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BIODIVERSITY ASSESSMENT AND METHODS OF INTEGRATION OF NFI DATA WITH REMOTE SENSING AND OTHER LARGE-SCALE DOCUMENTS.

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CONTENT

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BIODIVERSITY ASSESSMENT AND METHODS OF INTEGRATION OF NFI DATA WITH REMOTE SENSING AND OTHER LARGE-SCALE DOCUMENTS.

1. Definition of biodiversity

„*Today it is universally accepted that the conservation of biodiversity is essential for sustainable forest management*" (Ciancio and Nocentini 2004). In this sense, the basic question that needs to be answered is what one wants to conserve, and hence, what we understand under the term „biodiversity". Kaennel (1998) who made a thorough survey of literature focusing on biodiversity revealed a variety of formal and informal definitions of the term biodiversity.

Who is assessing biodiversity?

Already in 1990, Noss (1990) pointed out that „*biological diversity means different things to different people. To a systematist, it might be the list of species in some taxon or group of taxa. A geneticist may consider allelic diversity and heterozygosity...., whereas community ecologist is more interested in the variety and distribution of species and vegetation types.*"

Very often, the definition of biological diversity according to the Convention on Biological Diversity is cited. This document defines biodiversity as „*the variety and variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part*". This definition covers three fundamental components of diversity: genetic, species, and ecosystem diversity (Duelli 1997 in Larsson 2001, Merganič and Šmelko 2004). However, also this widely accepted definition like many others fails to mention ecological processes, such as natural disturbances, and nutrient cycles, etc., that are crucial to maintaining biodiversity (Noss 1990). The complexity of the understanding of the term biodiversity was well documented by Kaennel (1998).

Therefore, Noss (1990) suggested that for the assessment of the overall status of biodiversity more useful than a definition would be its characterisation that identifies its major components at several levels of organisation. Franklin et al. (1981) recognised three primary attributes of ecosystems: composition, structure, and function (in Noss 1990). "*Composition has to do with the identity and variety of elements in a collection, and includes species lists and measures of species diversity and genetic diversity. Structure is the physical organization or pattern of a system, from habitat complexity as measured within communities....Function involves ecological and evolutionary processes, including gene flow, disturbances, and nutrient cycling*" (Noss 1990). Commonly, compositional diversity was of interests and major concerns, while other (structural and functional) aspects received less attention (Franklin 1988 in Noss 1990). According to Larsson (2001), "*composition and structure determine and constitute the biodiversity of an area (Noss 1990), and are essential to the productivity and for forest ecosystem sustainability*", while functional diversity is defined as "*the diversity of ecological functions performed by different species, and/or the diversity of species performing a given ecological function*" (Larsson 2001).

What scale / level is of our concern?

All attributes of biodiversity can be monitored at multiple spatial scales. Noss (1990) recognised four hierarchical levels of organisations: genetic, species - population, ecosystem - community, and landscape. This multi-scaled concept of biodiversity has been realised and emphasised by a number of authors (Noss 1990, Larsson 2001, Humphrey and Watts 2004, Ciancio and Nocentini 2004, Estreguil et al. 2004, etc.) While applying this approach one should be aware of the fact that "*no single level of organization is fundamental, and different levels of resolution are appropriate for different questions*" (Noss 1990). From an operational forestry point of view, three scales must be considered: single tree, stand and landscape (Larsson 2001). "*However, this categorisation can be unhelpful as it ignores the effect of ecological processes operating across scales*" (Hansson 2001 in Humphrey and Watts 2004).

2. Biodiversity assessment

Due to the complexity of biodiversity and of forest ecosystems, „*complete assessments of biodiversity are not practically achievable*" (Humphrey and Watts 2004) because of "*impossibility of monitoring all taxa*"/features (Lindenmayer 1999). Therefore, "*means to reduce complexity are necessary*" (Christensen et al. 2004). „*Hence the search for reliable indicators or short-cut measures of biodiversity*" (Ferris and Humphrey 1999; Jonsson and Jonsell 1999; Noss 1999; Simberloff 1998 in Humphrey and Watts 2004). „*Because indicators, and the way they are derived, will change over time, it is most essential that any biodiversity assessment system is based on an enduring set of compositional, structural and functional characteristics*" (Allen et al. 2003). In addition, "a *complete long-term biodiversity strategy must take into account both interactions between the different geographical levels and the fact that different elements of biodiversity are dependent on different geographical scales, in different time perspectives*" (Larsson 2001).

2.1. Key factors

An interesting approach how to deal with the complexity of forest biodiversity has been applied in the BEAR project. Within the context of this project, key factors of forest biodiversity, i.e. factors that "*have a major influence on or directly reflect the variation in biodiversity within European forests*" (Larsson 2001), were identified according to different ecosystem components (i.e. composition, structure, function) and at different geographical scales (national/regional, landscape, stand scale). The identification of key factors and classification of scales has resulted in an operational tool for complex biodiversity assessment at multiple spatial scales, that is also applicable to practical biodiversity management and can be used as a basis for forest policy (Larsson 2001). This approach has been chosen because with the present knowledge it is possible to rather well identify the key factors, but their assessment through indicators needs further development and validation (Larsson 2001). Hence, according to this author, it is better "*to allow semi-qualitative assessments and a range of indicators and methods than omitting an important key factor*" (Larsson 2001). Nevertheless, from a long-term perspective, a standardised system of biodiversity indicators is needed.

2.2. Indicators

„*The use of indicators is one widely accepted way of achieving the reduction of complexity*" (Christensen et al. 2004). "*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 (eg. diversity) that are too difficult or expensive to measure for other species and communities*" (Williams and Gaston 1998; Landres et al. 1988 in Humphrey and Watts 2004). Hence, „*an indicator should constitute a good surrogate for biodiversity value*" (Rautjärvi et al. 2005, Stokland et al. 2004).

2.2.1. Requirements on indicators

„*Ideally, an indicator should be (1) sufficiently sensitive to provide an early warning of change; (2) distributed over a broad geographical are, or otherwise widely applicable; (3) capable of providing a continuous assessment over a wide range of stress; (4) relatively independent of sample size; (5) easy and cost-effective to measure, collect, assay, and/or calculate; (6) able to differentiate between natural cycles or trends and those induced by anthropogenic stress; and (7) relevant to ecologically significant phenomena*"(Noss 1990).

Similarly Ferris and Humphrey (1999) defined that the indicators must: „*be readily quantifiable, easily assessed in the field, repeatable and subject to minimal observer bias, cost effective, and ecologically meaningful (i.e. have a close association with, and identification of, the conditions and responses of other species)*“.

According to other authors, a good indicator should possess two main features: (1) it should be easy to inventory, (2) it should be strongly correlated with other species it is intended to represent (Ranius 2002, Humphrey and Watts 2004) and „*other entitites for which it is hypothesized to be indicative*“ (Stokland et al. 2004).

Of course, there is no single perfect indicator (Rainio and Niemela 2003) that will meet all the above stated requirements. Hence, a set of complementary indicators is needed (Noss 1990).

2.2.2. Categorisation of indicators

A variety of indicators and indicator systems have been proposed „*both at the EU level (MCPFE 2002) and within individual countries (e.g. Ministry of Agriculture and Fisheries 1994; Eeronbeimo et al. 1997)“* (Humphrey and Watts 2004). „*The nature of the indicators depends on who is using them and for what purpose*“ (Humphrey and Watts 2004).

Larsson (2001) has proposed a broad typology of indicators for assessment of biodiversity in European forests at a range of spatial scale. Estreguil et al. (2004) stated that there are three relevant sets of indicators for reporting biodiversity at European level: (1) „the improved Pan-European Indicators for Sustainable Forest Management“ from the Ministerial Conference on the Protection of Forests in Europe (MCPFE), (2) the key factors of forest biodiversity defined in the BEAR (Biodiversity Evaluation Tools for European Forests) Project, and (3) Indicators core set for Biodiversity and Nature Protection and for Terrestrial environment of the European Environmental Agency (EEA).

Christensen et al. 2004 divided indicators into (1) structural indicators, and (2) indicator species, while Noss 1990, Larsson 2001, Bradshaw and Møller 2004 distinguish (1) structural, (2) compositional, and (3) functional indicators based on the three aspects of biodiversity as defined by Frnaklin et al. (1981) (in Noss 1990).

2.2.2.1. Compositional indicators

“*Compositional diversity encompasses the identity and variety of elements in a collection or classification sytems, e.g. forest types and succession stages, species lists, the number of genes and allele variation within species*” (Stokland et al. 2003).

2.2.2.1.1. Indicator species

Species-based indicators represent a direct approach of biodiversity assessment (Christensen et al. 2004). That is probably the main reason why this method has often been used in biodiversity studies, although different species from different groups have been surveyed depending on the goal of the work. For example, for forest conservation purposes, bryophytes have been identified as useful indicators (Sim-Sim et al. 2004). Christensen et al. (2004) documented that „*fungal indicator species can add valuable information relevant for prioritisation in forest conservation*“.

On the basis a thorough analysis of the existing literature, Thormann 2006 stated that lichens are valuable indicators of forest health, but they have also been found as indicators of old-growth forests (Campbell and Freeden 2004, Motiejunaite et al. 2004, Uliczka 2003).

Arthropods have been suggested as indicators of sustainable forest management, although their use is problematic due to the difficulties in accurate species-level identification (Langor and Spence 2006). In fact, although this group encompasses the largest number of species in the world, only a few groups are relatively well-known, e.g. epigaeic carabid and staphylinid beetles and spiders, saproxylic beetles, butterflies and larger night flying moths (Langor and Spence 2006), while many species are still unknown. For example, of the insects as the largest group of arthropods, 50-90% of the existing species have still to be discovered (Thomas 2005).

Carabid beetles, as the best studied group of arthropods (Langor and Spence 2006), „*are frequently used to indicate habitat alternation*“ (Rainio and Niemela 2003). The beetle Osmoderma eremita is a useful indicator of stands with a rich beetle fauna in tree hollows, because it is easy to find and identify (Ranius 2002). According to Thomas 2005, „*butterflies are often the only group (from insects) for which accurate measures of change can be obtained*“. Based on his study the author concluded that „*butterflies represent adequate indicators of change for many terrestrial insect groups*“, as their extinction rates in Britain were found similar to those in a range of other insect groups over 100 years once recording bias is accounted for“. Nevertheless, the same author recommended that „*similar schemes be extended to other popular groups, especially dragonflies, bumblebees, hoverflies and ants*“ (Thomas 2005).

In European forests, invertebrates together with plants and fungi have been proposed as indicator species for assessing conservation value at the stand scale *(Roberge and Angelstam 2006).* At larger spatial scales, wide-ranging vertebrates, such as birds, can be used as indicators (Roberge and Angelstam 2006, Casanova and Memoli 2004), because they are abundant, widely distributed, and also because of *„a close connection between the overall biodiversity of an environment and the complexity of bird populations*“ (Casanova and Memoli 2004). Hence, birds are „*particularly useful indicators of the relations between animal communities and vegetation in forest environments*“ (Casanova and Memoli 2004). The advantage of using birds as indicators is that „*there are extensive databases on birds, they are easy to observe and they can be identified from their vocalizations*“ (Hoeven and Iongh 1999). From the many bird species, woodpeckers have been detected as good indicators of forest diversity (Virkkala 2006, Nilsson et al. 2001, Roberge and Angelstam 2006).

Since mammals occur in various types of environment, their use as indicators of biodiversity has been favoured (Casanova and Memoli 2004). The study of Azman (2001) showed that small-mammal population could be regarded as an indicator group for assessing impact of logging on a forest ecosystem. According to Casanova and Memoli (2004), from small mammals the best indicators of ecosystem functionality are carnivores, while Insectivores are sensitive indicators of the completeness of the alimentary chain, and rodents are useful in monitoring pollution. From the ungulates, “*the roedeer is a good indicator of the functionality of forest systems*” (Casanova and Memoli 2004).

“*The well-investigated vascular plants, which comprise approximately 300 000 species, are comparatively well suited as an indicator group in terrestrial habitats. Several examples show the good correlation of their diversity with overall diversity*” (Barthlott et al. 1998) or diversity of a particular group. For example, Schmit et al. (2005) presented that “*tree species richness can be used to predict macrofungal species richness*”. Within forest restoration processes, short-lived tree species (European mountain ash (Sorbus aucuparia L.), European white birch (Betula pendula Roth), Downy birch (B. pubescens Ehrh.), and Glossy buckthorn (Frangula alnus P. Mill.)) can serve as indicators of plant diversity (Kreyer and Zerbe 2006).

When examining species indicators, works that compare the performance of different taxa are particularly of high value. For example, Juutinen and Monkkonen 2004 studied beetles, birds, vascular plants, wood-inhabiting fungi, and a specified subgroup of assumed indicator species. Kati and Papaioannou 2001, and Kati et al. 2004 examined woody plants, aquatic and terrestrial herpetofauna, small terrestrial birds, orchids, and Orthoptera. All mentioned authors found that woody plants seem to be the best biodiversity indicators.

In contrast to taxonomical hierarchy, Noss 1999 identified several different groups of indicator species with regard to their requirements on area, dispersal, resources, ecological processes etc. In literature the concept of keystone species, umbrella species (e.g. Bollmann et al. 2004, Suchant and

Baritz 2001, Suchant 2001, Angelstam et al. 2000, Ranius 2002), focal species (Lambeck 1997), Red List species (*Schmidt et al.* 2006), threatened species (Martikainen and Kouki 2003), endemic species (Cassagne et al. 2006) is often used. Keystone species are defined as ecologically pivotal species whose impact on a community or ecosystem is large, and disproportionately large for their abundance (Noss 1999), and "*upon which a large part of the community depends*" (Noss 1991). Consequently, "*loss of keystone species produces cascade effects, i.e. the loss of other species or the disruption of processes*" (Larsson 2001). „*An umbrella species is a species which is so demanding that the protection of this species will automatically save many others*" (Ranius 2002).

Considering authenticity of community, Larsson (2001) proposed a category of alien (exotic, non-indigenous, introduced) species. These can be mainly found in disturbance corridors, from which they are spread to adjacent undisturbed habitats. "*In general, only a few introduced species survive in their new environment and eventually get naturalised without creating any problems*" (EEA 2006). However, successful exotic species „*may become a threat to indigenous species or to a whole ecosystem by disrupting the food chain or altering the habitat*" (EEA 2006).Usually, the species biodiversity connected to the alien species is lower, "*because of the time needed for example for invertebrates to adjust to the new species*" (Larsson 2001).

Table: Candidate indicator species

Species / Group of species	**Pros**	**Contras**
Fungi	ubiquitous indicator species for assessing conservation value	little known about fungal diversity, as they are usually cryptic and ephemeral
Plants	many are readily observable, easy to indetify its presence indicates appropriate habitat conditions for other species indicator species for assessing conservation value at the stand scale	some may not be attractive to public as a species of concern
vascular plants	comprise approx. 300 000 species good correlation of their diversity with overall diversity or diversity of a particular group (e.g. macrofungal diversity) woody plants seem to be the best biodiversity indicators	
Arthropods	the largest number of species in the world indicators of sustainable forest management	difficulties in accurate species-level identification only a few groups are relatively well-known (e.g. epigaeic carabid and staphylinid beetles, spiders, saproxylic beetles, butterflies, larger night flying moths)
Beetles		Finding threatened beetles requires very large sample sizes
Carabid beetles	the best studied group of arthropods indicate habitat alternation	crucial understanding of their relationship with other species is incomplete

beetle *Osmoderma eremita*	useful indicator of stands with a rich beetle fauna in tree hollows easy to find and identify umbrella species for the endangered beetle fauna in tree hollows	some beetles in tree hollows are more sensitive to habitat fragmentation than O. eremita, and may go extinct if only O. eremita is taken into consideration..
Insects	the largest group of arthropods	50-90% of the existing species have still to be discovered poor baseline knowledge most attempts to generalize involve large extrapolations from a few well-studied taxa
Butterflies	the only group (from insects) for which accurate measures of change can be obtained adequate indicators of change for many terrestrial insect groups	
VERTEBRATES	easily communicated and conspicuous species At larger spatial scales, wide-ranging vertebrates can be used as indicators, because they are abundant, widely distributed	
Amphibians, Reptiles	many require healthy environment and are sensitive to disturbances	some are difficult to monitor
Birds	abundant, widely distributed, well known and familiar species easy to observe, can be identified from their vocalizations extensive databases on birds a close connection between the overall biodiversity of an environment and the complexity of bird populations because of their mobility and large territory useful indicators of the relations between animal communities and vegetation in forest environments	some are migratory species some have been able to adapt to habitat loss and urban environment, some use built environments as its main nesting habitat
capercaillie *Tetrao urogallus*	an umbrella function for a rich mountain **forest** community **indicator** species for close-to-nature and structure-rich mixed mountain forests good surrogate for red-listed mountain forest bird species,	limited potential to indicate high species richness of beetles
woodpeckers	good indicators of forest diversity	
Mammals	Easily recognizable occur in various types of environment adapt themselves ethologically and physiologically to changes in the ecosystem around them, and hence, they are reliable indicators of the	feared by the general public and may not get support as an indicator species for an urban area (bear). Their populations fluctuate widely from year to year with no

	quality of forest habitats	apparent periodicity (vole) not often seen since they are nocturnal (bat) or living underground (vole) Many people are poorly educated about bats and have a bad impression of them.
small mammals	indicator group for assessing impact of logging on a forest ecosystem the best indicators of ecosystem functionality are carnivores, Insectivores are sensitive indicators of the completeness of the alimentary chain, rodents are useful in monitoring pollution	
Ungulates	roe deer is a good indicator of the functionality of forest systems	

Species / Group of species	**Pros**	**Contras**
Fungi	ubiquitous indicator for assessing conservation value	cryptic and ephemeral little information about fungal diversity
Plants	many are readily observable, easy to indetify its presence indicates appropriate habitat conditions for other species indicator species for assessing conservation value at the stand scale	some may not be attractive to public as a species of concern
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Arthropods	the largest number of species in the world indicators of sustainable forest management	difficulties in accurate species-level identification only a few groups are relatively well-known (e.g. beetles, spiders, butterflies)
Beetles	well-known	Finding threatened beetles requires very large sample sizes
Carabid beetles	the best studied group of arthropods indicate habitat alternation	incomplete crucial understanding of their relationship with other species
beetle *Osmoderma eremita*	easy to find and identify useful indicator of stands with a rich beetle fauna in tree hollows	some beetles in tree hollows are more sensitive to habitat fragmentation than O. eremita,

	umbrella species for the endangered beetle fauna in tree hollows	and may go extinct if only O. eremita is taken into consideration.
Insects	the largest group of arthropods	50-90% of the existing species have still to be discovered poor baseline knowledge most attempts to generalize involve large extrapolations from a few well-studied taxa
Butterflies	well-known and easily monitored thanks to their size and beauty adequate indicators of change for many terrestrial insect groups the only group (from insects) for which accurate measures of change can be obtained	
Amphibians, Reptiles	many require healthy environment and are sensitive to disturbances	some are difficult to monitor
Birds	abundant, widely distributed, well known and familiar species easy to observe, can be identified from their vocalizations extensive databases on birds close connection between the overall biodiversity of an environment and the complexity of bird populations because of their mobility and large territory useful indicators of the relations between animal communities and vegetation in forest environments	some are migratory species some have been able to adapt to habitat loss and urban environment, some use built environments as its main nesting habitat
capercaillie *Tetrao urogallus*	an umbrella function for a rich mountain forest community indicator for close-to-nature and structure-rich mixed mountain forests good surrogate for red-listed mountain forest bird species	limited potential to indicate high species richness of beetles
Mammals	Easily recognizable occur in various types of environment adapt themselves ethologically and physiologically to changes in the ecosystem around them, and hence, they are reliable indicators of the quality of forest habitats	some have adapted to urbanised environment (fox) feared by the general public and may not get support for an urban area (bear). some are not often seen since they are nocturnal (bat) or living underground (vole)
small mammals	indicator group for assessing impact of logging on a forest ecosystem carnivores are best indicators of ecosystem functionality Insectivores are sensitive indicators of the completeness of the alimentary	Their populations fluctuate widely from year to year with no apparent periodicity (e.g. vole)

	chain rodents are useful in monitoring pollution	
Weasel family *Mustelidae*	indicate the degree of naturalness	some have adapted to urbanised environment
Ungulates	roe deer is a good indicator of the functionality of forest systems	

2.2.2.1.1.1. Pros and contras of indicator species

According to Lindenmayer (1999), „*the concept of indicator species can make an important contribution to biodiversity conservation because of the impossibility of monitoring all taxa in species-rich forest environments*". Despite the fact that „*species have high potential, as such indicators are applicable to all ecosystems*" (Heer et al. 2004), this approach is not universally accepted (Williams and Gaston 1998 in Humphrey and Watts 2004), because "*indicator function is largely hypothetical*" (Gallego-Castillo and Finegan 2004). "*There is no implication of functional linkage amongst species groups, as the concept is essentially empirical*" (Humphrey and Watts 2004). For example, Bollmann et al. 2004 revealed that capercaillie (Tetrao urogallus), which is often implicitly attributed indicator function, is a good surrogate for red-listed mountain forest bird species, but its potential to indicate high species richness of beetles is limited. This example documents that "*the conservation of one targeted group does not guarantee the conservation of other groups as well*" (Kati and Papaioannou 2001). Hence, before selecting the indicator species there is a need to establish the link, and to test and validate the relationships between an indicator and the elements it is intended to indicate (Stokland et al. 2004, Humphrey and Watts 2004, Lindenmayer et al. 2000, Lindenmayer 1999). While amongst some species groups good correlations have been found (Sætersdal et al. 2003 in Humphrey and Watts 2004), in others they have not been proven (Johnson and Jonsell 1999 in Humphrey and Watts 2004). For example, Sverdrup-Thygeson (2001) found no significant correlations between wood-rooting fungus Fomitopsis pinicola and saproxylic beetles. "*Thus, species richness of one single species group is unlikely to be a good indicator for total biodiversity*" (Berglund and Jonsson 2001). Due to this, „*a single indicator group is not sufficient for decision-making processes in conservation*" (Medellin et al. 2000). Therefore, a multi-taxa approach is preferred (Kotze and Samways 1999), or otherwise "*the use of indicator species likely results in a loss of species*"(e.g. Juutinen and Monkkonen 2004). Nevertheless, "*their use appears promising as they alleviate communication among stakeholders*" (Angelstam et al. 2000) and enables the participation of public in monitoring, although in such cases it is recommended to use "*only few easily communicated and conspicuous species, e.g. vertebrates*" (Uliczka 2003), or bird species (Nally et al. 2004). The presence of indicator species on a particular site can also assist decision-makers and managers in e.g. assessing an area's conservation value (Uliczka 2003, Medellin et al. 2000) "*particularly if those species are at high risk of extinction or are considered ecologically, economically, or socially important*" (Noss 1999).

2.2.2.1.2. Forest types

Indicator species represent compositional indicators at stand scale. At higher level of organization, forest types are compositional indicators (Estreguil et al. 2004). They represent a very important set of habitat factors (Stokland et al. 2003).

Corona et al. 2004 cited the following definition of a forest type by Canadian Forest Service 1995: "*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*". "*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)*" (in Bradshaw and Møller

2004). At the European scale, several hundred to thousands of such stand types could be recognised (Bradshaw and Møller 2004). For example, "*Bohn et al. (2000) presented a map of 699 potential European vegetation types*" (Bradshaw and Møller 2004). After a comprehensive inventory of pan-European natural, semi-natural and anthropogenic habitats the European Nature Information System (EUNIS) (Barbati and Marchetti 2004) identified 377 forest types (Bradshaw and Møller 2004).

However, such a high number is difficult to monitor and map. Therefore, a system of higher order forest types that reflects broad-scale variation was required for biodiversity assessment (Bradshaw and Møller 2004). In the EU Habitats Directive (Annex 1, Council Directive 92/43/EEC) 51 forest types are selected (Bradshaw and Møller 2004) to be important for biodiversity conservation and hence, to be protected under Natura 2000 network (Barbati and Marchetti 2004). "*The BEAR project (Biodiversity Evaluation Tools for European Forests) proposed 33 forest types for biodiversity assessment (FTBAs). The classification system attempted to be scale independent and applicable at regional, landscape and stand scales*" (Bradshaw and Møller 2004). The FTBAs are a heterogeneous mixture of actual and potential forest types (Barbati and Marchetti 2004), and include also the types with a significant biodiversity value that are purely of cultural origin (hedgerow, coppice) (Bradshaw and Møller 2004). Based on this system, Barbati and Marchetti 2004, and Bradshaw and Møller 2004 proposed a simpler qualitative classification, which comprises 14 European forest types. "*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*" (Bradshaw and Møller 2004).

On the contrary, the quantitative approach to classification of forest types brings the advantages of objectivity and repeatability (Norddahl Kirsch and Bradshaw 2004). Norddahl Kirsch and Bradshaw 2004 analysed data gathered within NFIs and ICP Forest Level 1 data, and identified 17 and 16 forest types, respectively, with regard to actual tree species abundance. This study showed that "*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)*". Therefore, the authors suggest combining approaches if European forest biodiversity is to be faithfully described.
A comprehensive review of a scheme suggested by Barbati and Marchetti (2004) has resulted into a European Forest Types classification proposed for MCPFE reporting by Barbati et al. (2006, 2007). "*The process of revision has been based on a review of descriptions of actual and potential forest vegetation of Europe (Ozenda, 1994; Bohn et al., 2000) or of European forest regions (e.g. Mayer, 1984; Nordiska Ministerrådet, 1984; Ellenberg, 1996; Esseen et al., 1997; Quézel & Médail, 2003). The revision has been targeted to the following issues:*

(i) to ensure the European Forest Types being representative and comprehensive of the variety of forest conditions at pan-European level;

(ii) to ensure the criteria adopted to separate forest types being consistent with the purposes of MCPFE reporting" (Barbati et al. 2007).

The proposed classification system is hierarchical and is structured into 14 level I classes (Categories) and 76 level II classes (Types). The hierarchy follows the principle of increasing similarity in the natural conditions and level of anthropogenic modification. "*The category level is conceived to identify and reflect significant breaking points in the continuum of natural and anthropogenic factors affecting the state of European forests, as assessed by MCPFE indicators*". Types further describe the variety and the character of forest communities within each category in term of tree species composition, structural and floristic features (Barbati et al. 2007).

Regardless of the classification of forest types used in the biodiversity assessment, the benefit of forest typification is that forest types "*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*" (Corona et al. 2004). Forest types also enable comparison of ecologically similar forests, and are meaningful units for formulating policies and management regimes (Barbati et al. 2007). The maps of the forest types are powerful tools, which can be used to better understand spatial distribution of vegetation diversity (Rego et al. 2004). They could also "*give indication about where to intervene for re-establishing a more "natural" landscape biodiversity*" (Corona et al. 2004).

2.2.2.2. Structural indicators

"*Structural diversity refer to the physical organisation or pattern of a system, including the spatial patchwork of different physical conditions in a landscape, habitat mosaics, species assemblages of different plant and animal communities, and genetic composition of subpopulations*" (Stokland et al. 2003).

„*Structural indicators used to describe the conditions for forest biodiversity include vertical structure, age class distribution and the amount of dead wood" (Christensen et al. 2004). They 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*" (Christensen et al. 2004).

2.2.2.2.1. Deadwood

"*Dead and dying wood plays a key role in the functioning and productivity of forest ecosystems*" (Humphrey et al. 2004). "*Functionally, it represents an important component of the forest carbon pool*" (Stokland et al. 2004). However, "*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 Sherbune 1993; Sippola and Renvall 1999; Ferris et al. 2000)"* (in Montes et al. 2004), particularly „*for small vertebrates, cavity-nesting birds, and a host of lichens, bryophytes, polypores and other saproxylic fungi and invertebrates (Samuelsson et al. 1994; Esseen et al. 1997; Butler et al. 2001)*" (in Humphrey et al. 2004). „*In Scandinavia, it has been estimated that 6000—7000 species depend upon dead wood. This corresponds to about 25% of all forest species in the region*"(Stokland et al. 2004). Hence, deadwood is regarded as „*a key factor of biodiversity in the sense of species richness*" (Schuck et al. 2004, Ferris and Humphrey 1999) and as „*a key feature for the preservation of many threatened species*" (Ranius et al. 2003). It also acts as „*a surrogate for decomposition processes and habitat availability (Ferris and Humphrey 1999)*" (in Hahn and Christensen 2004). In addition, „*certain aspects of CWD are well known characteristics of old-growth forests (Siitonen et al. 2000)*" (in Hahn and Christensen 2004).

„*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)"* (in Humphrey et al. 2004).

In USA, deadwood or "*down woody material is an FIA (Forest Inventory and Analysis program of the USDA Forest Service) indicator that provides estimates of forest structural diversity, forest area fuel loadings, and national carbon sources*" (Woodall and Williams 2005).

"*In order to develop proper dead wood biodiversity indicators it is crucial to understand which qualities of dead wood are important to the wood-inhabiting species*" (Stokland et al. 2004). Several studies have pointed out that the mentioned roles of deadwood depend not only on its presence, but also on its amount that is accumulated in the forest ecosystem (Butler and Schlaepfer

2004). This feature has been accounted for in the MCPFE and BEAR deadwood indicators (Stokland et al. 2004). For the proper development and application of deadwood as an indicator, volume or biomass estimates from natural forests are taken as a reference (Hahn and Christensen 2004, Humphrey et al. 2004). "*However, the knowledge on 'natural' amounts of dead wood in European forests is fragmented with some forest types being intensively studied whereas others are sparsely researched*" (Hahn and Christensen 2004). Hahn and Christensen (2004) identified "*the lack of data from Southem Europe, which means that guidelines are only for north and central European forest types*".

As these authors pointed out, "*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*" (Hahn and Christensen 2004). According to Humphrey et al. (2004) "*coarse woody debris (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*". Although Travaglini et al. (2007) did not detect any significant differences in total deadwood volume among forest types, their analysis also showed the highest values of deadwood volume in central Europe, while the Mediterranean forests contain relatively little deadwood. "*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)*" (Hahn and Christensen 2004).

Stokland et al. (2004) presented five attributes of deadwood important to wood-inhabiting species: (1) type of deadwood, (2) tree species, (3) stage of decay, (4) dimension, (5) landscape patterns.

There exist different classifications of deadwood types: (i) standing vs. lying dead wood, (ii) coarse woody debris (CWD) vs. fine woody debris (FWD), (iii) snag, log, fallen branch, stump; that are interconnected. For example, standing dead wood are snags or stumps, which could belong either to CWD or FWD. Presently, the official indicators (MCPFE and BEAR) distinguish only between standing and lying deadwood. "*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 trees, whereas others, like fungi and mosses primarily utilise lying dead wood*" (Stokland et al. 2004).

However, as Stokland et al. 2004 found, diameter of dead trees is "*the quality that most species respond to*". In the studies dealing with deadwood it is customary to subdivide it into fine and coarse woody debris. "*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 Montes et al. 2004). "*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*" (Stokland et al. 2004). Hence, „*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*"(Montes et al. 2004).

The identification of tree species, or at least the distinction between coniferous and broadleaved wood is strongly recommended as different species are associated to these groups (Stokland et al. 2004). Similarly, „*the stage of decay is a very important quality for predicting the associated species composition*" (Stokland et al. 2004). Veerkamp (2003) found that „*the decaying stage of dead wood was the most important factor influencing the occurrence of decay fungi*".

„*Different dead wood qualities are not evenly distributed in the forest landscape. The variation is caused by landscape properties such as topography, soil conditions, productivity*" (Stokland et al. 2004). The disturbance factors causing decay, death, and the creation of deadwood in natural forests (drought, storm, fungal pathogens, insect disease, fire, mammals, natural thinnings) „*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)*“ (in Humphrey et al. 2004).

„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“ (Stokland et al. 2004). Complete dead wood inventories of dead wood were initiated in the National Forest Inventories (NFI) in Finland, Norway and Sweden during the 1990s. However, „due to the great variability within stands and across the landscape, field recording of deadwood is labour-intensive and expensive, if adequate sample sizes have to be ensured“ (Butler and Schlaepfer 2004b). Therefore, Butler and Schlaepfer 2004b presented a new method enabling efficient mapping and quantification of large snags by coupling colour infrared aerial photographs and a geographic information system (GIS).

Table: Factors influencing deadwood attributes

Factor	**Descritpion**
Location	Biogeographic region (European scale) Forest type
Climate	Temperature Humidity O_2 a CO_2 concentration
Soil	soil type
Topography	slope characteristics (e.g., slope aspect, position, and steepness).
Site productivity	
Species composition	
Disturbance	abiotic: drought, storm, wind, fire, slope failure (erosion, landslide), abiotic agents (acid rain) biotic: mammals, fungi, insects, disease (parasitic plants), natural thinning (suppression and competition), senescence
Human interventions	logging

Table: Deadwood attributes important to biodiversity

Attribute	**Description and Classification**
Amount	in m^3/ha or in % of living volume
Type of dead wood	(i) standing vs. lying dead wood (ii) coarse vs. fine woody debris (iii) snag, log, fallen branch, stump
Tree species	(i) species list (ii) coniferous vs. broadleaved
Dimension	diameter, length
Decay stage	defined by wood texture, shape, portion on ground, presence of twigs and bark, amount of wood fragmentation different classification systems with min. 3 classes from recently dead to almost decomposed
Decay rate	decomposition speed generally expressed through a constant *k* which indicates the percent mass, volume or density loss over time
Mortality type	(i) natural vs. management (ii) dry snag, mechanically broken, up-rooted, broken by rot, cut

	by beaver
Landscape pattern	spatial distribution e.g. scattered, clumped

2.2.2.2.2. Forest fragmentation

"*Landscape patterns represent the core of structural ecosystem diversity*" (Stokland et al. 2003) „*Structural indicators related to forest spatial pattern refer to the assessment of forest connectivity, forest fragmentation, forest isolation, edge/interior forest*" (Estreguil et al. 2004), while these patterns and their distributions have important implications for biodiversity conservation (Loyn and McAlpine 2001).

Landscape fragmentation has been identified as one of the fundamental reasons for the biodiversity loss (Roy and Sanjay-Tomar 2000). „*Habitat fragmentation is the breaking up of a large portion of a forested land into several smaller portions. The forest fragmentation can be explained in two phases. The first phase results in the reduction of total amount of forest areas whereas the second phase leads to the isolation of smaller patches.* (Laxmi-Goparaju et al. 2005)

„*Forest fragmentation is a critical aspect of the extent and distribution of ecological systems. Many forest species are adapted to either edge or interior habitats. When the degree or patterns of fragmentation change, it can affect habitat quality for the majority of mammal, reptile, bird, and amphibian species found in forest habitats (Fahrig, 2003)*" (in NCEA 2007). Due to this, "*international biodiversity agreements require assessing indicators of connectivity and fragmentation in forested ecosystems (e.g., MPLO, 2000; Malahide, 2004)*" (in Vogt et al. 2007).

There exists a number of works dealing with forest fragmentation and its effect on biodiversity (Riitters et al. 2002, Behera et al. 2005, Roy et al. 2005, Laxmi-Goparaju et al. 2005 etc.). However, „*currently, there are few tested and proven indicators for assessing and monitoring the forest fragmentation process*" (Loyn and McAlpine 2001), although the importance of landscape patterns as indicators have been recognised also in the MCPFE and BEAR systems. The methodology for their measuring and monitoring is poorly developed. Data from NFI field plots are inadequate for such purposes (Stokland et al. 2003). Holopainen et al. (2005) stated, „*the fragmentation of a forest area can be characterised by using some simple metrics of landscape, e.g. biotope areas, density, size, and variability*". "*McGarigal & Marks (1995) have documented that the patch density and mean patch size serve as fragmentation indices*" (Roy and Behera 2002). In this content, new technologies, such as remote sensing and GIS techniques seem promising as it was already documented by e.g. Laxmi-Goparaju et al. (2005), and Vogt et al. (2007). Using these tools, pattern and fragmentation processes are mostly measured with two approaches: (1) patch based metrics often calculated over a systematic fixed area grid from freeware such as Fragstats (McGarigal and Marks, 1995) and (2) area density scaling measures from the "amount adjacency" model based on image convolution (Riitters et al. 2002).

The USDA Forest Service developed an indicator of forest fragmentation using National Land Cover Data (NLCD). Before its calculation, the four NLCD forest cover classes (coniferous, deciduous, mixed, and wetland forest) are aggregated into one forest class and the remaining land cover classes into a non-forest class and a "missing" class consisting of water, ice/snow, and bare ground (Riitters, et al., 2002). A model classifies forest fragmentation based on the degree of forestland surrounding each forest pixel (a square approximately 30 meters on each edge) for various landscape sizes (known as "windows"). Three degrees of land cover were defined: (1)"core" if a subject pixel is surrounded by a completely forested landscape (no fragmentation); (2) "interior" if a subject pixel is surrounded by a landscape that is at least 90% forest; and (3) "connected" if a subject pixel is surrounded by a landscape that is at least 60% forest. Lanscape sizes range from 5.6 acres (a 5 by 5 pixel square) to 13,132 acres (a 243 by 243 pixel square) (Riitters et al. 2002, NCEA 2007).

A similar, though a more detailed classification of forest pattern was developed by Vogt et al. (2007b) who defined nine classes that cover a wide range of forest spatial pattern. According to these authors, "core" forest is the inner part of a forested region that is situated beyond a certain distance to forest boundary. In their works (Vogt et al. 2007a, 2007b), the center pixel is labeled core forest, if all 8 surrounding pixels are forest Hence, "core forest represents unfragmented habitat that is potentially suitable for interior forest species, while "patch" forests are isolated forest fragments where organisms are less likely to communicate with organisms outside the fragment" (Vogt et al., 2007b). The authors defined patch as a forested region that is too small to contain core forest. Edge is an exterior perimeter of core forest regions, i.e. „a transition zone between core forest and core nonforest".

Apart from the classes "core", "patch", and "edge", the authors also defined 'perforated' and 'connecting' features: corridors, shortcuts and branches, of which the branches could be viewed as 'broken connections.' The class 'perforated' refers to pixels of core forest that surround a nonforest patch, i.e. it is an interior perimeter or a transition zone between core forest and non-forested area ('holes') inside forests. 'Connector', i.e. corridor and shortcut, is a set forest pixels with no core forest, that connects at least two different core forests and connects to the same core forest unit, respectively. 'Branch' is defined as a set of forested pixels without core forest that is connected at one end only to non-core forested area, i.e. to a connector, edge or perforation (Vogt et al. 2007b). „*Corridors and shortcuts characterize potential movement pathways, and as relatively narrow features they may be vulnerable to future fragmentation and conversion to patch*"(Vogt et al. 2007). Corridors can be either structural, i.e. physical links between large forest regions, or functional defined by the movements of organisms. However, if corridors are to be beneficial for biodiversity conservation, they have to possess certain qualities (width, height, volume, maturity, sensu Hinsley and Bellamy, 2000) that meet the requirements of the most species (Bailey 2007). Vogt et al. (2007) presented a method for automated mapping of structural corridors with morphological image processing. As shown in their paper, „*the approach satisfies the assessment requirements of feasibility and repeatability when using continental-scale land-cover maps and it can be implemented at multiple scales*"(Vogt et al. 2007). The method was also applied in Estreguil et al. (2007).

Although habitat fragmentation per se is often considered a threat to biodiversity, "*biodiversity losses are most likely a result of the amount of regional habitat loss rather than fragmentation (Harrison, 1994; Fahrig, 1998, 2001, 2003; Harrison and Bruna, 1999; Rosenberg et al., 1997; Trzcinshi et al., 1999)*" (in Bailey 2007). The review of Bailey (2007) "*has indicated a lack of firm empirical evidence that species increase following attempts to increase connectivity in fragmented woods*". Hence, a scientifical demonstration of the benefits to biodiversity of increasing connectivity through the development of networks and corridors is required (Bennett, 2003 in Bailey 2007).

2.2.2.2.3. Forest edge

Forest edge is a specific type of ecotones that are defined as "*boundaries between different land use classes"* (Corona et al. 2004). „*As forest fragmentation increases beyond the fragmentation caused by natural disturbances, edge effects become more dominant, interior-adapted species are more likely to disappear, and edge- and open-field species are likely to increase*" (NCEA 2007). „*Forest plant and animal communities along fragmented forest edges can change with the introduction of exotic species (Jones and others 2000, Boulinier and others 2001, Pearson and Manuwal 2001)"* (in Riitters et al. 2002). The significance of forest edges in nature conservation was documented by e.g. Magura et al. (2001).

To quantify forest edge, edge measurements such as total edge length and edge density are used (Gallego et al. 2000, Riitters et al. 2004). Edge density defined as edge length per standard area is an „*indicator that does not depend on the size of the reference unit and can be computed directly for the administrative units to be compared*" (Gallego et al. 2000). Riitters et al. (2004) arranged these measures into the group of fragmentation indices, since fragmentation and forest edge are interrelated.

Traditionally, the length of edges/ecotones "*is determined by polygon delineation on the basis of visual interpretation of remotely sensed imagery (complete mapping) and subsequent perimeter mensuration on each delineated polygon (Haines and Chopping 1996). However, such a procedure might have omission and commission errors, unavoidable in image interpretation and classification by polygon delineation (Carfagna and Gallego 1999)*" (Corona et al. 2004). Therefore, Corona et al. (2004) proposed a forest ecotone survey procedure based on line intersect sampling that overcomes the above-mentioned shortcomings. "*Line intersect sampling (LIS) is an easy method for assessing the total length of a discrete population of land elements characterized by linear shapes, particularly when orthocorrected remotely sensed images are available (Corona 2000). LIS is a form of cluster sampling in which population elements crossed by a line transect are selected into the sample (Gregoire and Valentine 2003)*" (Corona et al. 2004). In Corona et al. (2004) "*ecotone length per unit area is estimated by visual interpretation of the changes from forest to other land use classes along each sampling line displaced on remotely sensed images from the land to be inventoried*". The authors found that this method reduces time needed for the estimation of ecotone length when compared with conventional forest polygon delineation and perimeter mensuration. According to Corona et al. (2004), "*the proposed procedure may also be used directly on the ground (on small areas) in the context of field surveys, e.g., by systematic selection of the sampling points (line centers) and randomly oriented line displacement with the help of a GPS device*".

Vogt et al. (2007a) detected forest edge in the proces of mapping spatial pattern using morphological image processing. This method was identified as a theory and technique for analysing the shape and form of objects (Soille 2003 in Vogt et al. 2007a). Forest pattern is clasified "by a sequence of logical operations such as union, intersection, complementation, and translation using geometric objects called 'structuring elements' (SE) of pre-defined shape and size" (Vogt et al. 2007a). In their work, Vogt et al. (2007a) consider two structuring elements SEs: an 8-neighbourhood (SE1) and a 4-neighbourhood (SE2), that pre-define which and how many pixels around the examined (center) pixel are accounted for in the analysis. In addition, the shape and dimension of SEs also define the direction and extent of the morphological operations. The authors used two operations: the 'erosion' that shrinks regions of forest and the 'dilation' that expands them.

The detection of forest edge starts from the forest-nonforest map. First, the nonforest patches are identified and removed by erosion using SE1. Then, the actual nonforest area is dilated in all directions using SE1 (nie SE2???). Dilations are repeated until there is no difference in the area clasification between two consecutive dilations. Forest edge consists of all forest pixels that are adjacent to non-forest area after subtracting forest patch pixels (Vogt et al. 2007a).

Image convolution is another approach used for detecting forest edge. This method uses a mowing window device of a predefined size to identify forest pattern. "*A moving window operates by moving a fixed-area window over the map so as to place a support region around each pixel. Measurements are made at each placement of the window, and the values are assigned to the location of the pixel at the center of the support region*" (Riitters, 2005).

Various forest species react differently to edge effects. The 100m width of the border classes (edge, connector, branch, perforation) corresponds to edge effects for many interior species (e.g. birds in Forman and Alexander 1998) and can be regarded as a permeability distance for invasive species. In Canadian woodland survey, an edge width is defined between 100 and 300m. Hence, it is useful when the edge width can be predefined for the analysis, i.e. edge width can be of one or more pixels. Both above-mentioned techniques enable to determine the desired width of the edge by defining the size of structural elements SE and the window dimension in the case of morphological and convolution approach, respectively. Vogt et al. (2007a) analysed the behaviour of the two approaches using various edge widths. They found that "*with increasing SE or window size, both methods increase the width of the perforated and edge regions at the expense of the core regions…*

The comparison of both methods revealed that the morphological approach is more accurate at the pixel level…Small patch regions remain patch regions and stay disconnected to neighboring core forest regions, and continuous forest boundaries are labeled as a single class“ (Vogt et al., 2007a).

2.2.2.3. Functional indicators

“*Functional diversity involves processes or temporal change, including disturbance events and subsequent succession seres, nutrient recycling, population dynamics within species, various forms of species interactions, and gene flow*” (Stokland et al. 2003).

From a functional point of view, species can be subdivided in categories like primary producers, herbivores, predators, and decomposers (Stokland et al. 2003). Belaoussoff et al. (2003) defined a functional group *„as a group of not necessarily related species exploiting a common resource base in a similar fashion. Within a functional group there is greater similarity in ecological resource requirements than within a guild, thereby implying that there is a greater degree of interspecific competition (Arthur 1984; Colwell and Winkler 1984). There is an overlap in resource requirements between species in a functional group. Disturbances would affect those species evenly by disrupting resources that they all use*“ (Belaoussoff et al. 2003).

The BEAR-project strongly recommends to include functional indicators in any Biodiversity Evaluation Tool. Within the framework of the BEAR project, fire, wind and snow, and biological disturbance have been identified as the most important functional key factors in the group of “natural influences”, while the area affected by a particular factor are suggested as possible indicators with high ecological significance (BEAR Newsletter 3).

Although “e*cosystem function in many cases might be more important than species diversity in gaining an understanding of ecosystem dynamics*” (Sobek and Zak 2003), “*structural and compositional indicators are considered to be more tractable for end-users* (Angelstam et al. 2001)” (in Humphrey and Watts 2004)

2.2.3. Pros and contras of indicators

„*It is an underlying assumption that biodiversity indicators predict the forest biodiversity*“ (Stokland et al. 2004). In fact, many of the proposed indicators need to be tested and require rigorous validation in order to be interpreted (Corona and Marchetti 2007). Failing and Gregory (2003) identified „*10 common ‘mistakes’ in developing and using forest biodiversity indicators from the standpoint of making better forest management choices. The mistakes relate to a failure to clarify the values-basis for indicator selection and a failure to integrate science and values to design indicators that are concise, relevant and meaningful to decision makers*“.

One common mistake in the construction and application of biodiversity indicators is mixing means and ends (Failing and Gregory 2003). For example, „*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. 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*“ (Stokland et al. 2004). In this context, „*it is crucial to establish the link between different indicator states and the biodiversity component it is intended to indicate*“ (Stokland et al. 2004) in order to to get an idea what the consequences of managing an indicator might be for dependant flora and fauna (Humphrey et al. 2004).

In addition, if the application of indicators is to be useful for monitoring trends in biodiversity it is necessary to ascribe quantitative values to them (Humphrey et al. 2004). However, „*data on indicator states alone does not say very much unless they are put into perspective*“ (Stokland et al. 2004). Thus, e.g. Humphrey et al. 2004 suggested a range of possible values for each examined measure (e.g. deadwood). The upper limit of the range can be defined by so called `natural reference values` (Stokland et al. 2004), i.e. the values from natural or virgin stands. However, for some areas such information is missing or occurring only sparsely (Hahn and Christensen 2004). Hence, where the information from such forests is unavailable, the values from ´best´ examples are used (Humphrey et al. 2004). The lower range limit for the different measures is more difficult to define, since there is very little information available on threshold values for sustaining key populations of species (Humphrey et al. 2004). A similar way how to determine the desired biological state of a forest indicator is to use baseline values together with some measure of variability under natural conditions (Ghazoul 2001). Nevertheless, when applying these values, one must bear in mind that „*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*“ (Ciancio and Nocentini 2004). In addition, due to high variability of natural conditions and anthropogenic influences within the world forest area, the values of biodiversity indicators are meaningful only if they refer to specific environment (Barbati et al. 2007, Travaglini et al. 2007). In this context, Barbati et al. (2007) suggested to use soundly ecologically based forest types classifications, e.g. European Forest Types classification proposed by Barbati et al. (2006, 2007).

Considering the policy making, Failing and Gregory (2003) note that „many of the proposed indicators remain cumbersome for managers to work with and, by sheer number, retain some of the drawbacks of the 'listing' approach. For example, although measurements of 'predation rates' or 'nutrient cycling rates' (listed under the function category at the community/ecosystem scale) may be useful information to a scientist trying to understand ecosystem processes and define hypotheses, they do not inform a stakeholder or decision maker (or, we suspect, most scientists) about the current status of biodiversity. Nor do such comprehensive listings provide a useful means for discriminating among policy alternatives that affect biodiversity. From the perspective of forest managers, a useful approach seems to be a combination of 2-3 compositional and 2-3 structural indicators, while the compositional indicators should be functionally linked to a broad range of other species (e.g. the extent and species composition of the broadleaved component in conifer forests); and the structural surrogates should act as surrogates for general species richness or diversity (e.g. the quantity and quality of deadwood) (Ferris and Humphrey 1999).

Since „*decision makers need a concise summary of biodiversity implications of a proposed policy, so that they can compare them with other bottom-line impacts and make informed choices about the inevitable trade-offs*“ (Failing and Gregory 2003), the authors propose weighting of indicators to reflect their importance to biodiversity and to construct a summary indicator or an index.

2.3. Biodiversity indices

Biodiversity indices are measures that quantify diversity using different statistical and mathematical approaches.

2.3.1. Requirements on indices

„*A useful biodiversity index should be flexible enough to enable the use of different biodiversity indicators for different ecosystems and spatial scales. It should be possible to calculate the index at different geographic locations and scales: for a particular project footprint, a larger landscape or ecoregion, an ecosystem type, a province or state, or the nation as a whole. And it should be scalable: that is, it should be possible to aggregate across regions at different scales and subsequently disaggregate in order to diagnose the source of major trends or unmask hidden trends. This approach allows both the presentation of a simple summary metric that can be used for communicating major trends and for making trade-offs with other social, economic or environmental objectives, and as well provides a basis for appropriate management action in response to observed trends*“ (Failing and Gregory 2003).

2.3.2.1. Indices characterising one component of biodiversity

2.3.2.1.1. Diversity indices

"*A great number of different methods can be used for the evaluation of species diversity (e.g. see Krebs 1989, Ludwig and Reynolds 1988). All of the proposed methods are usually based on at least one of the following three characteristics (Bruciamacchie 1996):*

* *species richness – the oldest and the simplest understanding of species diversity expressed as a number of species in the community (Krebs 1989);*
* *species evenness – a measure of the equality in species composition in a community;*
* *species heterogeneity – a characteristic encompassing both species abundance and evenness.*

The most popular methods for measurement and quantification of species diversity are species diversity indices. During the historical development, the indices have been split into three categories: indices of species richness, species evenness and species diversity (Krebs 1989, Ludwig and Reynolds 1988). The indices of each group explain only one of the above-mentioned components of species diversity" (Merganič and Šmelko 2004).

2.3.2.1.1.1. Richness indices

The term species richness was introduced by McIntosh (1967) to describe the number of species in the community (Krebs 1989). Surely, the number of species S in the community is the basic measure of species richness, defined by Hill (1973) as diversity number of 0^{th} order, i.e. N0. "*The basic measurement problem is that it is often not possible to enumerate all of the species in a natural community*" (Krebs 1989). In addition, S depends on the sample size and the time spent searching, its use as a comparative index is limited (Yapp 1979). „*Hence, a number of indices have been proposed to measure species richness that are independent of the sample size. They are based on the relationship between S and the total number of individuals observed*" (Ludwig and Reynolds 1988). Two such well-known indices are R1 and R2 proposed by Margalef (1958) and Menhinick (1964), respectively. Hubálek (2000), who examined the behaviour of 24 measures of species diversity in a data from bird censuses, assigned to the category of species richness-like indices also the index α (Fischer et al. 1943, Pielou 1969), Q (Kempton and Taylor 1976, 1978), and R500 (Sanders 1968, Hurlbert 1971).

2.3.2.1.1.2. Heterogeneity (diversity) indices

This concept of diversity was introduced by Simpson (1949) and combines species richness and evenness. "*The term heterogeneity was first applied to this concept by Good (1953), and for many ecologists this concept is synonymous with diversity (Hurlbert 1971)*" (in Krebs 1989). "*There are, literally, an infinite number of diversity indices (Peet 1974)*" (in Ludwig and Reynolds 1988). To investigate how communities are structured, two statistical distributions have been commonly fitted to species abundance data: logarithmic series and lognormal distribution. Due to the complexity of these statistical distributions and the lack of a theoretical justification, nonparametric measures of heterogeneity have been developed that assume no statistical distribution (Krebs 1989). Simpson proposed the first heterogeneity index λ, which gives the probability that two individuals picked at random from the community belong to the same species. It means if the calculated probability is high, then the diversity of the community is low (Ludwig and Reynolds 1988). "*To convert this probability to a measure of diversity, most workers have suggested using the complement of Simpson´s original measure*", i.e. 1-λ (Krebs 1989).

Probably the most widely used heterogeneity index is the Shannon index H‘ (or Shannon-Wiener function), which is based on information theory (Shannon and Weaver 1949). It is a measure of the average degree of "uncertainty" in predicting to what species an individual chosen at random from a community will belong (Ludwig and Reynolds 1988). Hence, if H´ = 0, then there is only one species in the community, whereas H´ is maximum (= log(S)) if all species present in the community are represented by the same number of individuals. Shannon index places most weight on the rare species in the sample, while Simpson index on the common species (Krebs 1989).

From other heterogeneity measures we mention Brillouin Index H (Brillouin 1956), which was first proposed by Margalef (1958) as a measure of diversity. This index is preferred being applied for data in a finite collection rather than H´. However, if the number of individuals is large, H and H´ are nearly identical (Krebs 1989). The indices N1 and N2 from Hill`s family of diversity numbers (Hill 1973), which characterise the number of "abundant", and "very abundant" species, respectively, also belong to diversity measures. "The McIntosh index is based on the representation of a sample in an S-dimensional hyperspace, where each dimension refers to the abundancy of a particular species" (Bruciamacchie 1996). According to the evaluation performed by Hubálek (2000), NMS "number of moves per specimen" proposed by (Fager 1972), $H´_{adj}$, which is an adjusted H´ by the d(H) correction (Hutcheson 1970), and R100 (Sanders 1968, Hurlbert 1971) can also be regarded as heterogeneity indices.

2.3.2.1.1.3. Evenness (equitability) indices

Lloyd and Ghelardi (1964) were the first who came with idea to measure the evenness component of diversity separately (Krebs 1989). "*Evenness measures attempt to quantify the unequal representation of species against a hypothetical community in which all species are equally common. The most common approach has been to scale one of the hetereogeneity measures relative to its maximal value when each species in the sample is represented by the same number of indivduals*" (Krebs 1989). Ludwig and Reynolds (1988) present five evenness indices E1 (Pielou 1975, 1977), E2 (Sheldon 1969), E3 (Heip 1974), E4 (Hill 1973), and E5 (Alatalo 1981), each of which may be expressed as a ratio of Hill´s numbers. The most common index E1, also known as J' suggested by Pielou (1975, 1977) expresses H' relative to maximum value of H' (= log S). Index E2 is an exponentiated form of E1. Based on the analysis of Hubálek (2000), McIntosh`s diversity D (McIntosh 1967, Pielou 1969), McIntosh`s evenness DE (Pielou 1969), index J of Pielou (1969) and G of Molinari (1989), are also evenness measures.

2.3.2.1.1.4. Complex diversity indices

On the contrary to species diversity indices that describe only one of the biodiversity components, the model BIODIVERSS proposed by Merganič and Šmelko (2004) estimates the species diversity degree of a stand from 5 diversity indices (R1, R2, λ, H' and E1) and thus integrates all the partial biodiversity components. The fundamental method of the model BIODIVERSS is a predictive discriminant analysis (StatSoft Inc. 1996, Huberty 1994, Cooley and Lohnes 1971). Using four discriminant equations, each for one species diversity degree, an evaluated forest stand is classified into one of the four pre-defined species diversity degrees. The method is based on the assumption, that if high species diversity is observed on a small area within the forest stand, we can presume that the species diversity of the whole examined forest stand will also be high. The probability of correct classification of the species diversity degree using the model BIODIVERSS is relatively high. Having only 1.5% sampling intensity, the success of classification already reaches approximately 90%. Although the model BIODIVERSS was designed for the determination of biological species diversity of the tree layer on a forest stand scale, the method can also be applied to regional or large-scale inventories if we assume that species diversity index determined on a sample plot represents a certain part of the evaluated area (Merganič and Šmelko 2004).

2.3.2.1.2. Structural indices

Structural diversity is defined as the composition of biotic and abiotic components in forest ecosystems (LEXER et al. 2000), specific arrangement of the components in the system (GADOW 1999) or as their positioning and mixture (HEUPLER 1982 in LÜBBERS 1999). According to ZENNER (1999) the structure can be characterised horizontally, i.e. the spatial distribution of the individuals, and vertically in their height differentiation. GADOW & HUI (1999) define the structure as spatial distribution, mixture and differentiation of the trees in a forest ecosystem.

To describe the structure and its components, „*the classical stand description (qualitative description of stand closure, mixture, density, etc.) and different graphical methods (diameter distribution, stand height distribution curve, tree map, etc.) can be very useful. However, they may not be sufficient to describe stand structure in detail since subtle differences will often not be revealed*“ (Kint et al. 1999). Therefore, a number of quantitative methods have been proposed. Partial reviews can be found in Pielou (1977), Gleichmar and Gerold (1998), Kint et al. (1999) Pielou (1977), Füldner (1995), Gleichmar and Gerold (1998), Kint et al. (1999), Lübbers (1999), Gadow and Hui (1999), Neumann and Starlinger (2001), Pommerening (2002) etc.

2.3.2.1.2.1. Indices characterising horizontal structure

„*The indices for spatial distribution or horizontal structure compare a hypothetical distribution with the real situation*“ (Neumann and Starlinger 2001). Probably the most well-known index is the aggregation index R proposed by Clark & Evans (1954) that describes the horizontal tree distribution pattern (or spacing as named by Clark & Evans (1954), or positioning as defined by Gadow & Hui (1999)). It is a measure of the degree to which a forest stand deviates from the Poisson forest, where all individuals are distributed randomly (Tomppo 1986). It is the ratio of the observed mean distance to the expected mean distance when individuals were randomly distributed. A similar measure is the Pielou index of nonrandomness (Pielou 1959), which quantifies the spatial distribution of trees by the average minimum distance from random points to the nearest tree (Neumann and Starlinger 2001). The Cox index of clumping (Strand 1953, Cox 1971) is the ratio of variance to mean stem number on sub-plots. Gadow et al. (1998) proposed an index of neighbourhood pattern based on the heading angle to four next trees. Another commonly used measures of horizontal structure are indices proposed by Hopkins (1954), and Prodan (1961), and methods by Köhler (1951) and Kotar (1993) (in Lübbers 1999).

According to Gadow & Hui (1999), mixture is another component of structure. For the quantification of mixing of two tree species, Pielou (1977) proposed the segregation index based on the nearest neighbour method like the index A of Clark & Evans, while the calculated ratio is between the observed and expected number of mixed pairs under random conditions. Another commonly used index is the index DM (from German Durchmischung) of Gadow (1993) adjusted by Füldner (1995). On the contrary to the segregation index, DM accounts for multiple neighbours (Gadow 1993 used 3 neighbours) and is not restricted to the mixture of two species (Kint et al. 1999).

Differentation is the third component of structure (Gadow & Hui 1999), that describes the relative changes of dimensions between the neighbouring individuals (Kint et al. 1999). Gadow (1993) proposed the differentiation index T, which is an average of the ratios of the smallest over the largest circumference calculated for each tree and its n nearest neighbours. Instead of the circumference, diameter at breast height can be used in this index to describe the horizontal differentiation as presented by Pommerening (2002). Values of the index T close to 0 indicate stands with low differentiation, since neighbouring trees are of similar size. Aguirre et al. (1998) and Pommerening (2002) suggested scales of five or four categories of differentiation, respectively.

2.3.2.1.2.2. Indices characterising vertical structure

While there are many indices that measure horizontal structure, there are only few for vertical structure (Neumann and Starlinger 2001). "*Simple measures such as the number of vegetation layers within a plot can be used as an index of vertical differentiation*" (Ferris-Kaan and Patterson 1992 in KINT et al. 1999). The index A developed by Pretzsch (1996, 1998) for the vertical species profile is based on the Shannon index H'. In comparison with H' the index A considers species portions separatelly for a predefined number of height layers (Pretzsch distinguished 3 layers). The index proposed by Ferris-Kaan et al. (1998) takes the cover per layer into account, but needs special field assessments (Neumann and Starlinger 2001). Therefore, using the same principles as Pretzsch (1996), i.e. Shannon index and stratification into height layers, Neumann and Starlinger (2001) suggested an index of vertical evenness VE that characterises the vertical distribution of coverage within a stand. The differentiation index T of Gadow (1993) is also applicable for the description of vertical differentiation, if the index is calculated from tree heights.

2.3.2.1.2.3. Complex structural indices

Complex structural indices encompass several components of structural diversity. For example, Jaehne & Dohrenbusch (1997) proposed the Stand Diversity Index that combines the variation of species composition, vertical structure, spatial distribution of individuals and crown differentiations. The Complexity Index by Holdridge (1967) is calculated by multiplying four traditional measures of stand description dominant height, basal area, number of trees and number of species. Hence, this index „*contains no information on spatial distribution nor aaccounts for within stand variation*" (Neumann and Starlinger 2000). Zenner (1999), and Zenner and Hibbs (2000) developed the Structural Complexity Index, that is based on the vertical gradient differenced between the tree attributes and the distances between the neighbouring trees. „*When all trees in a stand have the same height, the value for SCI is equal to one, the lower limit of SCI*" (Zenner and Hibbs 2000).

2.3.2.2. Complex indices combining more components of biodiversity

An example of a complex index is LLNS index proposed by Lähde et al. (1999). The index was suggested for calculating within-stand diversity using the following indicator variables: stem distribution of live trees by tree species, basal area of growing stock, volume of standing and fallen dead trees by tree species, occurrence of special trees (number and significance), relative density of undergrowth, and volume of charred wood. The LLNS index is calculated as the sum of diversity indices describing particular components (i.e. living trees, dead standing trees etc.). However, the index can also be applied during the field work, as the authors developed a scoring table for the indicator variables. The final value of LLNS is then obtained by adding all the scores together. The evaluation of this index using Finnish NFI data revealed, that the LLNS index differentiates even-sized and uneven-sized stand structures, the development classes of forest stands and site-types fairly well (Lähde et al. (1999).

Another complex index named as Habitat Index HI was also developed in Findland by Rautjärvi et al. (2005). The authors also use the name habitat index model as it was produced as spatial oriented model. The inputs in the model come from thematic maps from Multi-source NFI (MS-NFI) (predicted volume of growing stock, predicted stand age, and predicted potential productivity) and kriging interpolation maps from NFI plot data (volume of dead wood, and a measure for naturalness of a stand). The input variables were selected based on the forest biodiversity studies in Scandinavia. The index is of additive form where all input layers contribute to the result layer. All input variables (layers) are reclassified and enter the model as discreet variables, while each input layer is assigned a different weight according to its importance to biodiversity.

Meersschaut and Vandekerkhove (1998) developed a stand-scale forest biodiversity index based on available data from forest inventory. The index combines four major aspects of a forest ecosystem biodiversity: forest structure, woody and herbaceous layer composition, and deadwood. Each aspect consists of a set of indicators, e.g. forest structure is defined by canopy closure, stand age, number of stories, and spatial tree species mixture. The indicators are given a score determined on the basis of a common agreement. The biodiversity index is calculated as the sum of all scores, while its maximum value is set to 100.

2.3.2.3. Pros and contras of indices

Indices are regarded useful because they provide rapid, and easily calculated, ecological measures (Belaoussoff et al. 2003). Since they quantify biodiversity with a single summary statistics (Hubálek 2000, Merganič 2001), it makes comparisons between samples, communities and similar studies which use the same indices possible (Hubálek 2000, Belaoussoff et al. 2003). The indices can be used „*to determine quantitative critical values which need to be exceeded to ensure a minimal amount of biodiversity*“ (Pommerening 2002). Moreover, quantitative values cam be easily transformed into qualitative evaluation (low, medium etc. biodiversity degree), which is easier to understand outside the scientist community (Merganič 2001). “*Another important benefit of a forest biodiversity index is that it would facilitate learning over time*“ (Failing and Gregory 2003). As indices belong to non-parametric methods, their use eliminates some theoretical problems of application of parametric methods (Krebs 1989). In addition, usually their calculation is a simple procedure, that does not require large material supply and technical facilities (Merganič 2001).

The major criticism of biodiversity indices is that a number of them are only statistical artifacts and do not have any intrinsic biological meaning (Belaoussoff et al. 2003). In particular, this refers to diversity indices, since they attempt to combine, and hence to confound a number of variables that characterise community structure. In addition, the same diversity index value can be obtained for a community with low richness and high evenness as for a community with high richness and low evenness. This adds to the interpretation problem, making comparisons difficult and confusing (Ludwig and Reynolds 1988). Nevertheless, this is not the case of all indices. Values of some indices, e.g. Hill`s family of diversity numbers, can be easily ecologically interpreted (Ludwig and Reynolds 1988).

Regarding the interpretation, Belaoussoff 2000 showed that „*the use of different indices with the same data can result in different conclusions*“ (in Belaoussoff et al. 2003).

Another shortcoming of species diversity indices is that they are strongly correlated to plot size or an area of the evaluated forest stand. Therefore, it is suggested that if the biodiversity are to be appropriately presented, plot size should also be given (Merganič et al. 2004, Eckmüllner 1998).

A general problem with evenness measures is that they assume the total number of species in the whole community is known (Pielou 1969 in Krebs 1989). Since the observed species number in the sample is often less than true species numbers in the community (mainly in species-rich communities), the evenness ratios are usually overestimated (Sheldon 1969 in Krebs 1989). This is particularly true for the indices E1, E2, and E3 (Ludwig and Reynolds 1988). On the other hand, E4 and E5 are relatively unaffected by species richness and sampling variations, and hence tend to be independent of sample size (Ludwig and Reynolds 1988). In addition, an evenness index should be independent of the number of species in the sample, i.e. its value should stay constant regardless of the number of species present. However, as Peet (1974) and Ludwig and Reynolds (1988) showed, the indices E1, E2, and E3 are very sensitive to species richness, while the addition of one rare species to a sample containing only a few species greatly changes their values.

A thorough study of the performance of 24 species diversity indices based on 20 criteria was presented by Hubálek (2000), who for the estimation of species diversity within community suggested to use Fager´s `number of moves per spicemen`, exponential Shannon`s information, reciprocal Simpson`s lambda, and species richness (number of species).

3. INTERNATIONAL PROJECTS AND PROGRAMMES DEVOTED TO BIODIVERSITY

The increasing biodiversity awareness has resulted in a number of activities of scientific community. Various national and international projects have dealt with the biodiversity issue. Below we briefly describe some of them.

3.1 BioAssess (Biodiversity Assessment Tools)

„*The BioAssess (Biodiversity Assessment Tools) project was the first project to use standardised protocols to measure several major elements of biodiversity across Europe (in eight countries and six biogeographical regions) to simultaneously develop methods for assessing biodiversity, or "biodiversity assessment tools", and to quantify the impact on biodiversity of land use change, a major driver of change in biodiversity in Europe and elsewhere*“ (http://www.nbu.ac.uk/bioassess/).
„*The main purpose of the BioAssess project was to develop biodiversity assessment tools for inland terrestrial ecosystems, comprising sets of indicators of biodiversity, to assess the impact of policies on changes in biodiversity in Europe. "Biodiversity assessment tools" may be defined as a set of indicators, which provides information on status and trends in biodiversity for a range of stakeholders. This approach to monitoring acknowledges that a single measure of biodiversity is unlikely to satisfy most stakeholder needs, particularly those interested in trends in biodiversity at the European level*“ (http://www.nbu.ac.uk/bioassess/).
In the frame of the 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). 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 X 1 km and covered the same gradient from relatively natural forests to intensively managed agricultural areas. 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" (Koch and Ivits 2004).

3.2 BEAR (Biodiversiry Evaluation Tools for European forests)

„*Bringing together 27 partners of 18 European countries, the BEAR-project aimed at identifying a common scheme of key factors of forest biodiversity, to identify and describe a set of Forest Types for Biodiversity Assessment (FTBAs) and to define lists of indicators of biodiversity across European Forest Types. Making use of the expertise in the group, the BEAR-project found one of it central tasks to cover a wide biogeographic area and viewing biodiversity across Europe at several hierachical scales*“ (BEAR Newsletter 3).
„*The main achivements of the BEAR-project are:*
• *PRESENTATION OF A COMMON SCHEME OF KEY FACTORS OF BIODIVERSITY APPLICABLE TO EUROPEAN FORESTS.*

Key factors affecting or determining biodiversity include abiotic, biotic and anthropogenic factors that directly or indirectly influence biodiversity and its major components.
• IDENTIFYING EUROPEAN-LEVEL FOREST TYPES FOR BIODIVERSITY ASSESSMENT FTBAS.
The relative importance of key factors vary between different European forests as do the factors themselves, e.g. the species composition. The BEAR experts recommend that the management of biodiversity is to be based upon specific Forest Types for Biodiversity Assessment (FTBAs) defined as forest types which are uniquely influenced by a set of key factors of forest biodiversity. FTBAs do not reflect the proportions of tree species only, but reflect the whole composition and characteristics of the forest ecosystem in accordance to the geological, climatical and other biogeographical conditions of the area where they appear naturally.
• INDICATORS OF FOREST BIODIVERSITY.
During the BEAR-project the experts have agreed on that it is premature to define priority lists of indicators for operational use. This view was proposed and accepted as the current EU position. However, the BEAR-project has presented a gross list of potential biodiversity indicators to assess each key factor of forest biodiversity.
• RECOMMENDATIONS FOR ELABORATING BIODIVERSITY EVALUATION TOOLS and establishing schemes of biodiversity indicators for assessment of forest biodiversity on a European level“ (BEAR Newsletter 3).

3.3 ForestBiota (Forest Biodiversity Test Phase Assessments)

Under the ICP Forests Working Group on Biodiversity assessments the ForestBIOTA (Forest Biodiversity Test Phase Assessments) project was launched (ICP Forests 2003). ForestBIOTA was a joint action of 11 European countries that was carried out on 123 existing intensive monitoring (Level II) plots. It aimed *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.
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)" (Haußmann and Fischer 2004).
Specifically harmonized methods for forest biodiversity assessments have been proposed *„by further development and test wise implementation of monitoring methods for 1) forest type classification, 2) stand structure assessments, 3) deadwood assessments, 4) extended grond vegetation surveys and 5) epiphytic lichen monitoring*“ (Stofer 2006).

3.4. ALTER-Net (A Long-Term Biodiversity, Ecosystem and Research Network)

ALTER-Net is a five year project funded by the European Union's Framework VI programme, that began in April 2004. "*It is integrating capacity across Europe to assess and forecast changes in biodiversity, structure, functions and dynamics of ecosystems and their services*". The project involves 24 partner institutes from 17 European countries with the aim to build lasting integration of biodiversity research, monitoring and communication capacity. This is being achieved in a number of ways. "*In 2005 ALTER-Net launched the International Press Centre for Biodiversity Research (IPCB), a regularly updated online source of news and press releases about international biodiversity research, serving journalists and other users*". In 2006 and 2007, ALTER-Net ran summer schools aimed at equipping young researchers with the knowledge and skills to undertake integrated biodiversity research at a European level. Within the framework of ALTER-Net, two networks of sites are being developed: Long-Term Ecosystem Research sites (LTER) for European long-term terrestrial and freshwater biodiversity and ecosystem research, and Long-Term Socio-Ecological Research sites (LTSER), which could be used to determine the socio-economic implications of, and public attitudes to, biodiversity loss. Since the beginning of the project, a number of different reports have been published online dealing with various subjects related to biodiversity, e.g. its assessment, conservation, modeling and forecasting, but also different socio-economic and policy issues (www.alter-net.info).

3.5. SEBI 2010 (Streamlining European 2010 Biodiversity Indicators)

SEBI is joint pan-European activity with countries and other interested bodies to develop and implement biodiversity indicators for assessing, reporting on and communicating achievement of the 2010 target to halt biodiversity loss. The SEBI 2010 process consists of a coordination team and six expert groups. The coordination team is led by the European Environment Agency (and its European Topic Centre on Biological Diversity), ECNC (European Centre for Nature Conservation) and NEP-WCMC (World Conservation Monitoring Centre). The main tasks are to review, test, refine, document and help produce specific indicators in line with the 16 headline biodiversity indicators that have been agreed within the European Union and PEBLDS (Pan-European Biological and Landscape Diversity Strategy). The project was set up to coordinate activities in this field from national to pan-European level. The first completed output of the project is an initial set of indicators available at EU and pan-European levels to be published during 2007. (http://biodiversity-chm.eea.europa.eu).

3.6. DIVERSITAS

"*DIVERSITAS is an international, non-governmental programme with a dual mission:*

- *to promote an integrative biodiversity science, linking biological, ecological and social disciplines in an effort to produce socially relevant new knowledge; and*
- *to provide the scientific basis for the conservation and sustainable use of biodiversity.*

*DIVERSITAS achieves these goals by synthesizing existing scientific knowledge, identifying gaps and emerging issues, and promoting new research initiatives, while also building bridges across countries and disciplines. The Programme also investigates policy implications of biodiversity science, and communicates these to policy fora, including international conventions" (*www.diversitas-international.org*).*

DIVERSITAS was established in 1991, with the goal of developing an international umbrella programme that would address the complex scientific questions posed by the loss of and change in global biodiversity. In 2001, the main task of the Programme became a development of an international framework for biodiversity research. At the end of 2002, the published Science Plan identified three interrelated areas for further development. Currently, four DIVERSITAS Core Projects are identified, while each of them covers an important aspect of biodiversity science:

"• bioGENESIS aims to facilitate the development of new strategies and tools for documenting biodiversity, to understand the dynamics of diversification, and to make use of evolutionary biology to understand anthropogenic impacts;

• bioDISCOVERY focuses on developing a scientific framework to investigate the current extent of biodiversity, monitor its changes and predict its future changes;

• ecoSERVICES explores the link between biodiversity and the ecosystem functions and services that support human well-being and seeks to determine human responses to changes in ecosystem services;

• bioSUSTAINABILITY concerns itself with the science-policy interface, looking for ways to support the conservation and sustainable use of biological resources" (DIVERSITAS 2007).

In addition, DIVERSITAS has identified four topics that merit investigation on all levels represented by its Core Projects, namely mountain, freshwater and agricultural ecosystems and the problem of invasive species. Based on these topics, four Cross-cutting Networks have been established: The Global Mountain Biodiversity Assessment (GMBA) and the Global Invasive Species Programme (GISP), agroBIODIVERSITY and freshwaterBIODIVERSITY.

Both DIVERSITAS Core Projects and Cross-cutting Networks get implemented by International Project Offices, that help to build links to existing research institutes and programmes (www.diversitas-international.org).

More information about these and other activities related to biodiversity can be found online on several web sites: e.g. the European Community Biodiversity Clearing House Mechanism (EC-CHM) site (http://biodiversity-chm.eea.europa.eu) managed by the European Environment Agency. This site contains "the Bioplatform RTD catalogue", which is a database about scientists, organisations and networks performing work in the area of biodiversity, and aiming at stimulating contacts between biodiversity scientists and end-users of research results

Another site full of up-to-date data is CORDIS (i.e. Community Research and Development Information Service) devoted to European research and development and the exploitation of the results of European research (http://cordis.europa.eu).

4. Data sources used in biodiversity assessment

4.1. National Forest Inventories (NFI)

„*For an overview, the use of forest resource inventory data can be a cost effective method to obtain information for large areas*" (Söderberg and Fridman 1998), because "*forest inventories are a major source of information at least for traditional variables describing tree species composition, age, etc.*" (Estreguil et al. 2004). According to Corona and Marchetti (2007), "*NFIs have the advantage of providing objective information on key components of forest ecosystems, characterized in terms of amount, spatial and temporal distribution, and interaction on a multiple scale*". Data from NFIs are also usefull for biodiversity assessment. For example, for the biodiversity indicators that are related to forest composition, mainly in terms of species richness and the presence of species of particular conservation concern (threatened or endemic species), data from NFIs and other survey types can be used "*to generate species lists, which can be cross-referenced to national and international assessments of species status such as Red Lists and the appendices to the Convention on International Trade in Endangered Species of Wild Fauna and Flora*" (Corona and Marchetti 2007).

Basically, NFIs provide us with the information about the following themes: 1) forest area and land cover, 2) resource management (growing stock and the balance between increment and felling), 3) forestry methods and land use (felling systems, regeneration methods, road network density, specific methods such as ditching of swamp forests and soil scarification), 4) forest dynamics with regard to different disturbance factors (fire, storm, insect, browsing, 5) forest state (tree species composition, age distribution, dimension of living trees, tree mortality and deadwood), and partly also about 6) conservation measures, i.e. protected forest areas (Stokland et al. 2003).

Newton & Kapos (2002) distinguishes eight groups of biodiversity indicators appropriate for implementation at the forest management unit level: forest area by type, and successional stage relative to land area; protected forest area by type, successional stage and protection category relative to total forest area; degree of fragmentation of forest types; rate of conversion of forest cover (by type) to other uses; area and percentage of forests affected by anthropogenic and natural disturbance; complexity and heterogeneity of forest structure; number of forest-dependent species; and conservation status of forest dependent species (in Corona and Marchetti 2007).

„*The aim of the traditional National Forest Inventory (NFI) was to describe the main features of forests in terms of size, condition and change. But it was more concerned with their productive features than an extensive description of the forests*" (Rego et al. 2004). Only recently, variables more related to biodiversity have been introduced to NFIs (Söderberg and Fridman 1998). For example, „*recognition of the ecological importance of decaying wood has led to the incorporation of quantitative measures of deadwood in NFIs (e.g. Fridman and Walheim 2000)*“ (Humphrey et al. 2004). "*Integration of ecological assessment in the NFIs is a challenging task, in order to: (i) report on the status of forest ecosystems as per national and international requirements, (ii) assess the effectiveness of management, and (iii) improve our knowledge of ecosystem dynamics to design effective management systems*" (Corona and Marchetti 2007).

An increasing demand for information on non-productive functions of forests has caused that NFIs "*are developing as more comprehensive natural resources surveys, broadening their scope in two major directions (Kleinn et al., 2001): (i) including additional variables (such as biodiversity attributes), and (ii) expanding the target population towards non-traditional objects, like non-wood forest products and trees outside forests*"(Corona and Marchetti 2007).

However, "*in many cases simply adding some new attributes to existing lists of attributes and depending on traditional, established FI approaches fails to satisfy current information needs. To support the increasing demand for additional information on forest resource attributes (Corona et al., 2003), FIs must basically provide (Gillis, 2001): updated information; data types with uniform definitions, collected to the same quality standards; data that reflects consistent and complete area coverage; data suitable for accurate assessment of trend (change)*" (Corona and Marchetti 2007).

In addition, as Stokland et al. 2003 pointed out „*NFI field plots are inadequate for measuring landscape patterns*“ of structural ecosystem diversity because of the small plot size. In addition, in many cases precision guidelines for the estimates of many variables cannot be satisfied due to budgetary constraints and natural variability among plots, as McRoberts et al. (2005) stated for the USA. In neither of the cases it is efficient to increase the plot size or their number. Instead, other data sources, e.g. remote sensing, can be used more efficiently. In addition, „*projects in natural sciences face growing demand for rapid data generation, which results in the increasing application of integrated geoinformatics technologies such as: Digital Photogrammetry; Geographical Information Systems (GIS), Digital Elevation Model (DEM), Global Positioning System (GPS) or Remote Sensing (Gallaun et al. 2004, Kias et al. 2004)*“ (Wezyk et al. 2005). Field work itself „*has been enhanced by satellite positioning systems (GPS), automatic measuring devices, field computers and wireless data transfer*“ (Holopainen et al. 2005). „*The geoinformatic techniques make 3D spatial analyses possible, but if supplemented by the time factor (4D analyses), they allow determining the dynamics of changes within the natural environment (Agouris and Stefanidis 1996, Nayr and Reinhardt 1996, Armenakis and Regan 1996)*“ (Wezyk et al. 2005).

4.2. Specific monitoring programmes

For special purposes, specific monitoring programmes are needed. These programmes attempt to investigate particular features of a forest ecosystem that are of specific interest and their monitoring is not included within NFIs. Many of such surveys have been performed by NGOs (Heer et al. 2004). Although this kind of information can be of high value at a local or national scale, its applicability at a higher level (region, Europe) is restricted and requires pre-processing of data with regard to their quality, and biases and gaps in time and space (Heer et al. 2004).

"*Besides the species trend data which are collected by the NGOs, specific forest monitoring programmes are aiso increasingly collecting biodiversity trend data*" (Heer et al. 2004).

"*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 estabhished 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. 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"*(Haußmann and Fischer 2004). "*The strengths of the programme are its well established transnational monitoring and reporting infrastructure based on a common legal basis*" (Haußmann and Fischer 2004).

*"The strength of the Level I network is its representativity and the vast extent of its approximately 6000 permanent plots, arranged in a 16 x 16 km grid, throughout Europe" (*Haußmann and Fischer 2004). However, its value for representative biodiversity information is limited (Packalen and Maltamo 2001 in Haußmann and Fischer 2004) since in many cases only main tree species are assessed (Haußmann and Fischer 2004).

"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" (Haußmann and Fischer 2004). "*The monitoring programme of ICP forests has included the monitoring of ground vegetation in the Level II plots since 1996 (EC-IJNJECE 2002)*" (Heer et al. 2004).

4.3. Remote sensing

Remote sensing represents a powerful and useful tool for biodiversity assessment at the ecosystem and landscape level (Ghayyas-Ahmad 2001, Innes and Koch 1998, Foody and Cutler 2003). „*Modern remote sensing provides cost efficient spatial digital data which is both spatially and spectrally more accurate than before*" (Holopainen et al. 2005). Moreover, „*remote sensing technology can provide the kind of inormation that was previously not available to forestry management departments or which was not available on a scale appropariate for comprehensive analyses and planning projects*" (Schardt et al. 2005).

Remote sensing data have been successfully used for:

1. landscape characterisation (Roy and Sanjay-Tomar 2000, Laxmi-Goparaju et al. 2005 – forest fragmentation, Kozak et al. 2007 – identification forest and non-forest land)
2. habitat categorisation and estimation of their changes over large areas (Brotherton 1983; Cushman and Wallin 2000 in Humphrey and Watts 2004)
3. classification and mapping of land cover types (Ozdemir et al. 2005)
4. estimation of forest characteristics (Reese et al. 2003, Tuominen and Haakana 2005, Maltamo et al. 2006 - stand volumes of standing trees, Ingram et al. 2005 - stem density and basal area)
5. measuring vegetation (forest) structure (Maltamo et al. 2005, Ingram et al. 2005, Prasad et al. 1998, Wack and Oliveira 2005)
6. analysis of canopy surface and canopy gaps (Nuske and Nieschulze 2005)
7. identification of dead standing trees (Butler and Schlaepfer 2004) and estimation of their amount (Uuttera and Hyppanen 1998)
8. stratification for ground inventory (Roy and Sanjay-Tomar 2000, Ghayyas-Ahmad 2001, Jha et al. 1997) or to increase the precision of estimates (McRoberts et al. 2003, 2005; Olsson et al. 2005)

According to Innes and Koch 1998, „*remote sensing provides the most efficient tool available for determining landscape-scale elements of forest biodiversity, such as the relative proportion of matrix and patches and their physical arrangement. At intermediate scales, remote sensing* *provides an ideal tool for evaluating the presence of corridors and the nature of edges. At the stand scale, remote sensing technologies are likely to deliver an increasing amount of information about the structural attributes of forest stands, such as the nature of the canopy surface, the presence of layering within the canopy and presence of coarse woody debris on the forest floor*."

Nagendra (2001) who evaluated the potential of remote sensing for assessing species diversity distinguished three types of studies:

1. direct mapping of individuals and associations of single species,
2. habitat mapping using remotely sensed data, and prediction of species distribution based on habitat requirements,
3. establishment of direct relationships between spectral radiance values recorded from remote sensors and species distribution patterns recorded from field observations.

„*Direct mapping is applicable over smaller extents, for detailed information on the distribution of certain canopy tree species or associations. Estimations of relationships between spectral values and species distributions may be useful for the limited purpose of indicating areas with higher levels of species diversity, and can be applied over spatial extents of hundreds of square kilometres. Habitat maps appear most capable of providing information on the distributions of large numbers of species in a wider variety of habitat types. This is strongly limited by variation in species composition, and best applied over limited spatial extents of tens of square kilometres*" (Nagendra 2001).

Turner et al. (2003) recognise two general approaches to the remote sensing of biodiversity. „*One is the direct remote sensing of individual organisms, species assemblages, or ecological commu-nities from airborne or satellite sensors. … The other approach is the indirect remote sensing of biodiversity through reliance on environmental parameters as proxies*" (Turner et al. 2003), that can be clearly identified remotely.

„*Many different forms of remote sensing are available. Recently, interest has increased in laser scanner and synthetic aperture radar data, although most work to date has used photographs and digital optical imagery, primarily from airborne and space-borne platforms*" (Innes and Koch 1998). The utility of different remote sensing data for assessment purposes of various forest characteristics has been excessively studied in a number of works, e.g. Hyyppa et al. 2000, Lefsky et al. 2001, Tuominen and Haakana 2005, Thompson and Whitehead 1992.

4.4. Geographical Information Systems GIS

Geographical Information Systems also have a high potential in biodiversity assessment (Ghayyas-Ahmad 2001, Mironga 2004, Wallerman 2003). "*GIS provides the way to overlay different 'layers' of data: the ecological conditions, the actual vegetation physiognomy and human pressure indices (e.g., as deduced from the density of population or road network). It helps to assess disturbance levels; the spatial distribution of several species in order to determine biodiversity hotspots; past and present maps for monitoring land cover and land use changes. It provides possibilities to extrapolate observations e.g., to automatically define and map the potential area of a given species and to com-pare it with the locations where, it has been actually observed; or to combine different data sets for defining the potential list of species for a given forest type. GIS provides a database structure for efficiently storing and managing ecosystem-related data over large regions. It enables aggregation and disaggregation of data between regional, landscape and plot scales. It also assists in location of study plots and/or ecologically sensitive areas. GIS supports spatial statistical analysis of ecological distributions. It improves remote sensing information extraction capabilities, and provides input data/parameters for ecosystem modelling. The data generated through ground truthing and inte-gration of related attributes when used in GIS application result into significant features of biodi-versity and genetic resources*" (Roy and Behera 2002).

Mironga 2004 defined „*seven functions of GIS that are important for biodiversity modelling are database structure, ground surveys, spatial and statistical analysis, remote sensing integration, terrain analysis, data integration and data visualisation. Terrain analysis can be used to identify micro, meso and macroterrain indices. Data integration can be used to determine the environmental characteristics of known habitats of species. Data visualisation uses maps, graphs and statistics to process and facilitate understanding of the enormous amount of data that can be derived from the analysis of a species' habitat.*"

4.5. Combination

4.5.1. Why to combine different data sources?

The combination of different data sources is advantageous from several aspects. Probably the most important fact is that it reduces costs of data acquisition (Schmidt et al. 2005) and data processing in comparison to purely field-based methods. In addition, as McRoberts et al. 2005, Olsson et al. 2005 and others showed, the combination of satellite imagery and NFI at the same time increases precision of estimates of inventory variables (e.g. forest area, volume) as well as their changes. Integrating modern technologies can also be time efficient. An important aspect of the combination of field data and modern tools is that it enables spatial depiction of forest resources (Moisen and Edwards 1999, McRoberts 2005). In addition, since the use of aerial imagery is an old and widely used remote sensing technique, retrospective analyses can be accomplished (Nuske and Nieschulze 2005).

4.5.2. How are different data sources combined?

The combination of field data and data obtained from other sources, e.g. GIS, RS, can be performed in different ways depending on the purpose as shown below.

(1) Identification of the areas for ground survey

Prior to fieldwork, remote sensing data can be used to determine which plots are to be visited. Several NFIs use forest/nonforest classification of RS data. For example, in Canada`s NFI ground plots are only established in forested locations, which are determined from aerial photographs (Omule and Gillis 2005). In addition, the accessibility of the plots is also examined on RS images. While in Canada, "*inaccessible plot locations are replaced with suitable subjectively selected matches, and difficult-access plot location are subsampled*" (Omule and Gillis 2005), in the US Forest Inventory field crews visit only the plots determined from aerial photography to have accessible forest land (McRoberts et al. 2005).

(2) Increase precision of estimates

The precision of estimates of many variables can be enhanced using ancillary data. "*One source of ancillary data used by the US Forest Inventory and Analysis (FIA) program is classified satellite imagery in the form of land cover maps. These maps are used to derive strata which are then used with stratified estimation*" (McRoberts et al. 2005).

Stratified estimation requires that (1) the plots are assigned to strata, and (2) that the proportion of land area for each stratum is calculated. "*The first task is accomplished by assigning plots to strata on the basis of the stratum assignments of their associated pixels.*" The proportion of pixels in strata is calculated from the number of pixels with centers in each stratum. McRoberts et al. 2001 recommended creating four strata: forest F, non-forest NF, forest edge FE, non-forest edge NFE (see McRoberts et al. 2005). Similar post-stratification on the kNN dataset with the aim to improve estimates of variables of interest, e.g. tree species composition, have been implemented in the

Swedish NFI from 2005 onwards, while their results showed that 3-5 strata should be used (Olsson et al. 2005).

Another possibility to increase precision of estimated attributes is to derive the regression or ratio between the information obtained from RS data and those collected in the field. This is the principle of two-phase sampling, while RS data represent the first phase and ground data the second phase. Scheer and Sitko 2005 applied this method for the estimation of timber growing stock by combining field data with IKONOS satellite data, while different spectral signatures were used as auxiliary variables to derive their relationship (spectral reflectance models) with timber stock Holopainen et al. 2005 proposed a similar method based on two-phase sampling for the inventory of drought damages.

(3) Regionalisation of information

Field data is supplemented by RS data in order to characterise certain areas, e.g. state, regions, municipalities and forestry holdings (Tomppo 1996, Kangas 1996 in Katila 2005), or even forest stands (Tomppo 1987, in Katila 2005). For this purpose, "*some prediction method is needed in order to transfer information gathered in the field to locations of the image not corresponding to field plots. One of the most successful prediction methods in this context is the non-parametric k-nearest neighbours (k-NN) method*" (Koistinen et al. 2005). This method has been found as one of the most used and efficient non-parametric classification procedures, which are needed for modelling the complex relationships between spectral signatures and forest attributes (Maselli et al. 2003). The principle of the k-NN technique is that each pixel is assigned a vector of forest variables interpolated from the k spectrally nearest field plots (Olsson et al. 2005). "*This technique is a non-parametric approach to predicting values of point variables and nearest neighbor techniques with resulting root mean on the basis of similarity in a covariate space between the point and other points with observed values of the variables*" (McRoberts et al. 2002). "*It provides wall-to-wall maps of forest attributes, retains the natural variation found in the field inventory (unlike many parametric algorithms), and provides precise and localized estimates in common metrics across large areas and various ownerships*" (Finley et al. 2005).

Besides the k-NN method, there are several other possible smoothing and nonparametric regression methods, e.g. kernel smoothers, orthogonal series estimators (e.g. wavelet estimators), spline smoothers, local polynomial regression, or a nonparametric Bayesian regression method (Koistinen et al. 2005).

Koistinen et al. 2005 tested the applicability of local linear regression (LLR) and of its special case local linear ridge regression (LLRR). Based on their results, they concluded that LLR is not suitable for regional prediction with data from NFI and two adjacent Landsat 5 (tm) images due to statistical reasons, since its pixel-level bias deviated significantly from zero. In addition, when applying LLR, they encountered „*numerical problems due to singularities which arose from the discrete nature of the feature vectors.*" Although LLRR is better suited, it did not seem to have a clear advantage over the 5-NN (i.e. k=5) method. The authors suggest that further work is needed to determine the advantages of applying LLRR, mainly for smaller target regions.

Another way of regionalising forest attributes is to utilize the principles of geostatistics. "*The concept of geostatistics is quite simple: it arises from the assumption that near sample data are more related than distant ones. The analysis of data spatial variability and autocorrelation in geographical space (domain) is the basis for all geostatistical approaches*" (Scheer and Sitko 2005). "*Several geostatistic tools exist to measure the spatial variability. The best known tool is the semivariogram, also called variogram*" (Buddenbaum et al. 2005), which is a discrete function of spatial variability depending on the distance between the sample plots. The experimental variogram and fitted variogram models are then used for interpolation in a spatial domain employing kriging methods, from which ordinary kriging is the most widely used method. The advantage of geostatistical interpolation is that it provides us with the information about the estimation error

(Scheer and Sitko 2005). An example of such a regionalisation of tree species diversity degree from regional forest inventory data was presented by Merganič et al. 2004. Similarly, by combining ground and satellite IKONOS data Scheer and Sitko 2005 regionalised timber growing stock. Wallerman 2003 examined the applicacility of different kriging methods as well as the Bayesian state-space model for the prediction of stem volume per hectare using Landsat TM and field sampled data. The author compared the prediction accuracy of the analysed methods with Ordinary Least Squares regression using only RS data. Based on his results, ordinary and simple kriging methods seem promising because of the large reduction of root mean square error and othe rpractical aspects. Although the Bayesian state-space model did not provide improve prediction, Wallerman 2003 this model may be of use in higher complexity modeling.

(4) Identification and mapping of specific areas

A number of works presented how RS can help in identification, mapping, and monitoring of areas of specific interest. For example, Kozak et al. (2007) quantified changes of forest cover in the Carpathians using Landsat images for the years 1987 and 2000, while the single-date forest-non-forest maps were derived with the help of ancillary data, i.e. CORINE Land Cover and the Shuttle Radar Topography Mission digital elevation model. In urban and peri-urban areas in Sweden colour infrared aerial photographs have been interpreted in stereo models to obtain spatial and temporal information on biodiversity necessary for spatial planning for biodiversity (Groom et al. 2006). Groom et al. (2006) also presented that RS data can be used for mapping and monitoring disturbances in mountain vegetation cover. "*Fires can be monitored and analysed over large areas in a timely and cost-effective manner by using satellite sensor imagery in combination with spatial analysis as provided by Geographical Information Systems*" (Sunar and Özkan 2001).
Fuller et al. (1998) combined field surveys of plants and animals with satellite remote sensing of broad vegetation types in order to map biodiversity. Using a statistical classifier, they produced a land cover map from satellite images (Landsat TM), on which 14 land-cover classes were identified. Validation of this clasification recorded 86% correspondence between field and map data. „*The species data were used to generate biodiversity ratings, based on species `richness' and `rarity', which could be related to the vegetation cover. This inter-relationhelped to generate a biodiversity map of the Sango Bay area which has since been used to aid conservation planning*" (Fuller et al. 1998). „*Debinski et al. (1999) had used remotely sensed data and GIS to categorize habitats, and then determined the relationship between remotely sensed habitat categorizations and species distribution patterns.*" (Roy and Behera 2002).
According to Nagendra (2001), direct estimations of spectral radiance values may be useful for indicating areas with different levels of species diversity. While this technique „*provides indicators for further data collection on the ground, relationships between spectral values and species diversity may have to be calculated afresh for each image, thereby reducing its generality*" (Nagendra 2001).

(5) Prediction of species distribution

„*Remote sensing, geographic information systems (GIS) and spatial modelling in combination make a rapid tool for converting species observations to predictions of current or future species ranges based on environmental surrogates… When analyzing species distributions varying spatial scales, different environmental surrogate variables are considered important. General climatic variables are utilised for defining global scale niches of species and habitats (Peterson & Vieglais 2001, Pearson & Dawson 2003,) while topographic and geologic variables are taken into account at more limited, regional scales (Thuiller et al. 2003). At landscape scales, patterns and dynamism of land cover (Griffiths & Lee 2001), vegetation (Austin 1999) or land use are regarded (Dale et al. 2000, Vellend 2004)*" (Toivonen 2005).

Luoto et al. (2002) predicted plant species richness using generalized linear modeling (multiple regression models), environmental variables derived from Landsat TM images and topographic data extracted from a digital elevation model. According to this study, "*utilization of satellite imagery and GIS to study and predict vascular plant species rich-ness shows promise for revealing distributional patterns that might not otherwise be apparent*". Multiple regression models have also been used to study avian species richness, while climatic variables (precipitation, temperature, radiation etc.), topography, ecosystem diversity estimated as the number of ecosystems per quadrat from a map of global ecosystems, and latitude were used as independent variables (Rahbek and Graves 2001).

Accurate, remote sensing based habitat maps, in conjunction with detailed information on species requirements on habitat, can generally be used to model the distribution of species (Nagendra 2001). „*In a detailed analysis in the tropical forests of the Western Ghats of India, Nagendra and Gadgil (1999b) mapped a landscape into seven habitat types ranging from secondary evergreen forests to paddy fields, using supervised and unsupervised classification of IRS 1B LISS 2 satellite imagery. Plant communities (angiosperms excluding grasses) distributed in these habitat types were surveyed in the field using 246 plots of 10 m by 10 m. Habitat types could be identified on the basis of supervised classification with an accuracy of 88%, and plots belonging to different habitat types differed significantly in their species composition ($p<0.05$)*"(Nagendra 2001). According to Nagendra (2001), „*the degree of correspondence between habitat maps and species distributions depends on the degree of habitat map generalization*" (Nagendra 2001). For biodiversity assessment, habitat mapping appears limited to the scale of tens of kilometres (He *et a*l. 1998), because „*species diversity varies not only between, but also within habitats*" (Nagendra 2001).
Buehler et al. (2006) successfully modelled cerulean warbler (Dendroica cerulea) habitat from remotely sensed vegetation and landform data based on the Mahalanobis distance statistics D^2. "*D^2 is a measure of dissimilarity and represents the difference in the standard multivariate squared distance between the ideal cerulean location and other locations on the study area (Clark et al. 1993)*" (Buhler et al. 2006). Seven explanatory variables (average solar exposure pre year, distance to nearest stream, elevation, slope, relative slope position, terrain relative moisture index, coverage of mature decidous forest) were selected a priori with regard to cerulean habitat requirements. The coverage of mature (>30 years old) deciduous forest was derived through a supervised classification of LandSat TM satellite imagery. Elevation was taken from the United States Geological Survey digital elevation model. A slope coverage was generated using the ArcGIS Spatial Analyst SLOPE command, and the remaining variables (average solar exposure pre year, distance to nearest stream, relative slope position, terrain relative moisture index) were calculated in ArcInfo. The model was developed in ArcView (Environmental Systems Research Institute, Redlands, California) extension (Jenness Enterprises, Flagstaff, Arizona), while the known cerulean locations mapped in the field represented the response variable. After the model was developed, a cutoff vaue of the Mahalanobis distance was selected so that it "*maximized the difference in cumulative frequency between the cerulean locations and the study area as a whole (Browning et al. 2005). By setting the cutoff value in such a manner, we maximized the ability of the model to discriminate between cerulean habitat and conditions available on the rest of the study area*" (Buehler et al. 2006).
The potential of different remote sensing indices to describe and monitor species richness was examined by Koch and Ivits (2004) using the data sampled within the BioAssess project. Within this research a fused Landsat-IRS image was selected as a standard dataset. „*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*". A hierarchical segmentation based classification scheme based on the CORINE database was used. With this scheme, the segments were classified into two levels: (1) project coarse level, (2) country level based on country specific characteristics. „*Extracted classes from visual interpretation and segmentation-based classification were used to quantify the land use intensity gradient and to*

calculate landscape indices". The calculated landscape metrics were then related to terrestrial based species diversity indices. „*Stepwise linear regression was applied to model species richness for woody plants, carabids and birds as dependent variable*". The results of the analysis showed that „*the derived remote sensing indices showed good potential in predicting species diversity data*" (Koch and Ivits 2004).

Griffiths et al. (2000) who examined the potential to predict plant diversity only from landscape structure measures derived from remotely sensed data presented that the results obtained from such a model proved difficult to interpret, which „*highlighted the need to obtain data on both landscape quality and landscape structure*". „*The key habitats of species can be identified by combining satellite- and field-based habitat data, landscape structure and species abundance information (Saveraid et al. 2001, Scribner et al. 2001)*" (Kerr and Ostrovsky 2003).

(6) Development of indicators

„*The ENVIP Nature project is an example of the application of remote sensing and GIS techniques in landscape ecology and conservation biology, targeted at the development of indicators for nature conservation*" (Groom et al. 2006). Within this project, the indicators for the criteria ‚naturalness', ‚vulnerability', and ‚threat' have been developed based on the analysis of the extent, spatial configuration and selected shape parameters of the habitat map (Groom et al. 2006).

4.6 Downscaling

The term downscaling (named also disaggregation (e.g. McBratney 1998, Rajat-Bindlish and Barros 2000, Hopmans et al. 2002, Arnell et al. 2004, Bougadis and Adamowski 2006) or top-down approach (e.g. Gon et al. 2000, Roy and Behera 2002)) refers to transfering information estimated/obtained at a very large scale to a fine resolution (Seem 2004) in such a way that „*it can be interpreted in light of local circumstances*" (Riitters 2005).

According to Tatl et al. (2004), downscaling is a solution of the problems that arise when modelling interconnections between global and regional scales, „*such as prediction of regional, local-scale climate variables from large-scale processes*" (Tatl et al. 2004). Downscaling coarse resolution data is particularly useful for impact assessment studies (Miller et al. 1999), because such data are an inadequate basis for the assessments (e.g. of the effects of climate change on land-surface processes) at regional scales (Wilby et al. 1999). This is because the resolution of data sources (e.g. of Remote Sensing sensors, General Circulation Model GCM, etc.) is too coarse to resolve important sub-grid scale processes and because the output of specific models (e.g. GCM) is often unreliable at individual and sub-grid box scales. By establishing relationships between grid-box scale indicators and sub-grid scale predictands, downscaling represetns a practical means of bridging this spatial difference (Wilby et al. 1999).

The term downscaling is most often used in climate studies. In this field, there exists a number of different downscaling methods „from simple interpolation to more sophisticated dynamical modelling, through multiple regression and weather generators" (Prudhomme et al. 2002). They are are divided into two main categories with regard to the approach used:

- statistical (empirical) downscaling

In climate studies, statistical downscaling relates large-scale circulation patterns to local weather records (Bugmann et al. 2000). „*Statistical downscaling (SD) methods apply climate variables from General Circulation Models (GCMs) to statistical transfer functions to estimate point-scale meteorological series*" (Diaz-Nieto and Wilby 2005). „*Statistical downscaling adopts statistical relationships between the regional climate and carefully selected large-scale parameters [c.f von Storch et al., 1993; Wilby et al., 2004; Goodess et al., 2005]. These relationships are empirical (i.e. calibrated from observations) and they are applied using the predictor fields from GCMs in order to construct scenarios*"(Schmidli et al. 2007).

„*Several SD methods have been proposed in recent years*“ (Nguyen et al. 2006). From general statistical methods, regression models (Murphy 2000, Schoof and Pryor 2001, Wood et al. 2004, Spak et al. 2007), and artificial neural network (ANN) (Schoof and Pryor 2001, Harpham and Wilby 2004, Harpham and Wilby 2005, Khan et al. 2006) are most popular, but other methods are also applied. In the last IPCC report, three regression-based methods that try to overcome the imperfection of point-wise variability of empirical downscaling were mentioned: randomization, inflation, and expanded downscaling (Burger and Chen 2005). Mpelasoka et al. (2001) adapted ANN and multivariate statistics (MST) in order to derive changes of site precipitation and temperature characteristics. For downscaling climate model outputs for use in hydrologic simulation Wood et al. (2004) applied three SD methods: linear interpolation, spatial disaggregation, and bias-correction and spatial disaggregation. The K-nearest neighbour (K-nn) as an analog-type approach is used in Gangopadhyay et al. (2005). Other popular specific SD methods in climatic and hydrologic studies are based on the Statistical Downscaling Model (SDSM) (Nguyen et al. 2006, Harpham and Wilby 2005, Khan et al. 2006), which is a conditional resampling method (Harpham and Wilby 2005), and the Stochastic Weather Generator (LARS-WG) (Nguyen et al. 2006, Khan et al. 2006).

- dynamical downscaling

"*Dynamical downscaling uses regional climate models (RCMs) to simulate finer-scale physical processes consistent with the large-scale weather evolution prescribed from a GCM [c.f. Giorgi et al. 2001; Mearns et al. 2004]*“ (Schmidli et al. 2007). While statistical downscaling methods are based on empirical models, regional models explicitly describe the physical processes affecting climate (Giorgi 2001). „*These models are able to generate a dynamically consistent suite of climate variables, but there is significant uncertainty in parameterization of sub-grid-scale processes, and the computational costs of RCMs are high*“ (Spak et al. 2007). There exists a number of RCMs developed by different institutes that cover different parts of the world (see e.g. Schmidli et al. 2007, Spak et al. 2007). The RCMs have also been used to derive hydrologic model (Wood et al. 2004, Payne et al. 2004).

In addition, Bouwer et al. (2004) identified kriging as a spatial downscaling technique.

In biodiversity studies, the term downscaling is not as common as in the papers dealing with climate and hydrology, although the scaling problem has been recognised by several authors (see e.g. Nagendra and Gadgil 1999, Kerr and Ostrovsky 2003). Roy and Behera (2002) used the term „top down“ approach instead and identified its four main features: (1) stratified approach, (2) extrapolation on large areas, (3) systematic monitoring, (4) spatial environmental database.
Although it is generally accepted that for large-scale biodiversity assessments „*remote sensing is by far the best technique to gather information on large areas*“ (Jongman et al. 2006), because, when compared to other survey techniques, remote sensing is „*unique in its possibilities for providing census data, i.e.complete large area coverage that can complement sample data*“ (Inghe 2001 in Groom et al. 2006), the information gathered in this way is constrained by the resolution of the sensor (Jongman et al. 2006). Therefore, in the downscaling process the image data need to be combined with ancillary infomation. For example, in the ENVIP Nature project, a ‚normal‘ land cover map was transformed into an ecologically meaningful data set called the ‚broader habitat map‘ using ancillary GIS data such as digital terrain model or specific management information derived from topographical maps (forest road network, tourist hot spots) (Groom et al. 2006). Similarly, in order to derive the forest-non-forest maps from Landsat imagery, Kozak et al. (2007) used a digital elevation model and the Corine Land Cover data.
Riitters (2005) demostrated downscaling using the results of U.S. national assessments from land-cover maps that were derived from satellite imagery (Landsat Thematic Mapper). The author identified the map extent, map resolution in terms of the land cover classes, and the „habitat model“ as candidates for downscaling.

„*Considering map extent, it is trivial to examine the indicator values at any particular location, or to summarize the values within and arbitrary map extent such as watershed*“ (Riitters 2005).
Considering resolution, more specific land cover classes can be identified for a specific analysis using either the original map or local maps that can provide more detailed thematic resolution (Riitters 2005).
The author identified the most opportunities for downscaling in the domain of the habitat model. A moving window device was used to measure habitat structure characterized by habitat amount and habitat spatial pattern (connectivity).„*A moving window operates by moving a fixed-area window over the map so as to place a support region around each pixel*“ (Riitters 2005). Habitat amount is defined as the proportion of pixels in the support region that are forest, while connectivity is estimated as the percent of {forest, forest} adjacent pixel pairs in the support region. The term support region refers to a shape and size of a moving window. „*Measurements are made at each placement of the window, and the values are assigned to the location of the pixel at the center of the support region*“ (Riitters 2005).
„*Choosing a particular support region size based on home range size (Riitters et al. 1997) constitutes downscaling in the domain of the habitat model*“(Riitters 2005). „*Scaling in the habitat model domain include choosing a specific indicator, or setting a threshold value for an indicator*“(Riitters 2005). Riitters (2005) presented a number of approaches how the indicators ‚habitat amount‘ and ‚connectivity‘ can be used for downscaling in the habitat model domain. Considering specific habitat or movement requirements (e.g. size, density, adjacency, corridors, etc.) of a particular species, it is possible to pre-define a threshold value for habitat amount and/or habitat spatial pattern and thus to find suitable habitats for the species.
The same sort of analysis based on the threshold value of an indicator can be combined with downscaling in the spatial domain. Riitters (2005) showed how the reduced map extent affects the results of the analysis. While for reasonably large extents (e.g. millions of hectares) the types of trends are usually monotonic, „*for smaller extents (e.g. thousands of hectares), departures from monotonic forms create the opportunity for very localized interpretations of structure*“ (Riitters 2005).
Although with the moving window device any indicator can be measured, according to Riitters (2005) the amount and connectivity of habitat seem to be sufficient, while other indicators, e.g. those obtained from a patch-based approach, such as perimeter-area ratio, or amount of core forest, can be recovered from a moving window analysis by combining the two mentioned indicators.
Another approach how to identify suitable habitats for a species of interest was presented by Buehler et al. (2006), who developed the Mahalanobis distance statistic model of potential habitat for cerulean warbler (for details see Chapter 3.5.2, point (5)).

As an alternate method to the moving window, Vogt et al. (2007a) used morphological image processing for mapping land-cover spatial pattern. Classification algorithm is here „*defined by a sequence of logical operations such as union, intersection, complementation, and translation using geometric objects called ‘structuring elements’ (SE) of pre-defined shape and size*“ (Vogt et al. 2007a). In their work, Vogt et al. (2007a) consider two structuring elements SEs: an 8-neighbourhood (SE1) and a 4-neighbourhood (SE2), that pre-define which and how many pixels around the examined (center) pixel are accounted for in the analysis. In addition, the shape and dimension of SEs also define the direction and extent of the morphological operations. The authors used two operations: the ‘erosion’ that shrinks regions of forest and the ‘dilation’ that expands them. The image processing starts with the forest - non-forest map, in which core forest is identified by applying erosion on SE1. It means that „*the center pixel of SE1 is core forest if all eight neighbors are forest*“ (Vogt et al. 2007a). Similarly, patch, edge, and perforated forests were detected in successive steps. The result was a forest pattern map with four classes: core, patch, edge, and perforated forest (Vogt et al. 2007a). The comparison of the morphological image processing and image convolution (i.e. window approach) revealed that the morphological approach is more accurate at the pixel level. Hence, „*summary statistics and trend analyses at landscape level will* *also be more accurate. These improvements will allow an unsupervised and precise spatial pattern analysis at both, the pixel and landscape level*" (Vogt et al. 2007a).

Vogt et al. (2007b) presented an extension of the described morphological image processing for identifying and pixel-level mapping of structural corridors. In this paper, the authors applied a more detailed clasification of land cover, since they considered nine classes of forest pattern including corridors. Apart from the two fundamental operations used in the previous study, an additional morphological operation known as 'skeletonization' (Calabi and Hartnett 1968 in Vogt et al. 2007b), that refers to „*a process which iteratively removes the boundary pixels of a region to its line representation*" (Vogt et al. 2007b), was used in this application. The paper shows „*how the approach can be used to differentiate between relatively narrow ('line') and wide ('strip') structural corridors by mapping corridors at multiple scales of observation, and indicate how to map functional corridors with maps of observed or simulated organism movement*" (Vogt et al. 2007b).

In forestry, regression models are often used to estimate properties of a single tree, a stand or a whole region. The models are „usually developed to estimate a measure which is expensive, slow or even impossible to gauge, such as a height or volume of a tree. Normally these variables are predicted with models, which are estimated from a sample of the population in question or from another similar kind of population" (Räty and Kangas 2007). If a regression model was fitted to a large study area, it may have poor statistical properties when applied to smaller subregions. Räty and Kangas (2007) presented an approach how to localize general regression models. The authors understand under the term ‚localization' a refitting (or adjusting) of the original model to the sub-area data in order to improve the estimates of the model parameters, while no new elements (e.g. variables) are added to the model. The neccessity for a localization is indicated by the performance of the residuals of the model. „*If there is a global trend in the residuals, the localization is surely worthwhile. A global trend means a change in the residual values which follows spatial location in the area. A global trend is a phenomenon which is on average true*". For the selection of localization areas, Räty and Kangas (2007) used the local indicators of spatial associations (LISA) derived from global indices Moran 's coefficient I and Geary 's ratio c. „*LISAs were calculated from the residuals of a form height regression model,which was fitted to the original data*". The improvement of the model performance in localization versus global fitting was measured with a standard deviation, a root mean square error (RMSE) of model residuals, and a relative bias. The study revealed that the localization removed the local bias of the global model.

According to Räty and Kangas (2007), „*another method could be a localization based on kriging… A variogram could be used to determine an appropriate size for a localization area. However, the simplicity of the LISA method may then be lost*".

5. References

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