currently used practices are yet largely unavailable. Here we introduce two large scale experiments on the impacts of forest management and restoration on biodiversity of boreal forests. In the first experiment we concentrate on the effects of prescribed burning on species richness of two species groups, which are known to be largely dependent on the amount and quality of decaying wood: saproxylic beetles and polypore fungi. In the second experiment, we study the immediate surroundings of small forest brooks, which are defined as valuable habitats in the Finnish Forest Act. Our aim is to determine the effect of silvicultural practices conducted at the valuable habitats or in the vicinity of them on long term persistence of the special characteristics of these habitats. Our preliminary results indicate that preserving the characteristic features of valuable brook habitats will help to preserve the overall species richness of boreal forests.

Keywords: Boreal forest; prescribed burning; restoration; species richness; valuable habitat.

Introduction

Background

Conservation and economical utilization of natural resources forms a confrontation between the two, in which promotion of one unavoidably compromises the other. Last few decades have shown that excessive unsustainable utilization of natural resources causes extinctions and impoverishment of biodiversity (Meffe and Carroll 1997; Hanski 2000; Hanski and

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Ovaskainen 2002; Benton et al. 2003; Fahrig 2003). To conserve biodiversity, some practical actions have been undertaken, but research based knowledge about the impacts of the currently used management and restoration practices are still largely unavailable (Meffe and Carroll 1997; Noss 1999; Simberloff 1999; Yoccoz et al. 2001). Nevertheless, the best guarantee for successful sustainable management, restoration practises and conservation of biodiversity is to have scientific information based on controlled and replicated experimental research (Simberloff 1999).

Our research is based on controlled experiments comparing results in replicated treatments. We concentrate on two major issues of conservation of forest biodiversity. First, we study the effects of prescribed burning on species richness and the role of the amount and quality of decaying wood in preserving high species diversity. In these experiments, we study the species richness and abundance of beetles and polyporal fungi. Second, we study the importance of so called valuable habitats (see below) in conserving forest biodiversity and the effects of some silvicultural practices on the survival of species richness of these habitats. In these experiments, we study the species richness and abundance of vascular plants, mosses, polyporal fungi and polypore dwelling beetles. All our studies are conducted in the boreal coniferous forest zone in Finland.

The role of forest fire in conserving biodiversity

Forest fires are one of the most important disturbances of unmanaged boreal coniferous forests (Zackrisson 1977; Granström 1993; Zackrisson et al. 1996). Natural forest fires are effective in remoulding the age structure of the trees, increasing the proportion of deciduous trees, particularly birch and aspen (Wikars 1992), and they have an effect on soil decomposing and nutrient release (Wein 1983; Granström and Schimmel 1993; Zackrisson et al. 1996; Dahlberg et al. 2001). All of these effects contribute to the diversity of species the forests are able to maintain. During the last thousand years, boreal forests have burnt on average every 50 to 200 years (Zackrisson 1977; Engelmark 1984; Larsen and MacDonald 1998). However, during the 20th century, natural forest fires became rare due to effective fire suppression. Because of this, the diversity of species typically associated with burnt forests has decreased. Moreover, several forest species are now considered threatened or already extinct or they are likely to be extinct after a time lag (e.g. Ehnström et al. 1993, 1995; Höier 1995; Hanski 2000; Rassi et al. 2001; Hanski and Ovaskainen 2002).

Approximately 40 species of insects in Northern Europe are considered to be dependent on forest fires (Heliövaara and Väisänen 1984; Lundberg 1984; Wikars 1997). Majority of these species are beetles (*Coleoptera*), but there are also some true bugs (*Heteroptera*), flies (*Diptera*) and moths (*Lepidoptera*). Most of these species require burnt or decaying wood for their survival. They are mainly fungivores (feeding on fungi growing on decaying wood), saproxyls (feeding on decaying wood) or predators (Muona and Rutanen 1994). Typically these species survive in the area only few years after the fire (Wikars 1992). In addition to fire dependent species, there are several species that benefit from the warm microclimate and are most abundant in recently burnt areas. These species are not specialised to burned areas but survive best in warm, sun-exposed areas provided that suitable decaying wood is available.

To conserve viable populations of fire dependent species, we must ensure a continuum of burnt areas in time but also in space. Time and space interval for adequate fire continuum is not known, but it is likely that forest fires should occur more frequently than is the case today. Because natural forest fires seldom occur in Finland anymore, the distribution of fire dependent species has become highly fragmented, and is almost entirely dependent on silvicultural burnings and conservational restoration practices. To ensure the long term survival of fire dependent species in managed forests and thus maintain forest biodiversity, it is vital to determine the frequency of burning that is necessary to achieve this goal. To determine the adequate frequency of managemental burnings, information is needed about the colonisation and long term succession patterns as well as life history of fire dependent species.

The role of decaying wood in conserving biodiversity

Availability and diversity of resources are factors that strongly limit species richness and biodiversity (Begon et al. 1990; Meffe and Carroll 1997). In boreal forests, the limiting resource for a large variety of species representing several taxa is the availability of decaying wood (Kaila et al. 1994, 1997; Samuelsson et al. 1995; Martikainen et al. 2000). In Finland alone, there are at least 4000 species which are dependent on decaying wood (Siitonen 2001). Best known decaying wood dependent taxa are beetles (*Coleoptera*) and polypores (*Polypore*) (Speight 1989). Many old growth forest species are highly specialised on decaying wood of a particular tree species, on particular species of polypore or on particular degree of wood decay (Heliövaara and Väisänen 1984; Hansson 1992). Diversity of beetles has been noted to be particularly high in those areas that have had a prolonged and continuous supply of slowly decaying wood (Nilsson and Baranowski 1997; Martikainen et al. 2000). Many of the threatened beetle species in Finland live in old growth forests and most of them are dependent on decaying wood at some stage of their life cycle (Rassi et al. 2001).

Efficient forest management requires that injured trees and decaying wood are removed from the forest. It is likely that these managemental practises have contributed to the decline of biodiversity in forests (Simberloff 1999). Therefore, by implementing new management practises, in which some trees are injured and decaying wood is left into the forest, we may be able to stop the decline of forest biodiversity. Our objective is to determine the optimal number, size and species of trees that should be left to decay that is necessary to able colonisation and long term survival of species that are dependent on decaying wood.

Conservation of valuable habitats

In Finland, ensuring biological diversity of the forests is taken into account both in Forest Act and in the Nature Conservation Act. This is based on forming and maintaining protected areas, but also on sustainable forest management in commercial forests. In the Forest Act certain key biotopes have been defined as valuable habitats, where rare and demanding species are expected to occur. The valuable habitats are defined to be in a virgin state or in a state close to that and to display the permanent characteristics which are needed to preserve their characteristic species composition. To meet the criteria of the Forest Act the valuable habitats need to be relatively small (the average size is about 0.6 ha (Tenhola and Yrjönen 2000)) and clearly distinguishable from the surrounding habitats. In Finland, term valuable habitat includes habitats of special importance (Forest Act, Section 10), protected nature types (Nature Conservation Act) and other valuable habitats that are recommended to be preserved in good forest management. The purpose of the Forest Act is to develop ecologically, economically and socially sustainable utilization of forests.

The habitats of special importance are listed in the Section 10 of the Forest Act. The list includes the immediate surroundings of boreal springs, brooks and rivulets, small lakes, grass-and-herb-rich hardwood-spruce swamps, fern-rich hardwood-spruce swamps, eutrophic fens located south of the Province of Lapland, fertile patches of herb-rich forest, heathland forests on undrained peatlands, gorges and ravines, cliffs and underlying forest stands, sandy soils, exposed bedrock, boulder fields, sparsely treed mires and alluvial forests with poorer wood yield than nutrient-poor mineral soils. In these habitats endangered and rare species are assumed or, in some cases, known to occur. In habitats of special importance, some forestry operations are prohibited (e.g. clear-cutting) but some gentle operations are allowed (e.g. single felling and gentle logging). There is no systematic empirical information on whether the valuable habitats harbour more species than other habitats or whether the area of a valuable habitat is large enough to allow long term survival of populations. Therefore, our first aim is to determine whether there are more species in valuable habitats than in control habitats. The second aim is to determine experimentally the effect of some silvicultural practices conducted in the valuable habitats or in the vicinity of them on long term survival of species richness of the habitats.

Description of the conducted and planned experiments

The effect of fire combined with the amount and quality of decaying wood on the species richness of beetles

Our main burning experiment was established during the winter 2001-2002 at Evo, Southern Finland in collaboration with the University of Helsinki and Finnish Forest Research Institute. We selected 24 two hectare plots of 80-years-old forest. The main tree species of these areas is Norway spruce (*Picea abies*) with some birch (*Betula spp.*) and Scotch pine (*Pinus sylvestris*). During the winter 2001–2002 the areas were harvested such that the volume of standing trees in each treatment area was set to 50 m^3 / hectare. Cut down trees were left to the harvested areas such that the volume of cut down wood was 5m^3, 30m^3 or 60 m^3 per hectare. The control areas were not harvested. Each treatment and control was replicated six times out of which three were burnt and three were left unburnt during the summer 2002.

From each of the 24 experimental areas, we collect beetles with window traps. Immediately after the burning in 2002, when the areas had cooled enough to allow moving on them, we settled 10 window traps on each area. The setup consists of five freely hanging traps and five traps attached to standing trees. In 2002, the trapping season begun in June and lasted until the end of September. To study the short-term colonization patterns in detail, the traps were emptied weekly. In 2003, the beetles were trapped from mid May to the end of September. Initially, our aim is to follow the colonisation and diversity of beetle fauna on these areas for three years (until the end of 2004), but a long term, at least 20 year monitoring will also be established on this experimental design.

Different beetle species require decaying wood of a different tree species as their food resource. However, when the wood has burnt the specialisation is reduced (Wikars 1997). With the window traps that were attached to standing trees in the experimental setup described above, we will determine the effect of the species of a decaying standing tree on the number and diversity of beetles. In each area the five window traps are allocated such that three of them are on spruce (main tree species on all areas) and two on birch. In the unburnt plots, the experimental trees were damaged during spring 2002 so that they have started to decay. Comparing the catch from traps on different tree species and between burnt and unburnt trees, we obtain valuable information about the importance of the quality of the reserve trees on the abundance and species richness of beetles.

In this experiment, we will also study the distribution and abundance of polypore fungi and beetles living in the fruiting bodies of the polypores. In spring 2005, we will collect samples from three common species of polypores (*Fomes fomentarius*, *Fomitopsis pinicola* and *Trametes spp.*) from our study plots. Sampled polypores will be brought into the laboratory and emerging insects will be collected and identified.

Based on our results we will be able to directly measure the effects of prescribed burning on abundance and species richness of beetles and estimate the optimal amount and quality of decaying wood for maintenance of species richness. We can also give managemental recommendations about prescribed burning and the quality and quantity of decaying wood that should be saved.

The effect of forest fire on the risk of forest damage by bark beetles

The most abundant beetle species dispersing to burnt forest are bark beetles (*Scolytidae*), the larvae of which forage in dying or damaged trees. Some bark beetle species (e.g. *Ips typographus*) are also known to attack healthy trees and they can be considered a risk for commercial forestry. Our burned study plots as well as unburnt plots offer a very suitable habitat for bark beetles because of the large amount of recently died trees. Therefore, a dramatic increase in population sizes of bark beetles can be expected and there may be an increased risk of forest damage also in the surrounding, healthy forests.

In spring and summer 2003, we studied the colonization of bark beetles to our study plots and the dispersion of them to the surrounding forests. The experimental setup consisted of a straight line of six free-hanging window traps deposited at right angle to the border of the study plot. One of the traps was settled 25 meters inside the treated study plot, one at the border of the plot and four to the surrounding forest, at the distances of 25, 50, 75 and 100 meters. Our aim is to study how the abundance of bark beetles depends on the amount of decaying wood and whether there is a difference between burnt and unburnt plots. In addition, this study will yield information on the distribution and abundance patterns of bark beetles in the healthy forests outside the study plots.

On the basis of these experiments we will be able to estimate the negative effects of restoration tools, particularly the effect of prescribed burning and increasing the amount of decaying wood on the risk of forest damage.

The succession of beetle fauna on old silvicultural burnings

In addition to the long term experimental monitoring of the succession of beetle diversity, we have studied the succession of beetle species assemblages on old silvicultural burning areas. The aim of this study is to complement the long term monitoring experiment by providing information on long term species succession with a short term study and to determine how long after the fire the burnt areas remain a suitable habitat for fire dependent beetle species. This study was conducted during the summer 2002 in Evo. We selected 20 silvicultural burning areas and 20 clear-cut areas, age of the areas ranging from 2 to 18 years. Methods of beetle capturing were the same as in our main experiment: 5 free-hanging window traps and 5 window traps attached to standing trees were settled on each area. Aim of this study is to determine how long after the fire the burnt areas remain a suitable habitat for fire dependent beetle species. Using information from this study we will be able to estimate the minimum frequency of managemental burnings in a given area that is necessary to maintain viable populations of fire dependent species.

Importance of valuable boreal brook habitats in conserving species richness

In spring 2002, we selected 20 areas containing a valuable brook habitat from boreal coniferous forests in Central Finland. These study forests are administered by the Finnish Forest and Park Service and by the forest industry enterprise UPM. From each of the 20 areas we selected three 0.1 hectare forest plots: one within the valuable brook habitat, one along the same brook but outside the protected habitat and one from the nearby forest. From each 0.1 hectare plot we determined the amount and quality of dead wood and the species richness of polypore fungi and epiphytic mosses. We counted all dead wood larger than 5 cm in diameter, identified the tree species, and measured base diameter, length of the tree and the decay stage. Based on the base diameters and the lengths of the trees we calculated the overall volume of the dead wood. The decay stage was classified on seven categories. All polypore fungi having a perennial fruiting body were identified and counted. All mosses growing on tree trunks at 50 to 250 cm height were collected and identified.

The experimental design of this study is paired because there was likely to be differences between the study areas depending on environmental factors not measured in our experiment. In the statistical analysis, we used nested and mixed model analysis of variance.

The effect of silvicultural practices on species richness of valuable forest habitats

The valuable habitats in our experiment are the immediate surroundings of boreal forest brooks and rivulets. In the Forest Act, the concept “immediate surroundings” is inadequately defined and it is poorly known how large areas of virgin forest around the brooks are needed to preserve the species composition of these valuable habitats. In our experiments, our aim is to study the importance of the width of a buffer zone on the biodiversity of the forest brooks.

At our study brooks, the surrounding forests are mature boreal coniferous forests, the main tree species being Norway spruce (*Picea abies*) with some birch (*Betula spp*.) and Scotch pine (*Pinus sylvestris*). In forestry terms, the forests are in the state of final felling. As a treatment, the buffer zone is created by clear-cut felling adjacent to the valuable habitat. The width of the buffer zone along the valuable habitat (the forest left between the brook and the clear-cut) will be 10 metres or 30 metres. In control areas, no clear-cut will be done. The length of our study section along each book will be 60 metres.

Before felling, the ecological variables as well as variables describing the tree stand will be measured. Also environmental factors affecting the habitat, such as temperature, light, moisture and water quality, are measured. Ecological variables included are vascular plants, mosses, polypore fungi and beetles inhabiting polypore fungi. In case of some polypore species, fruiting body samples will be collected and beetles living in the polypore will be reared. The plants and mosses will be monitored from one meter wide study lines which are drawn orthogonally from the brook to the edge. There will be three study lines at each study area. Every square metre of the line will be examined separately. In each square, plant and moss species will be identified and the percentage of coverage of each species will be noted.

After the first year measurements, the treatments described above will be conducted. Control areas will not be felled. The first year measurements will be a dependent control for each area, and the control area will be an independent control. In the first stage, the monitoring will be carried out yearly for four years. However, because the changes in the species composition are likely to proceed slowly, long-term monitoring will be formed and the monitoring will be carried out for at least 20 years.

Table 1. Nested ANOVA for the volume of decaying wood.

Source	SS	df	MS	F	Sig.	eta[2 b]
Area	16.5–E10	19	8.7–E09[a]	2.678	.004	.538
Treatment(Area)	14.0–E10	39	3.6–E09	2.956	.000	.042
Error	315.9–E10	2598	1.2–E09			

[a] Error term for the area has been calculated using Satterwaithe approximation
[b] eta^2 = proportion of variance explained

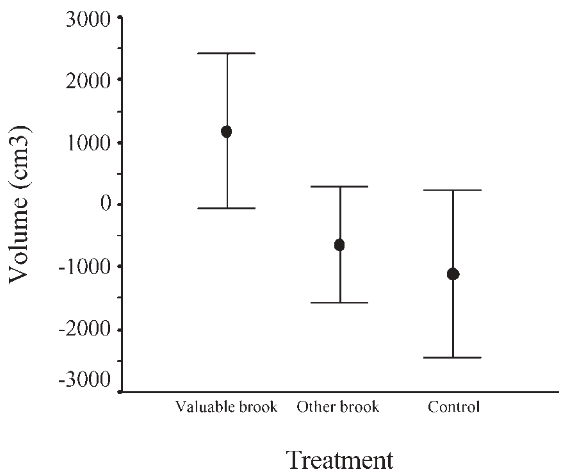

Figure 1. Volume (mean ± SE) of decaying wood at valuable brooks, other brooks and at control plots. For the figure the volume of decaying wood is standardised to mean of zero.

Preliminary results and discussion on the importance of valuable boreal brook habitats in conserving species richness

Decaying wood

The volume of decaying wood varied between the study areas but there was also a significant effect of the treatment (Table 1). However, the effect of treatment was small explaining only about 4% of the variation, while the difference between the areas explained about 54% of the variance. Although the variance is great and the difference between the treatments is only around 0.2 m^2 per hectare, the volume of decaying wood is greatest on the valuable brook habitat (Figure 1).

In addition to the volume of decaying wood also the quality measured as decay stage was significantly affected by the treatment, while there was no significant difference between the areas (Table 2). The decay stage of the dead wood was highest in the valuable habitats (Figure 2).

Table 2. Nested ANOVA for the decay stage of the dead wood.

Source	SS	df	MS	F	Sig.	eta^2
Area	227.83	19	11.991[a]	1.654	.088	.435
Treatment(Area)	323.45	39	8.293	7.613	.000	.103
Error	2829.31	2597	1.089			

[a] Error term for the area has been calculated using Satterwaithe approximation

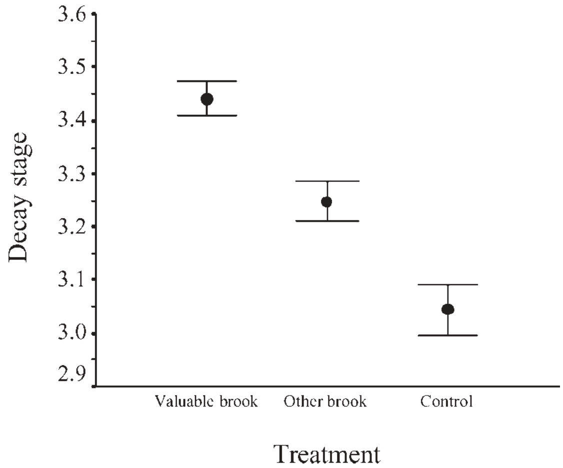

Figure 2. Decay stage (mean ± SE) of the dead wood at valuable brooks, other brooks and at control plots.

Table 3. Mixed model ANOVA for the number of moss species.

Source	SS	df	MS	F	Sig.	eta^2
Area	62.583	19	3.294	1.267	.260	.388
Treatment	19.900	2	9.950	3.828	.031	.168
Error	98.767	38	2.599			

Our results imply that the surroundings of brooks which are conserved by the Forest Act contain larger volumes of decaying wood than the nearby forests and other brook habitats. Although the differences were statistically significant they were relatively small: the mean difference between conserved brook habitats and control forests was only about 0.2 m^3 decaying wood per hectare. However, the average decay stage was highest at valuable brook habitats. This difference in the quality of decaying wood may be of importance for the diverse species community inhabiting decaying wood in boreal forests.

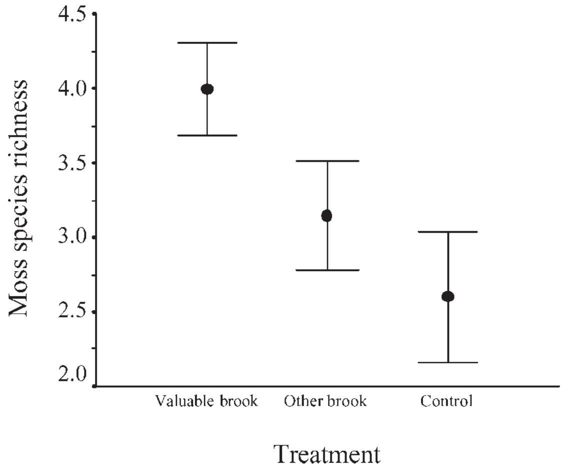

Figure 3. Species richness of mosses (mean ± SE) at valuable brooks, other brooks and at control plots.

Table 4. Mixed model ANOVA on the number of polypore species.

Source	SS	df	MS	F	Sig.	eta^2
Area	73.667	19	3.877	2.231	.017	.527
Treatment	6.633	2	3.317	1.909	.162	.091
Error	66.033	38	1.738			

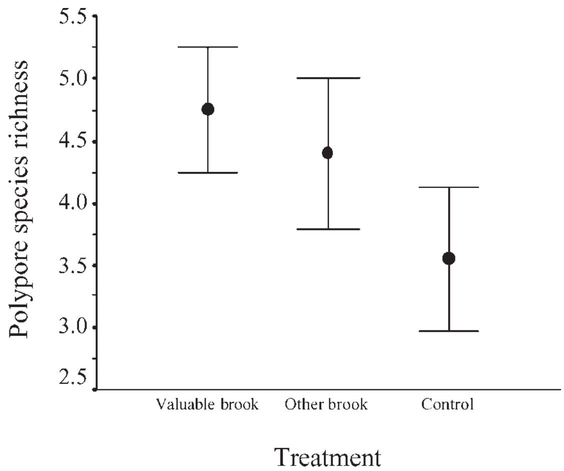

Figure 4. Species richness of polypores (mean ± SE) at valuable brooks, other brooks and at control plots.

Mosses and polypores

The species richness of mosses did not vary across study areas but there was a significant effect of the treatment (Table 3). The valuable brook habitats were most species rich while control areas were most species poor (Figure 3).

There were differences in species richness of polypores across the study areas but no significant effect of the treatment (Table 4). However, there was a tendency of the same direction as in mosses and decaying wood: valuable habitats had higher species richness than control habitats (Figure 4).

The species richness of epiphytic mosses was highest at the valuable brook habitats and smallest at control forest. Similar tendency, although not statistically significant, was found in the species richness of polypore fungi. Concerning these taxa, it seems that preserving the characteristic features of valuable brook habitats may help to preserve the overall species richness of boreal forests. This poses new challenges to silviculture: in the future it will be crucial for managers recognize the special characters that make a valuable habitat. Our ongoing studies are aimed to help in this task and to give answers to the question of what are the most important characteristics of valuable habitats that allow the occurrence of higher species richness.

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Factors Controlling the Spatio-Temporal Dynamics of a Forested Landscape Affected by Fire in Central Spain

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Abstract

Mediterranean landscapes are dynamic systems that undergo temporal changes in composition and structure in response to stochastic varying factors, including disturbances such as fire. Determining landscape dynamics according to the different fire regimes is a central question in disturbance ecology. In this paper we explored the spatio-temporal dynamics of an area centered on Sierra de Gredos (Central Spain) subjected to spatially heterogeneous fire regimes. A set of 13 Landsat MSS images, covering a period of 16 years, were processed and classified to obtain annual profiles of landscape patterns and fire regime. The study area was divided into a lattice of 45 sites of 10 × 10 km characterized annually for landscape composition, structure, topographic complexity and fire regime properties. An exploratory analysis of the spatio-temporal dynamic system was carried out using a multivariate approach. Temporal Principal Component Analysis AnalysEs (TPCA) at one year time step over spatially integrated variables were calculated to identify the most significant variables explaining the system. On the other hand, Temporal Spatial Cluster Analysis AnalysEs (TSCA) on 45 sites over temporal integrated variables allowed to identify regions with different dynamics. The results indicated the existence of significant variables, related to composition, structure and cumulative effect of fire, explaining the spatio-temporal system dynamic. These "driving" variables showed trends and stochastic temporal patterns characterized for a non-stationary and non-periodic nature. The asynchronic or delayed relationship among variables requires time series analysis to quantify this kind of complex relationships. The contagious nature of spatial clusters indicated the high scale dependence of the phenomena studied.

Keywords: Spatio-temporal dynamics; forest fires; stochasticity; asynchrony; PCA; cluster analysis.

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Introduction

One of the major challenges in “disturbance ecology” is to predict variability in fire effects across heterogeneous landscapes (Urban et al. 1987; Turner et al. 1989). Landscape pattern dynamics in human-landscapes affected by spatially heterogeneous fire regimes can be enormously complex due to several patterns and processes that are continuously interacting, evolving, and adjusting in time and space. The estimation of the relative significance of disturbances in a system dynamic is yet a mathematical challenge as disturbances act simultaneously with other ecological processes that affect landscape patterns with possible feedbacks among them (Baker 1995; Levin 1992). The nature of vegetation change based on disturbance and succession and the non-equilibrium state of vegetation are the main ecological processes, in disturbed landscapes, that must be defined. Hence, a conceptual framework able to mathematically estimate them is required to have a realistic knowledge of system dynamic.

The current state of knowledge about landscape-fire interaction is typified by different conceptual frameworks of spatial and temporal scales and the modeling approaches underlying. Usually, the common empirical approaches to explain changes in landscape patterns due to the effect of disturbance have consisted on static comparisons among sites before and after the disturbance using parametric or non-parametric difference tests (Leitner et al. 1991; Knick and Rotenberry 1997). In this kind of studies, the temporal component is avoided and only spatial differences are considered. In others, a chronosequence of sites aged by the time since last fire allow to study the disturbance effects from a static point of view (Frelich and Reich 1995). In others, when comparing undisturbed to disturbed areas (Mladenoff et al. 1993) or areas with different fire regimes, the selected sites are located under a random sampling technique to avoid spatial dependence. This kind of sampling design show statistical advantages opposite to temporal series data collected from contiguous areas in which the temporal and spatial dependence of data make classical statistical inference methods unsuitable.

In this work we propose to explore the role of fire regime on temporal dynamics of heterogeneous disturbed sites. Multivariate data analysis appear as an indicated tool to identify biological systems without superimposing a priori theories on the data showing their ability to describe multiple scale phenomena detecting both step (abrupt) and monotonic (gradual) patterns in time and space. These techniques allow to characterize spatially temporal dynamics of multiple variables identifying similar zones due to the inherent dynamics of them obtaining a parsimonious description of temporal and spatial patterns.

In this sense the main objectives were to use multivariate analysis as an exploratory tools to identify the nature of the spatio-temporal interactions between fire succession and landscape patterns with the aim: i) to identify the main system variables driving the temporal dynamic of the study area, ii) to identify different temporal dynamics according to environmental and fire regime differences, iii) to assess the spatial distribution of the dynamics observed and finally, iv) to choose the most adequate statistical tools to quantify the spatio-temporal relationships between variables.

Methods

The study site

The study area is located in Sierra de Gredos (province of Avila, Spain). The site, 95 × 55 km in size (518 331 ha), is located between the UTM coordinates 4430–44845 and 277–372 in the zone 30 North . The area is mountainous, with elevations up to 2200 m and a strong elevation gradient, from the Southeast to the Northwest. The largest portion of the territory was comprised between 400 and 600 m, in gentle slope (up to 10% inclination) and in either flat or exposed to S and SE. The potential vegetation of the area corresponds mainly to four different types: sclerophyllous oak forest (*Pyro bourgaeanae-Quercetum rotundifoliae*); humid deciduous oak forest (*Arbutus unedonis-Quercetum pyrenaicae*); sub-humid deciduous oak forest (*Luzulo forsteri-Quercetum pyrenaicae*) and woody-legume shrublands (*Cytiso purganti-Echinospartetum barnadessi*). Nevertheless, despite the different temperature and precipitation conditions marked by the said vegetation units, managed pine woodlands dominate the landscape.

Imagery Acquisition and Processing

Satellite images and DTM (Digital terrain Model) data were processed to derive land cover and terrain information for the study area, respectively. Landsat MSS (1975–1990) images were selected for this study due to its temporal resolution and span. The MSS images used were obtained from the satellites Landsat 1-5 (Path 217 and Row 32, for the images between 1975-1978 and Path 202 and Row 32 for the images between 1980–1990) for the following dates: July 20, 1975; August 19, 1976; August 18, 1978; September 30, 1980; September 13, 1982; October 2, 1983; October 12, 1984; October 15, 1985; November 3, 1986; September 21, 1988; October 20, 1989; and August 20, 1990. The images were geometrically and radiometrically corrected for atmospheric variations and for the topographic effect to make them comparable (Viedma and Moreno 2003). Later, annually NDVI indices (Normalized Difference Vegetation Index) were calculated. Fire perimeters ≥ 10 ha occurred during the period 1975–1990 were mapped based on a multitemporal analysis of the NDVI indices (Viedma and Moreno 2003).

The land-use/land-cover (LULC) legend of the CORINE programme was applied to obtain annual land-cover maps of area. To classify the 13 Landsat MSS images a set of ancillary data: digital LULC CORINE map of 1987, analogic maps of LULC for the years 1976–1978 and the DTM were used. A mixed supervised-unsupervised classification method using "temporal context" and incorporating information on topography and ecology stratification was applied (Viedma and Moreno 2003).

Landscape structural features were assessed by computing the following indices: Patch Density (PD), Mean Patch Size (MPS), Shannon's Diversity Index (SHDI), Mean Core Area of patches (MCA), Edge Density (ED) and Contagion (CON). These spatial metrics were calculated yearly during 1975–1990 using FRAGSTATS software (McGarigal and Marks 1995). A digital terrain model (DTM) with lines at 100 m was used to assess the topographic complexity through landscape metrics on topographic patches created from the combination of elevation, slope and aspect ranges.

Spatio-Temporal analysis of the system: Identification of driving variables and spatial patterns.

Annual series of landscape patterns (composition and structure) and processes (fire regime) were obtained from a square lattice of land sites of 10 × 10 Km over the study area (Table 1). Temporal Principal Component Analysis (TPCA) on the main variables during 1975–1990, integrated spatially over the 45 sites, was applied to identify temporal patterns on the interacting variables. We would seek, through orthogonal axis of PCA, the synchronous or delayed dynamics among variables. The TPCA for 1975–1990 was carried out on a matrix of

Table 1. Main variables considered for each 45 sites of 10 × 10 K min which the entire area was divided.

Variables	Generic Name	Variables
Landscape structure	Patch properties	PD, MCA, MSI
	Landscape entropy and distribution	IJI, SHDI, MNN
Landscape composition	Pine woodlands	Pine Woodlands
	Shrublands	Shrublands
	Pastures	Pastures
	Croplands	Croplands
	Dehesas	Dehesas
	Bare soils	Bare soils
	Water bodies	Water bodies
Topographic complexity	Patch properties	PD, MCA, MSI
	Landscape entropy and distribution	IJI, SHDI, MNN
Fire properties	Annual burned area	
	Accumulated burned area	
	Fire frequency	No of fires (1975–1990)

14 temporally dynamic variables (landscape structure and composition and, annual burned and accumulated burned area) plus 7 temporally constant variables (6 related to topographic complexity and 1 related to fire regime: number of fires between 1975–1990) pooled for the 45 sites in which the study area was divided. The TPCA analysis was developed in SPSS version 11.5. We extracted the r-principal components with eigenvalues greater than 1 using the Varimax method of components rotation on standarized variables. Later, we qualitatively followed the temporal dynamic of each PC over time.

Temporal Spatial Cluster analysis (TSCA) allowed to explore broader scales of land cover patterns over time. Cluster Analysis was based on the Ward's method due to its statistically behaviour that was similar to PCA. The matrix used for TSCA was constituted by 45 columns (sites) and 167 rows (14 variables × 12 years + 7 fixed variables). We characterized each clustered sites by the statistically different variables using ANOVA and Bonferroni post hoc test. Finally, we qualitatively followed the temporal dynamic of significant variables to show up the differences and similarities in time.

Results

During the study period (1975–1990), a total of 121 fires were recorded, which swept across a total area of 29 144 ha (nearly the 8% of the forested area and 5.6% of the entire study area). The mean annual rate of burned area was 0.7%, thus corresponding to 142 years of fire rotation period in a non random process. Fires were not distributed across the whole area, but concentrated mainly in the central mountain chain ("the Sierra"), with an apparent high degree of contagiousness. The study area was dominated by shrublands and pastures, with pine-covered areas being relatively scarce.

Table 2. Main PCs obtained from the TPCA on the main variables during 1975–1990 integrating the 45 sites.

PCs	Name of the PC	Variables	% Variance explained	% Variance accumulated
1	Shrublands	Shrublands (75–90), IJITOP, NPTOP, SHDITOP	25.7	25.7
2	Pastures	Pastures (75–90)	21.8	47.5
3	Pine Woodlands	Pine woodlands (75–90)	13.9	61.5
4	Landscape Entropy	IJI (75-90), SHDI (75–90)	7.3	68.8
5	Patch Geometry	MSI (75–90)	6.1	74.9
6	Spatial Distribution	MNN (75–90), Crops (75–83)	4.2	79.1
7	Spatial Fragmentation	NP (75–90)	3.8	82.9
8	Bare Soils	Bare Soils (75–90), MNNTOP	3.2	86.2
9	Patch Size	MCA (75–90)	2.6	88.8
10	Regeneration	Regenerated area (78–90), Area burned (76,78,82,86)	2.1	90.9

Temporal Dynamics (Multiannual Patterns)

The TPCA on selected variables showed that with only 10 PCs near 91% of the variance in the original data set was retained, which suggests considerable data structure (168 variables were expressed by fewer generalized variables). Several PCs were described by the temporal dynamic of only one variable (Table 2). Only few PCs included different variables (i.e., CP6) and few collected certain spatio-temporal variability of the original data (i.e., PC1 and PC8). The distribution of variables in each PC indicated that temporal variability was strongest than the spatial variability, and that the temporal dynamic of the main variables was uncorrelated or delayed in time. Finally, annual burned area did not explain so much of the temporal variance observed in data due to its stochastic nature. Opposite, the cumulative effect of fire, measured by the regenerated burned area had an important role in explaining the overall data variability (Table 2).

The temporal dynamic of landscape composition variables was evidently uncorrelated in a synchronic way showing almost three temporal patterns (Figure 1 A–C). On one hand, the pattern of shrublands and pastures (Figure 1 A), on other, the dynamic of pine woodlands and accumulated burned area (Figure 1 B) and finally, bare soils with annual burned area (Figure 1 C).

The temporal dynamic of landscape structure variables was again uncorrelated in a synchronic way (Figure 2 A–B). There was a evident relationship among the patch properties (NP, MSI and MCA) (Figure 2 A) with a similar enveloping, but with a cyclic components clearly inverse. These behaviours were predictable due to the nature of the patch metrics. These opposite behaviours, but in the same frequency, clearly indicated that landscape tended to be less fragmented with patches of bigger size less complex geometrically. On the other hand, the temporal dynamic of the landscape entropy variables (IJI and SHDI) showed common behaviour in period and intensity but with little delay (lag-time) in the last years (Figure 2 B). Landscape properties associated to patch metrics showed a different temporal pattern than landscape properties related to the entropy. These dynamics indicated that landscape metrics were sensitive to different temporal scales. The TPCA indicated that a delayed temporal relationship among

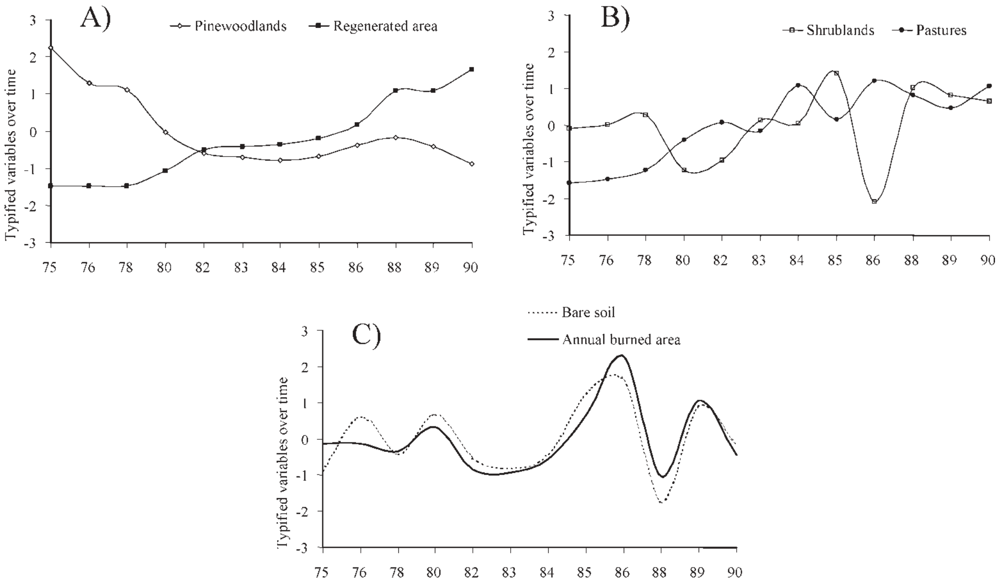

Figure 1. Temporal dynamic of the variables related to landscape composition and fire regime integrated over the 45 sites.

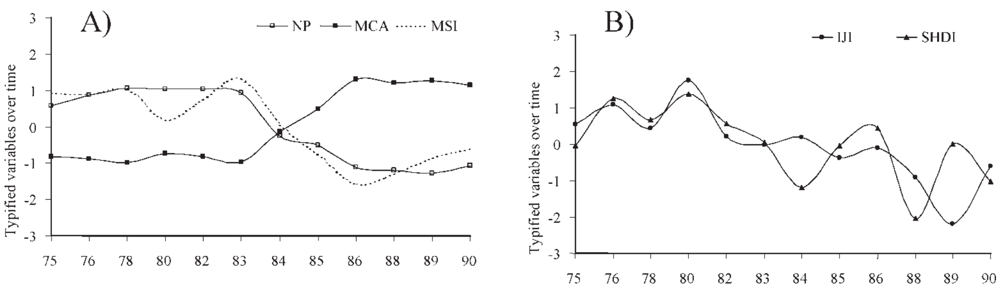

Figure 2. Temporal dynamic of the variables related to landscape structure integrated over the 45 sites.

variables could be determined. The observation of non-periodic cyclic dynamics implies that temporal series techniques may be suitable to quantify them.

The Temporal Spatial Cluster Analysis (TSCA) on the 45 square sites was carried out testing the effects of different number of clusters in the stability of results. We checked that with only 4 clusters the results were rather stable due to any increment in the number of cluster did not produce any global segregation and not increased the spatial variability explained. The spatial distribution of these clusters showed a high spatial dependence with clear contagiously pattern (Figure 3).

An ANOVA analysis and Bonferroni post hoc test were carried out to determine significant variables separating the clustered sites. Land cover proportions alone were enough to obtain distinctive clusters according to landscape disturbance. The main results indicated that

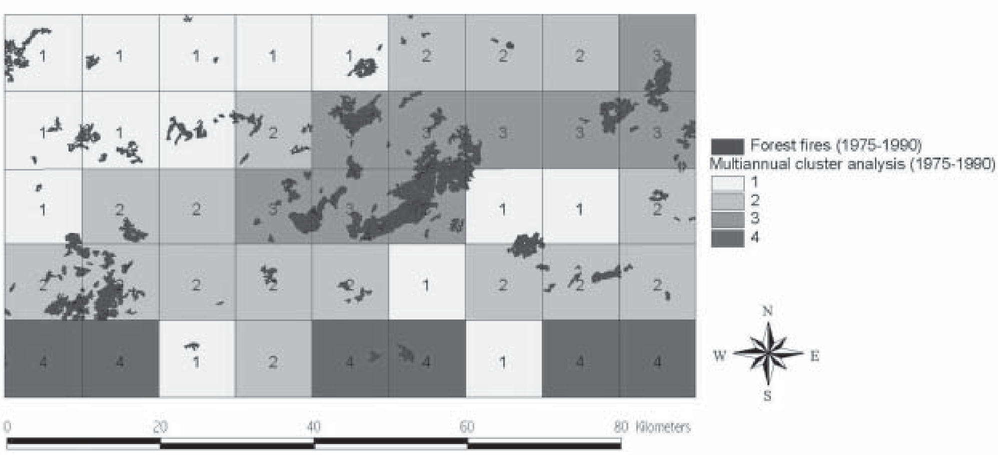

Figure 3. Spatial patterns of the clustered sites according to the temporal dynamic of landscape composition, structure and fire regime variables.

Cluster 1 was defined by the spatial domain of pastures (Figure 4 A). It is a high elevated zone located in the NW part of the study area. Its spatial configuration was defined by a great spatial homogeneity (SHDI) with irregular patches of big size, although this spatial structure was not significantly different of the rest of clusters. The fire incidence was low.

The Cluster 2 was defined by a mixed vegetation composition of shrublands and pastures (Figure 4 B). It was the most fragmented area with high heterogeneity and high spatial contiguity between patches. It was spatially allocated around the central mountain chain (the Sierra). The fire incidence was medium. This region constituted a transition zone between two regions with very different dynamics. The Cluster 3 was defined by the spatial domain of pine woodlands (Figure 4 C) It was a fragmented area with the highest spatial heterogeneity. It is located in the central mountain chain with the highest fire frequency and size. The annual burned area did not show any statistical evidence that allow the characterization of the cluster obtained. However, the regenerated burned area characterized significantly Cluster 3. (Figure 4 D). Finally, the Cluster 4 was defined by the spatial domain of croplands and the absence of shrublands (Figure 4 E and B). It was the area less fragmented with high spatial and compositional homogeneity. This area is located in the South of the study area with a very soft topography. The fire incidence was the lowest.

Finally, we tried to relate the dynamics of variables temporally integrated by TSCA with the variables spatially integrated by TPCA. The aim was to determine what variables spatially integrated were able to separate dynamically regions averaged in a temporal way. The results indicated that the temporal dynamic of pastures (PC 1) integrated for entire study area was redraw by the region defined by cluster 2 (dominated by a mixture of pastures and shrubs) and cluster 1 (dominated by pastures) (Figure 5 A–B).

The general integrated dynamic of shrublands (PC 2) fitted with the dynamic of shrublands in cluster 3 (the area dominated by pine woodlands) and cluster 2 (shrubland + pasture mosaic) (Figure 5 C–D). The temporal dynamic of pine woodlands in the entire area was the reflection of pines dynamic in cluster 3 (Figure 5 E–F) and finally, croplands dynamic was unable to separate among clustered regions and was rather similar in clusters 1, 2 and 4 (Figure 5 G–H) taking no effect on spatial associations.

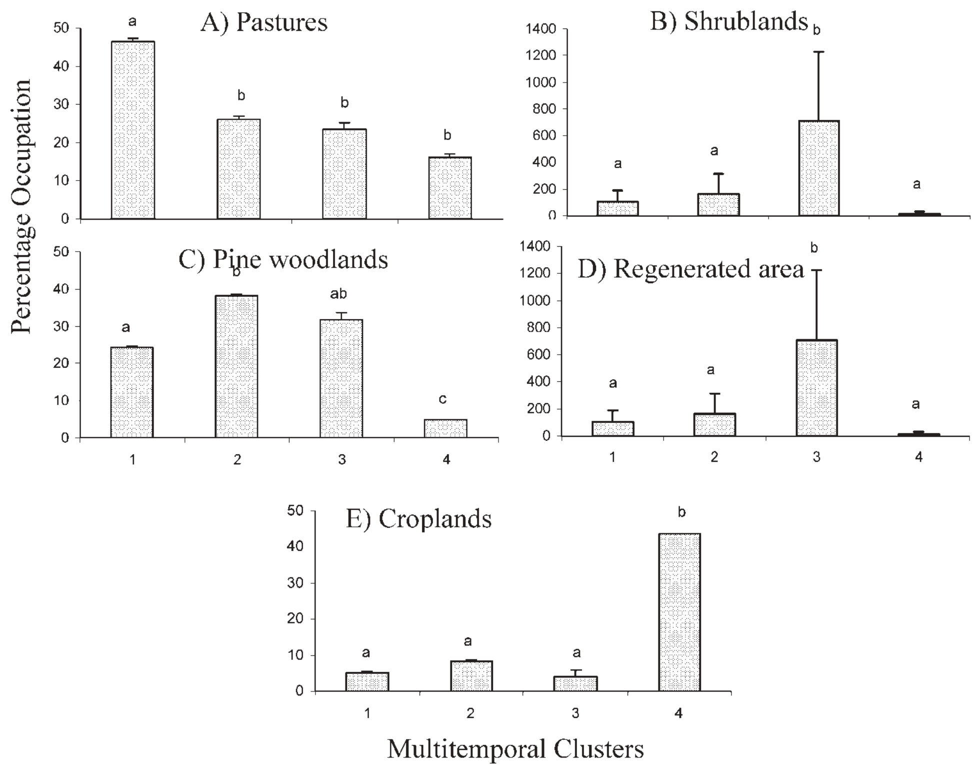

Figure 4. Percentage occupation of the main landscape composition variables and regenerated area (in hectares) for the regions identified by TSCA, indicating the statistical separability from ANOVA analysis.

Discussion

The landscape analysed underwent changes in vegetation composition and structure in response to cyclically fluctuacting disturbance patterns and the recovery process thereon associated. Several PCs were described by only one variable in its temporal dynamic and, few collected certain spatio-temporal variability of the original data. The distribution of variables in each PC indicated that temporal variability was strongest than the spatial variability, and that the temporal dynamic of the main variables was uncorrelated or delayed in time. Finally, annual burned area did not explain so much of the temporal variance observed in data due to its stochastic nature whereas the cumulative effect of fire, measured by the regenerated burned area, had an important role in explaining the overall data variability. We have found two broad groups of variables according to their temporal behavior: one dominated by cycles of high frequency and non-periodic oscillations and, the other one controlled by cycles of low frequency and certain trend. In general, large amplitude and regularity of the oscillations indicate that only few factors or interactions are dominating dynamics whereas short and non regular cycles indicated that more factors were implied on them (Kendall et al. 1999). Although the most significant variables of each PC were uncorrelated, we have emphasized through the qualitative analysis of temporal dynamics that a cross-correlationship was acting among PC variables in the framework of time series, indicating that a delayed temporal

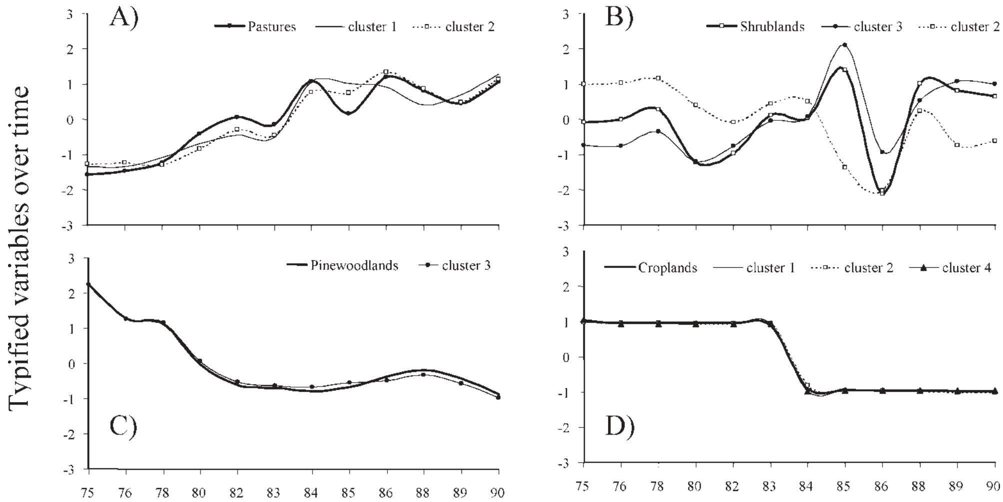

Figure 5. Comparisons among temporal dynamic of the main landscape composition variables integrated over the 45 sites and the dynamic of the same variables on the regions established by TSCA.

relationship among variables could be determined. The observation of non-periodic cyclic dynamics imply that temporal series techniques may be suitable to quantify them.

Landscape composition and structure dynamics are the result of different processes acting across finite spatial domains. With a multitude of influential factors there may be no single scale of regulation at lanscape scale. But when some processes are more influential than others then, they may determine the spatial scale of dynamical patterns (Bjørnstad et al. 1999). A characteristic spatial scale implies similarity of the dynamics within the area of influence and dissimilarities out of it. In this work, the regions defined by means Clusters and Principal Components indicated that the temporal dynamic of pastures, shrublands, pine woodlands and accumulated burned area were enough to segregate spatially the study area. But landscape structure variables were not determinant in the spatial scale of these dynamics because different processes produced similar patterns. Finally, the spatially contagious nature of the identified regions will require statistical analysis able to handle the degree of spatial autocorrelation among sites within each cluster and the spatial dependences among clusters for defining a “spatial model” of causal relationship among variables.

The most common techniques applied to time series analysis resulted inadequate for our analysis because require long series and did not take into account the role of explanatory variables. Two statistical techniques crucial to study cycling non-periodic phenomena are AutoRegresive models (AR) and the Spectrum Analysis (SA) and, to define a temporal causal model taking into account lag time delays the SISO models (Single Input-Single Output) turn into the most suitable causal statistical tool for quantifying these relationships.

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The Sicilian Phanerophytes: Still a Noteworthy Patrimony, Soon a Lost Resource?

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Abstract

This paper is focused on the Sicilian phanerophytes. The rather high species-richness of this group depends not only on the central position of the island within the Mediterranean area and its probable key-role as a refuge-area during Pleistocene glaciations, but also on its wide latitudinal/altitudinal range, its high topographic and substrate heterogeneity and the high frequency and intensity of disturbance in the last 10 000 years. The present paper contains either an up-to-date check-list of Sicilian native phanerophytes or a critique reference list, which together shall be considered as "first-aid tools" for a better knowledge of this topic. Furthermore, this work aims to put in evidence the present lack of information on the biology, the demography and the population genetic structure of most part of the Sicilian phanerophytes. Future researches should 1) clarify the dispersal paths of Sicilian phanerophytes during the Pleistocene glaciations, 2) the role played by habitat and landscape fragmentation on the present species-richness and peculiarity of Sicilian forest flora and 3) quantify the risk-level connected with the low genetic flux between (and the low variability within) Sicilian phanerophyte populations.

Keywords: Sicily; phanerophytes; intra-specific diversity; genetics; conservation.

Introduction

The Mediterranean Basin is one of the most important plant biodiversity hot-spots of the World Médail and Quézel 1997; Heywood 1998; Myers et al. 2000). Due to its peculiar position, Sicily played and still plays a major role for plant dispersal, survival and evolution within this region. On the island (ca. 25 000 km^2) live about 2700 native vascular plants, many of which are endemic, rare and/or threatened taxa (Di Martino e Raimondo 1979;

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Raimondo et al. 1994; Brullo et al. 1995). The same occurs if we consider only phanerophytes, represented by nearly 100 infrageneric taxa (S. Pasta, pers. data).

The phytogeographic relevance of the Sicilian forest flora was already put in evidence by Quézel (1995) and Barbero et al. (2001). The supra- and the oro-mediterranean belts (Brullo et al. 1996b) of the island give hospitality to some trees endemic to Sicily (*Abies nebrodensis* and *Betula aetnensis*) or to Southern Italy (*Pinus laricio* subsp. *calabrica*, *Quercus petraea* subsp. *austrotyrrhenica* and *Sorbus aucuparia* subsp. *praemorsa*) and clearly deriving from more widespread Mid-European or Oromediterranean taxa, but also to many other woody species typical of the mid-european deciduous woodland and shrubland, such as *Acer* spp., *Fraxinus excelsior*, *Malus sylvestris*, *Salix cinerea*, *Sorbus aria* s.l., *Sorbus torminalis*, *Taxus baccata*, *Tilia platyphyllos*, *Ulmus glabra*, etc.

Many other deciduous trees and shrubs live in the meso- (and even in the thermo-) mediterranean belt (Brullo et al. 1996b). Among them we find many noteworthy endemics – such as *Salix gussonei* (closely related to *Salix pedicellata*), *Zelkova sicula*, *Quercus gussonei* and *Q. leptobalanos* (closely related to *Q. cerris* and to *Q. congesta*, respectively: cfr. Brullo and Marcenò, 1985) – many species belonging to the Eastern Mediterranean floristic element, such as *Platanus orientalis*, *Carpinus orientalis*, *Ostrya carpinifolia*, *Celtis tournefortii* s.l., *Quercus cerris* and *Q. congesta*, *Crataegus orientalis* subsp. *presliana*, etc., and some lauriphyllous evergreen trees such as *Ilex aquifolium* and *Laurus nobilis*.

Among the thermophilous woody species which dominate the Mediterranean sclerophyllous evergreen maquis (*Arbutus unedo*, *Ceratonia siliqua*, *Chamaerops humilis*, *Erica arborea*, *Myrtus communis*, *Olea europaea* var. *sylvestris*, *Phillyrea angustifolia* and *P. latifolia*, *Pistacia lentiscus*, evergreen *Quercus*, *Rhamnus alaternus* and *R. lycioides* subsp. *oleoides*, *Viburnum tinus*, etc.) – many of which belong to tropical families and/or genera - some are endemic to Sicily, like *Rhamnus lojaconoi* (closely related to *R. alaternus*: cfr. Raimondo 1979) and *Cytisus aeolicus*, or to the Central Mediterranean area, as *Genista thyrrena* and *G. aetnensis*.

Many drought-resistant plants which live in the harshest areas of Sicily belong to the Tethydic or to the SE-Mediterranean element (*sensu* Takhtajan 1984) and are needle-leaved (*Pinus halepensis*, *P. pinaster* subsp. *hamiltonii* and *P. pinea*, *Juniperus oxycedrus* subsp. *macrocarpa* and *J. turbinata*) or show an ephedroid habit (e.g. *Tamarix* spp., etc.); other ones behave as summer (semi)deciduous and often belong to tropical families and/or genera, like *Euphorbia dendroides*, *Rhus pentaphylla* and *R. tripartita*, etc.

The rather high species-richness of Sicilian woody flora and the overwhelming variability of its landscape depends also on 1) its wide latitudinal (35° to 39° N) and altitudinal (up to 3300 m a.s.l. on Mount Etna) ranges; 2) its high topographic and substrate (both soil and rock types) heterogeneity; 3) the high frequency and intensity of both natural (volcanic eruptions, earthquakes, wildfires) and anthropic disturbance in the last 10 000 years. Besides, during Pleistocene glaciations, together with – and perhaps more than – the Italian Peninsula (Bennett et al. 1991; Hewitt 1996; Taberlet et al. 1998; Trewick et al. 2002), Sicily seems to have played a role of refuge-area for many mesophilous deciduous trees (e.g. *Fagus sylvatica*, *Populus tremula* and *Juniperus communis* s.l.), some of which spred again from the island to recover the neighbouring areas of Mid- and Northern Europe during the warmer (in Europe) and wetter (on Mediterranean mountains) periods.

The present work aims to provide an up-to-date list of Sicilian native phanerophytes and to put in evidence the enormous threats (either at species or at population level) these plants undergo due to human activities. Infact, more than 9 millennia of more or less continuous human impact (Malone and Stoddart 2000) induced a strong reduction, fragmentation and transformation of the Sicilian natural landscape. The most affected vegetation units were forest and pre-forest communities. During the last century, a steep increase of anthropic pressure (urbanization, especially along the coasts; afforestation practices using allochthonous germplasm probably inducing genetic contamination of local races; change of agricultural techniques, etc.) determined a further reduction of species-richness and caused an even more severe fragmentation and degradation of the remnant semi-natural and sub-natural communities (i.e. woodlands, shrublands, garrigues, grasslands and coastal vegetation), so that many woody species disappeared or still survive with very small and scattered populations. Nevertheless, residual patches of the forest and pre-forest communities still survive in the most unsuitable areas, and in some areas it is still possible to record not only rather high values of wilderness and species-richness but also very peculiar synecological (Brullo and Marcenò 1985; Brullo et al. 1996a, 1999) and structural-typological (La Mantia et al. 2000, 2001) features.

Materials and methods

In the following paragraph we provide a list of all the Sicilian native phanerophytes linked to forest and pre-forest habitats (even those whose origin is still uncertain); neither introduced subspontaneous species, nor hybrids, nor doubtful species (e.g. *Quercus sicula* Borzì) have been taken in account. This list takes origin from the most recent biosystematic and genetic literature concerning these plants; among the papers concerning the ecophysiology and/or the productivity of Sicilian phanerophytes, just the few ones which provide some comparison between different Sicilian and/or Italian populations have been considered.

Results

The taxomomic and nomenclatural treatment of the taxa listed in Table 1 follows Pignatti (1982) and several recent monographs.

Discussions and Conclusions

At present the most interesting relic Sicilian forest and pre-forest areas are protected, as they fall into the Etna, Alcantara, Nebrodi and Madonie parks or within the 80 nature preserves. Nevertheless, the knowledge on the biology, the demography and the population genetic structure of most part of the Sicilian phanerophytes is still unsatisfactory. As a matter of fact, the only well studied taxa are *Abies nebrodensis*, *Cytisus aeolicus*, *Q*. cfr. *petraea* and *Zelkova sicula*, while there is no recent literature about more than 2/3 of the considered taxa (cf. Table 1). Besides, the studies on the Sicilian *Fagus sylvatica*, *Quercus ilex* and *Q*. gr. *pubescens* populations do not cover the whole range of these species within the island.

Many phanerophytes are experiencing a demographic crisis, which seems to be mainly linked to habitat distruction and discontinuity. Thus, future researches on Sicilian phanerophytes, in some cases common at the need of research in Europe (Scarascia Mugnozza et al. 2003), must pay a particular attention on three topics: 1) the dispersal and genetic differentiation paths of Sicilian populations (to, out from and within the island) during the Pleistocene glaciations; 2) the role played by habitat fragmentation and landscape heterogeneity on the present diversity, rarity and peculiarity of Sicilian forest flora; 3) the risk level connected with the low genetic flux between (and the low variability within) Sicilian populations.

Table 1. Checklist of all the (certainly or doubtfully) native Sicilian phanerophytes living in forest and pre-forest habitats. In bold: endemic taxa; (D): taxa whose origin is still unclear (i.e. probably introduced by men in the past centuries); cv.-rns: the available references, concerning some autochthonous cultivars, are not shown; ER = extremely rare (< 50 individuals growing in the wild); VR = very rare (< 500 individuals growing in the wild); R= (< 5000 individuals growing in the wild); L = localised (quite common but showing a somewhat narrow geographic, ecological or dynamic amplitude); C = common (5000–500 000 individuals growing in the wild); VC = very common (> 500 000 individuals growing in the wild); U = unknown; I = increasing; S = steady; D = decreasing (during the last 50 years); A: available (for the numbers cfr. literature); NA = not available.

Taxon	Presence/distribution, demographic trend and major threats	Reference(s)
***Abies nebrodensis* (Lojac.) Mattei**	ER; S; out-breeding difficulties([1]), genetic pollution hazard with *Abies alba* (not confirmed), low fitness	Ducci et al. 1999; Gramuglio 1967; Michelozzi 1997; Parducci et al. 1999, 2001a-b; Raimondo et al. 1990; Schicchi et al. 2000; Schicchi et al. 2003; Vendramin et al. 1995a-b; Vicario et al. 1995
Acer campestre L.	C; S: habitat disturbance	NA
Acer monspessulanum L.	L; S: habitat disturbance	NA
Acer obtusatum Waldst. et Kit.	L; S: habitat disturbance	NA
Acer opalus Miller	L; S: habitat disturbance	NA
Acer platanoides L.	R; S: habitat disturbance	NA
Acer pseudoplatanus L.	R; S; habitat disturbance; genetic pollution hazard (not confirmed)	NA
Alnus glutinosa (L.) Gaertner	L; D: habitat disturbance	NA
Arbutus unedo L.	C; D; habitat disturbance, genetic pollution hazard, low fitness	NA
***Betula aetnensis* Rafin.**	L; S: global warming	Biondi and Baldoni 1984
Carpinus orientalis Miller	R; U: habitat disturbance	NA
Celtis australis L. (D)	VC; I (but D in rural areas)	NA
Celtis tournefortii Lam. s.l.	L; S: habitat disturbance, out-breeding difficulties([1])	Troia 1997
Chamaerops humilis L.	C; D: land-use change	NA
Crataegus laevigata (Poiret) DC. (= "*C. oxyacantha*" Auct.)	L; U: habitat disturbance, land-use change	NA
Crataegus monogyna Jacq.	VC; S: land-use change (local cultivars are disappearing)	NA
Crataegus orientalis M. Bieb. subsp. *presliana* K.I. Chr. (= *C. laciniata* Ucria)	L; U: habitat disturbance, land-use change	NA
***Cytisus aeolicus* Guss.**	VR; D; out-breeding difficulties([1]), low fitness	Conte et al. 1998
Erica arborea L.	C; D: land-use change	NA

Table 1. continued.

Taxon	Presence/distribution, demographic trend and major threats	Reference(s)
Fagus sylvatica L.	L; D: global warming, habitat disturbance	Comps et al. 1991; Demesure et al. 1996; Denk et al. 2002; Giannini 2003; Giannini and Borghetti 2001; Leonardi and Menozzi 1995, 1996; Rossi et al. 1996; Vettori et al. 2004; Paffetti et al., in press
Ficus carica L.(D)	VC; U: land-use change (local cultivars are disappearing)	A (cv.-rns)
Fontanesia phillyraeoides Labill.	ER; U: habitat disturbance, low fitness	NA
Fraxinus angustifolia Vahl subsp. *angustifolia*	C; D: land-use change, habitat disturbance (local cultivars are disappearing)	Crescimanno et al. 1993; Ilardi and Raimondo 1999
Fraxinus excelsior L.	ER; U: global warming, habitat disturbance, out-breeding difficulties([1])	Ilardi and Raimondo 1999
Fraxinus ornus L.	C; S (D in rural areas, where local cultivars are disappearing)	Crescimanno et al. 1993; Ilardi and Raimondo 1999
Genista aetnensis Rafin.	L; I (often used for afforestation purposes outside its natural range)	NA
Genista thyrrena Valsecchi	L; S: land-use change	De Castro et al. 2002; De Marco et al. 1987
Ilex aquifolium L.	L; S: global warming, habitat disturbance, out-breeding difficulties([1])	NA
Juniperus communis L. s.l.	L; S: global warming, habitat disturbance	Adams and Pandey 2003
Juniperus oxycedrus L. subsp. *macrocarpa* (Sm.) Ball	L; D: habitat disturbance	NA
Juniperus turbinata Guss.	L; D: habitat disturbance, out-breeding difficulties([1])	NA
Laurus nobilis L. (D)	L; S	NA
Ligustrum vulgare L.	VR; U: habitat disturbance	NA
Malus sylvestris (L.) Miller	L; U: habitat disturbance (D in rural areas, where local cultivars are disappearing)	NA

Table 1. continued.

Taxon	Presence/distribution, demographic trend and major threats	Reference(s)
Mespilus germanica L.	L; D: habitat disturbance, out-breeding difficulties(1)	NA
Myrtus communis L.	C; D: habitat disturbance (local cultivars are disappearing)	NA
Ostrya carpinifolia Scop.	L; S: global warming, habitat disturbance	NA
Phillyrea angustifolia L.	L; U: habitat disturbance	NA
Phillyrea latifolia L.	C; S	NA
Pinus halepensis Miller	L; D (native populations) and I (planted stands and disturbed areas): genetic pollution hazard	NA
Pinus laricio Loudon subsp. *calabrica* Cesca et Peruzzi	L; D: habitat disturbance, sylvicultural practices change, genetic pollution hazard with *Pinus nigra* s.l.	Rafii et al. 1996; Cesca and Peruzzi 2002; Paci and Ricciardi 1988
Pinus pinaster Solander subsp. *hamiltonii* (Ten.) Huguet del Villar	L; U: land-use change, habitat disturbance	Anzidei et al. 2000; Catalano 2004; Destremau et al. 1976; ElAlami et al. 1996; Falleri 1994; Idrissi-Hassani and Lebreton 1992; Vendramin et al. 1998
Pinus pinea L.	L; U (native populations) and I (planted stands): land-use change, genetic pollution hazard	
Pistacia lentiscus L.	VC; I: genetic pollution hazard	NA
Pistacia terebinthus L.	C; S (but D in rural areas)	NA
Platanus orientalis L. (D)	L; D: habitat disturbance, genetic pollution hazard, parasitic attacks	NA
Populus alba L.	C; S: habitat disturbance	NA
Populus nigra L.	VC; S: habitat disturbance, genetic pollution hazard	NA
Populus tremula L.	R; U: global warming	NA
Pyrus communis L. (incl. *Pyrus pyraster*)	L; U: land-use change (local cultivars are disappearing)	A (cv.-rns)
Pyrus spinosa Forsskaal (incl. *Pyrus amygdaliformis*)	VC; I: land-use change	A (cv.-rns)
Quercus amplifolia Guss.	C; D: habitat disturbance, out-breeding difficulties(1), genetic pollution hazard, oak decline syndrome	Arena 1958; Brullo et al. 1999; Di Noto et al. 1995, 1998; Dumoulin-Lapègue et al. 1997; Fineschi et al. 1995a-b-c; Fineschi et al. 1998; Petit et al. 2002a-b; Ronsisvalle et al. 1984(2)

Table 1. continued.

Taxon	Presence/distribution, demographic trend and major threats	Reference(s)
Quercus cerris L.	L; D: global warming, habitat disturbance, parasitic attacks, out-breeding difficulties([1])	Brullo et al. 1999
Quercus coccifera L. s.l. (incl. *Quercus calliprinos* Webb)	R; D: land-use change, out-breeding difficulties([1])	La Mantia 1998
Quercus congesta C. Presl	L; U: habitat disturbance, genetic pollution hazard, oak decline syndrome	([2])
Quercus dalechampii Ten.	C; U: habitat disturbance, genetic pollution hazard, oak decline syndrome	([2])
***Quercus gussonei* (Borzì) Brullo**	C; U: habitat disturbance, oak decline syndrome	Brullo et al. 1999
Quercus ilex L.	VC; D: genetic pollution hazard	Burgarella et al. 2003; Fineschi et al. in press; Gramuglio et al. 1973; La Mantia 1999; La Mantia et al. 2003; Lumaret et al. 2002; Michaud et al. 1995; Toumi and Lumaret 2001
***Quercus leptobalanos* Guss.**	L; U: habitat disturbance, genetic pollution hazard, oak decline syndrome	([2])
Quercus petraea (Mattuschka) Liebl. subsp. *austrotyrrhenica* Brullo, Guarino et Siracusa	L; U: global warming, habitat disturbance, genetic pollution hazard	Brullo et al. 1999; Bruschi et al. 2003; Dumoulin-Lapègue et al. 1997; Fineschi et al. 1995a-b
Quercus suber L. s.l.	C; D: habitat disturbance, genetic pollution hazard, oak decline syndrome	Jiménez Sancho 2001; Toumi and Lumaret 1998
Quercus virgiliana (Ten.) Ten.	C; S: habitat disturbance, genetic pollution hazard, oak decline syndrome	([2])
Rhamnus alaternus L.	VC; U: habitat disturbance	NA
Rhamnus cathartica L.	L; U: habitat disturbance	NA
***Rhamnus lojaconoi* Raimondo**	ER: habitat disturbance	Raimondo 1979
Rhamnus lycioides L. subsp. *oleoides* (L.) Jahandiez et Maire	L; U: habitat disturbance	NA
Rhamnus saxatilis Jacq. subsp. *infectoria* (L.) P. Fourn.	R; U: habitat disturbance	NA
Rhus pentaphylla (Jacq.) Desf.	L; U: habitat disturbance	NA
Rhus tripartita (Ucria) Grande	L; U: habitat disturbance	NA

Table 1. continued.

Taxon	**Presence/distribution, demographic trend and major threats**	**Reference(s)**
Salix alba L. s.l.	VC; D: habitat disturbance	NA
Salix apennina A. Skvortsov	L; U: habitat disturbance	NA
Salix caprea L.	L; U: habitat disturbance	NA
Salix cinerea L.	L; U: habitat disturbance	NA
Salix fragilis L.	L; U: habitat disturbance	NA
Salix gussonei Brullo et Spampinato	L; U: habitat disturbance	Brullo and Spampinato 1988
Salix pedicellata Desf.	C; D: habitat disturbance	NA
Salix purpurea L.	C; D: habitat disturbance	NA
Sorbus aria (L.) Crantz s.l.	R; U: global warming, habitat disturbance	NA
Sorbus aucuparia L. subsp. *praermosa* (Guss.) Nyman	R; U: habitat disturbance	NA
Sorbus domestica L.(D)	C; S: land-use change (local cultivars are disappearing)	A (cv.-rns)
Sorbus torminalis (L.) Crantz	L; U: global warming, habitat disturbance	NA
Tamarix africana Poiret	VC; I: habitat disturbance	NA
Tamarix dalmatica Baum	L; U: habitat disturbance	NA
Tamarix parviflora DC.	L; U: habitat disturbance	NA
Tamarix gallica L.	C; U: habitat disturbance	NA
Taxus baccata L.	ER; S: global warming, habitat disturbance	NA
Tilia platyphyllos Scop.	ER; U: global warming, habitat disturbance	NA
Ulmus canescens Melville	L; D: habitat disturbance, parasitic attacks	Scialabba et al. 1997
Ulmus glabra Hudson	R; D: global warming, habitat disturbance, parasitic attacks	Scialabba et al. 1997
Ulmus minor Miller	C; D: habitat disturbance, parasitic attacks	Scialabba et al. 1997
***Zelkova sicula* Di Pasquale, Garfì et Quézel**	ER; D: global warming, habitat disturbance, low fitness	Fineschi et al. 2002; Nakagawa et al. 1998; Yu-Feil et al. 2001

[1] Very often represented by scattered and/or small-sized populations.
[2] Owing to the present disagreement upon the taxonomic treatment of the deciduous Sicilian taxa of the genus *Quercus*, it is actually unclear if all of them have been studied. For the references concerning the whole see *Q. amplifolia*.

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