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Review

Contribution of large-scale forest inventories to biodiversity assessment and monitoring

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ABSTRACT

Statistically-designed inventories and biodiversity monitoring programs are gaining relevance for biological conservation and natural resources management. Mandated periodic surveys provide unique opportunities to identify and satisfy natural resources management information needs. However, this is not an end in itself but rather is the beginning of a process that should lead to sound decision-making in biodiversity conservation. Forest inventories are currently evolving towards multipurpose resource surveys and are broadening their scope in several directions: (i) expansion of the target population to include non-traditional attributes such as trees outside the forest and urban forests; (ii) forest carbon pools and carbon sequestration estimation; (iii) assessment of forest health; and (iv) inclusion of additional variables such as biodiversity attributes that are not directly related to timber assessment and wood harvesting.

There is an on-going debate regarding the role of forest inventories in biodiversity assessment and monitoring. This paper presents a review on the topic that aims at providing updated knowledge on the current contribution of forest inventories to the assessment and monitoring of forest biodiversity conditions on a large scale. Specific objectives are fourfold: (i) to highlight the types of forest biodiversity indicators that can be estimated from data collected in the framework of standard forest inventories and the implications of different sampling methods on the estimation of the indicators; (ii) to outline current possibilities for harmonized estimation of biodiversity indicators in Europe from National Forest Inventory data; (iii) to show the added value for forest biodiversity monitoring of framing biodiversity indicators into ecologically meaningful forest type units; and (iv) to examine the potential of forest inventory sample data for estimating landscape biodiversity metrics.

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1. Introduction

Statistically-designed inventories and biodiversity monitoring programs are gaining relevance for biological conservation and natural resource management. Mandated periodic surveys provide unique opportunities to identify and properly satisfy natural resource management information needs. However, this is not an end in itself but rather is the beginning of a process that should lead to sound decision-making in biodiversity conservation.

From this perspective, forest inventories can be regarded as effective tools for estimating the kind, amount and condition of forest resources over large areas. The use of statistical sampling coupled with periodic re-measurements of permanent sample units provides the basis for measuring changes in forest conditions and constructing models to estimate trends (Lund et al., 1998). The information is generally reported for management and/or administrative units (e.g. district, province, country) and/or for thematic or resource classes (e.g. forest type, age).

Large-scale forest inventories, such as National Forest Inventories (NFIs), have gained ground over the last decades as mandated programs for providing the information necessary to fulfill reporting obligations under international agreements such as the FAO Global Forest Resource Assessment, the Kyoto protocol, the United Nations Convention on Biological Diversity (CBD), the Ministerial Conference for the Protection of Forests (MCPFE–Forest Europe), and the Montréal Process. For this purpose, the use of data from stand-wise inventories has often been discontinued in favor of regional and national forest inventories where plots are the primary sampling units rather than forest stands (Motz et al., 2010).

All sample-based inventories over large areas share a common methodological feature: sample units are objectively selected by rigorous probabilistic rules as a means of guaranteeing the credibility of estimates (Olsen and Schreuder, 1997).

Traditionally, large-scale forest inventory data are analyzed in the framework of design-based inference which assumes population values are fixed constants; the randomization distribution resulting from the sampling design is the basis of the inference. In this framework, the bias and variance of an estimator of a population parameter are determined from the set of all possible samples (the sample space) and from the probability associated with each sample. Usually, forest inventories adopt sampling schemes in which a set of points is randomly selected from the study region in accordance with a spatial sampling design. Subsequently, plots of adequate radius or angle-counts with a predefined basal area factor are established with centers at the selected points, and forest attributes are recorded for the trees included in the plots, or in the angle-counts (e.g. De Vries, 1986; Schreuder et al., 1993; Fattorini et al. 2006).

Forest inventories are currently evolving towards multipurpose resource surveys (Lund, 1998; Corona and Marchetti, 2007; Tomppo et al., 2010) and are broadening their scope in several directions: (i) expansion of the target population to include non-traditional attributes such as trees outside the forest and urban forests; (ii) forest carbon pools and carbon sequestration estimation; (iii) assessment of forest health; and (iv) inclusion of additional variables such as biodiversity attributes that are not directly related to timber assessment and wood harvesting.

Biodiversity monitoring is an essential prerequisite to support management decisions to maintain multiple forest ecosystem functions in the long term. Thus, assessing and monitoring biodiversity status should be regarded as strictly tied to sustainable forest management (see Criterion 4, Forest Europe, UNECE and FAO, 2011). In particular, the ecosystem approach fostered by CBD (2000) brings into sharper focus that the many components of biodiversity control the stores and flows of energy, water and nutrients within ecosystems, and provide resistance to major perturbations. Hence forest resource inventories must expand from traditional variables related to wood and timber production to the assessment of the composition, structure and function of forest ecosystems, and must provide a better understanding of the roles of the components of biological diversity in the provision of multiple forest ecosystem functions.

Forest inventory and biodiversity survey methods are similar in many ways, but also have multiple differences (Newton and Kapos, 2002). The debate regarding the potential role of forest inventories in biodiversity monitoring is still open. Some authors argue that the actual capability of forest inventories to directly support biodiversity management is still generally poor around the world (Lindenmayer et al., 2006). However, Tomppo et al. (2010) demonstrate that despite the timber-oriented approach that largely prescribes the information collected by European NFIs, a substantial proportion of forest biodiversity attributes can be estimated from NFI data (Winter et al., 2008; Chirici et al., 2011). Additionally, several studies and exercises have been carried out in the last decades to find ways of effectively integrating biodiversity issues within forest inventories (e.g. Corona et al., 2003; Motz et al., 2010). Recently Chirici et al. (submitted for publication) demonstrated that NFIs can report comparable or harmonized estimates of indicators for multiple biodiversity features (forest categories, deadwood, forest age, forest structure and naturalness), but for others (ground vegetation and regeneration) NFIs should invest more in harmonization efforts (see also Web references, COST Action E43).

Building on the premise that forest inventories have the potential to make substantial contributions to the large-scale assessment and monitoring of forest biodiversity, this paper provides a review of issues that lead to a more complete realization of that potential, with major focus on European NFIs. The remainder of the paper is organized into three sections: Section 2 includes the previously mentioned review; Section 3 includes a brief follow-up discussion with several recommendations; and Section 4 includes a brief summary and comments on future prospects. The main review part of the paper, Section 2, consists of a sequence of sub-sections that begins with a brief general discussion of biodiversity indicators and progresses to the estimation of meaningful landscape-level biodiversity metrics. Section 2.1 focuses on selection of forest biodiversity indicators that can be estimated using standard forest inventory variables; Section 2.2 focuses on sampling issues and includes two relevant examples; Section 2.3 focuses on harmonized estimation of forest biodiversity indicators to facilitate and enhance international reporting; Section 2.4 focuses on the utility of estimating forest biodiversity indicators by forest habitat types; and Section 2.5 focuses on sample-based estimation of landscape metrics that contribute to identification of critical changes in the spatial pattern of forest habitat types that lead to biodiversity loss.

2. Estimating indicators of forest biodiversity using forest inventory data

2.1. Forest biodiversity indicators

Building a comprehensive picture of forest biodiversity is a complex task, because it is not feasible to measure or quantify it in its entirety. An alternative is to use indicators, a practical way of measuring biodiversity by redefining it in terms of measurable attributes relevant to the scale and purpose for which it is being assessed (Williams, 2004). Based on the assumption that a large variety of forest structures and/or tree species generally also provide large numbers of habitats for different plant and animal species (Winter and Möller, 2008; Winter et al., 2008; Motz et al., 2010), measures related to trees and stand structure play a key role in the derivation of biodiversity indicators (Larsson, 2001).

Forest biodiversity indicators are generally selected following two main approaches:

(i) counting the number of plant species in a given area (plant species richness, see Section 2.2.1) or using a mathematical construct to summarize in a single number or index value information on the diversity of plant species and/or habitats (see Table 2, Evenness). This type of indicator is always compositional and it is a direct measure of biodiversity *per se*; however, this does not imply that diversity indices directly and clearly inform the status of biodiversity and trends: e.g. the diversity index of a plot/forest area might increase due to the introduction of new invasive species which replace native species;

(ii) by means of structural indicators based on key structural features (e.g. deadwood, variability in tree size, large trees) or quantified by indices of structural complexity (see McElhinny et al., 2005); structural indicators act as correlate or surrogate for other direct measures of biodiversity; such indicators are gaining ground at the operational level not only because structural components are easier to measure, but also because the effect of management actions on structural components can be direct and easy to assess, compared to the impact, e.g. on faunal species (McElhinny et al., 2005).

Standard forest inventories are theoretically designed to monitor forest changes. As a rule, forest biodiversity indicators can be assessed with known and low costs and periodic re-measurements can reflect changes at 5–10 years periods, which may be optimal from an ecological point of view (Larsson, 2001).

Thus, the question becomes: what are the types of forest biodiversity indicators for which forest inventories can provide information? According to Newton and Kapos (2002), biodiversity indicators can be referred to the following general groups: (i) forest area by forest type, and successional stage and phases relative to land area; (ii) protected forest area by type, successional stage and phases, and protection category relative to total forest area; (iii) degree of fragmentation of forest types; (iv) rate of conversion of forest cover (by type) to other uses; (v) area and percentage of forests affected by anthropogenic and natural disturbance; (vi) complexity and heterogeneity of forest structure; (vii) number of forest-dependent species; (viii) conservation status of forest dependent species. Recommended forest biodiversity indicators also include tree mortality and recruitment, exotic weeds, introduced herbivore impacts, and woody debris (e.g. Allen et al., 2003).

Plant species richness (mainly tree species considered) is widely used as a compositional biodiversity indicator. Chiarucci and Bonini (2005) demonstrated the capabilities of forest inventory, design-based, probabilistic sampling for estimating the richness of vascular plant taxa over large forest areas. Such sampling schemes allow quantitative floristic techniques to be applied to compare sites and monitor changes. The same authors argue that although probabilistic sampling designs limit the probability of finding rare species, they may be useful for detecting changes of indicators of general features of the flora such as the relative proportions of native and invasive alien species. The assessment of such indicators is rather straightforward when floristic data can be cross-referenced to national and international reference lists of native versus alien species such as the Atlas Florae Europeae or DAISIE European Invasive Alien Species Gateway (see Web references).

2.2. Sampling considerations when estimating forest biodiversity indicators

Most papers devoted to the assessment of plant diversity are based on the assumption that individuals are selected from the community by means of simple random sampling with replacement. Nevertheless, as emphasized by Heltshe and Forrester (1983), the individuals in separate sampling units are not randomly or independently drawn because they all belong to the same parent community and, therefore, are in close spatial proximity. In addition, conventional probability sampling designs are not always adequate for assessing species richness, therefore additional complex analyses can be required. Section 2.2.1 which follows addresses this topic using as an example a statistical-based technique applicable to the assessment of plant species richness.

Other common forest biodiversity indicators suitable for estimation using forest inventory data are indices accounting for basic tree diversity aspects (Gadow, 1999; Pommerening, 2002) such as the diversity of tree locations, species diversity (e.g. the Shannon index based on stem number or basal area per tree species) and the diversity of tree dimensions (e.g. stem diameters, tree heights). Section 2.2.2 addresses this topic with a focus on the effects of different sampling techniques on the estimation of tree diversity indicators.

2.2.1. Inventorying plant species richness from standard forest inventories

Rarefaction curves (RCs) are powerful tools to explore the effects of different sampling options (number of plots, plot size, spatial layout) on the assessment of plant species richness over large forest areas. RCs constitute order-free curves showing the increase in the number of observed species as the number of plots increases.

Accordingly, a curve that approaches an asymptote indicates that few additional species would be observed if the number of plots is further increased; on the other hand, a curve which sharply rises near its end means that many new species could be observed on additional plots. To optimize survey efforts, account must be made for the expectation that more species would be observed when many small plots are surveyed instead of a small number of large plots (e.g. see Fig. 1). This expectation derives from the increased likelihood of observing the same species on neighboring sites within large plots which, in turn, results from the spatial dependence among locations of plants caused by a mixture of both exogenous (e.g. disturbances or underlying environmental conditions) and endogenous processes acting on species distribution (dispersal, spatial competition).

2.2.2. Inventorying tree diversity from standard forest inventories

Besides tree species composition, other tree-related variables routinely measured on sample plots are tree size (e.g. stem height and diameter at breast height, dbh) and tree position. According to Motz et al. (2010), this enables the estimation of indices of stand structural complexity combining measures for variations in tree-related variables (e.g. basal area, tree spacing). These authors showed that in the majority of cases angle-count sampling is less suitable for assessing such indices than is fixed-radius plot sampling. Generally the loss in precision of quantitative information on forest structure or the loss in efficiency of sampling forest structure is less when the variation in tree size is small. Thus, in forests with even-aged, mono-specific stand management, which usually results in relatively less diameter variation, both sampling methods are equally efficient, whereas for uneven-aged, mixed species forests, the fixed-radius plot sampling method is the most suitable sampling method for the majority of tree diversity indices.

Maltamo and Uuttera (1998) also investigated sampling measures of forest structure by means of the angle-count method: the results of their study revealed compensation for the lack of precision incurred by angle-count sampling can be achieved to some degree by substantially increasing sample sizes relative to those for conventional forest inventories, using the same sampling method for estimating current stand volume. Kleinn and Vilčko (2006a,b) and Nothdurft et al. (2010) demonstrated that methods associated with spatial statistics can be used to correct biases resulting from sampling methods such as distance or *k*-tree methods.

The number of trees measured within a fixed-radius plot is influenced both by the plot size and by the tree definition adopted. This may strongly affect the resulting diversity indicators. For instance, McRoberts et al. (2008) demonstrated that tree dbh alpha-diversity (Magurran, 2004) was highly correlated with mean dbh at plot level for a dataset of 12,000 plots from the NFI of the USA. Thus, the minimum dbh adopted for tree mensuration has a relevant impact on estimates of biodiversity indicators. Tree diversity, both in terms of dbh and species, is affected by the minimum tree dbh threshold: the estimate of diversity linearly decreases as the minimum dbh increases (McRoberts et al., 2009).

2.3. Harmonizing the estimation of forest biodiversity indicators

A comprehensive assessment of the capabilities of European NFIs for assessing the state and changes in key components of biodiversity has been reported by Chirici et al. (2011) and Winter et al. (2008). The main findings indicate that although many variables are feasible for assessment in NFIs most biodiversity indicators whose estimation can be harmonized are based on tree-related variables, while few indicators can be estimated in a harmonized way for other components such as ground vegetation. The former indicators (Table 1) represent the current contribution of NFIs to forest biodiversity monitoring, not the potential best set of indicators for monitoring forest biodiversity nor the set required to completely satisfy international reporting commitments.

Although different plot sampling schemes (spatial layout of plots, number, size, fixed/variable radius sampling) are currently applied in European NFIs some slight changes in the protocols and thresholds for measuring tree-related variables could facilitate the harmonization process. Notably, the harmonization of estimates of indicators based on number of trees, basal area, growing stock and deadwood volume would benefit from the adoption of the following standard data collection procedures: (i) a minimum dbh close to zero for tree stem callipering; (ii) recording the tree coordinates; and (iii) for lying deadwood, measuring the diameters at both the ends of the elements and the adoption of a minimum diameter threshold not larger than 10 cm and minimum length threshold not longer than 1 m. To estimate harmonized forest age indicators, the ages of tree stems should be acquired by field assessment. Further investigations are still needed for ground vegetation and regeneration indicators. Ground vegetation indicators have the same potential relevance as tree related indicators for biodiversity assessment. In light of this, NFIs should also make a consistent investment in developing valid sampling procedures for these forest vegetation components (Chirici et al., 2011, submitted for publication).

Table 1
Forest biodiversity features and related indicators that can be estimated using European NFI data (Chirici et al., 2011).

Feature	Indicator
Forest type	1.1. Forest category according to the system of nomenclature developed by the EEA (2006)
Forest structure	2.1. Relative abundance of native tree species in terms of basal area
	2.2. Number of native tree species
	2.3. Proportion of plots with 1, 2, 3 and more native tree species
	2.4. Largest diameter trees
	2.5. Standard deviation of the tree heights
	2.6. Number of vertical layers
	2.7. Frequency distribution of standard deviation classes of dbh
	2.8. Shannon index for tree species
Forest age	3.1. Dominant age
	3.2. Mean age
	3.3. Weighted mean age
	3.4. Old trees
Deadwood	4.1. Deadwood volume by decay class, tree species, and horizontal/vertical position
Ground vegetation	5.1. Cover of ground vegetation
	5.2. Cover of shrub species
	5.3. Presence/absence of shrub species
	5.4. Presence/absence of shrub genus
	5.5. Presence/absence of ground vegetation life forms
Naturalness	6.1 Naturalness index

2.4. The utility of forest typological classifications

Clearly, sample-based data collection, periodic re-measurements of permanent sample units, and the use of standard field protocols are all arguments in favor of using standard forest inventories to assess and monitor selected forest biodiversity indicators. Forest biodiversity monitoring capabilities can be further enhanced if such indicators are referenced to forest habitat types by explicitly considering the spatial variability and internal ecological heterogeneity of the forest population of interest (Rondeux, 1999). In particular, forest types enhance biodiversity assessments (Larsson, 2001) by using a typological classification of forest area to frame biodiversity indicators collected over wide areas into smaller, more homogeneous units characterized by similar key determinants of biodiversity (e.g. assemblages of forest dominant species).

Linking sample data on biodiversity indicators to ecologically meaningful forest type units brings substantial advantages for forest biodiversity assessment: (i) it allows improved understanding, interpretation and communication of data on biodiversity variables by enabling comparison of ecologically similar forests; (ii) it enables a more detailed and richer analysis of biodiversity indicators in a specific forest habitat such as the relationship between the vertical structure of forest habitat and vertebrate and invertebrate fauna diversity (Rego et al., 2004); and (iii) it provides a basis for stratified sampling, thus ensuring that different forest habitats are adequately represented in the plots (Winter et al., 2011, submitted for publication).

To this end, several forest typological approaches have been devised in Europe in the framework of NFIs (e.g. Rego et al., 2004) and sustainable forest management regional strategies (e.g. Corona et al., 2004a), or to cross-link NFI sample data to units of European

Table 2

Presence, richness and diversity of forest habitat types in selected European countries, classified by the 14 categories of European Forest Types (EFTs, EEA, 2006). Richness is expressed as total number of EFTs in a country. Diversity is expressed as Evenness = $-\frac{\sum_i p_i \ln p_i}{\ln 14}$, where: i = 1, . . . , 14th EFT; p_i = share of the forest area of ith EFT out the total forest area in the country. (Source: Elaboration from NFIs data collected in MCPFE/Forest Europe reporting; Reference year: 2010, except: *2005; **2000).

Country	1. Boreal forest	2. Hemiboreal and nemoral coniferous and mixed broadleaved-coniferous forest	3. Alpine forest	4. Acidophilous oak and oak-birch forest	5. Mesophytic deciduous forest	6. Beech forest	7. Mountainous beech forest	8. Thermophilous deciduous forest	9. Broadleaved evergreen forest	10. Coniferous forests of the Mediterranean, Anatolian and Macaronesian regions	11. Mire and swamp forest	12. Floodplain forest	13. Non-riverine alder, birch or aspen forest	14. Introduced tree species forest	*EFTs richness*	*EFTs diversity*
Austria		■	■	■	■	■	■	■			■	■	■	■	11	0.59
Belarus	■	■		■	■						■	■	■		7	0.45
Belgium				■	■	■					■	■		■	6	0.53
Bulgaria		■	■		■	■	■	■		■				■	8	0.61
Croatia		■	■	■	■	■	■	■	■	■		■		■	11	0.76
Cyprus								■		■				■	3	0.03
Czech Republic		■	■	■	■	■	■	■			■	■	■	■	11	0.51
Denmark		■		■	■	■					■	■		■	7	0.54
Estonia	■	■									■		■	■	5	0.41
Finland	■	■	■								■		■	■	6	0.34
France		■	■	■	■	■	■	■	■	■	■	■	■	■	13	0.85
Germany		■	■		■	■	■				■		■	■	8	0.53
Hungary				■	■	■		■			■	■	■	■	8	0.56
Iceland			■										■	■	3	0.30
Ireland				■	■						■	■	■	■	6	0.24
Italy		■	■		■	■	■	■	■	■		■	■	■	11	0.66
Latvia	■	■									■		■	■	5	0.46
Lithuania		■			■						■	■	■	■	6	0.47
Netherlands		■		■	■	■							■	■	6	0.61
Norway	■	■	■	■	■	■					■	■	■	■	10	0.53
Poland		■	■	■	■	■	■	■			■	■	■	■	11	0.51
Slovakia		■	■	■	■	■	■	■			■	■	■	■	11	0.68
Slovenia		■	■	■	■	■	■	■				■			8	0.56
Spain		■	■	■	■	■	■	■	■	■		■	■	■	12	0.60
Sweden	■	■	■	■	■	■					■	■	■	■	10	0.51
Switzerland*		■	■	■	■	■	■	■			■	■	■	■	11	0.63
UK**		■		■	■	■						■	■	■	7	0.45
Ukraine		■	■	■	■	■	■	■		■	■	■	■	■	12	0.68

relevance for the assessment of forest condition such as the European Forest Types (EFTs; see EEA, 2006; Barbati et al., 2007, 2011). Table 2 presents a comprehensive snapshot of the variety of forest habitats found in European countries expressed in terms of richness and diversity of the EFTs. The main value of such stratification should not be simply to seek identification of countries with the greatest variety of forest habitat types, i.e. exhibiting greatest internal ecological heterogeneity of a forest population; rather, an EFT-based stratification sets the scene for further analysis of the values taken by biodiversity indicators in Europe. An example is shown in Table 3 where the total volume of deadwood (sum of standing and lying components) for different forest types in a sample of European countries is reported. Reporting estimates by EFTs reveals variability in the amount of deadwood with different vegetation zones. For the examined countries, the greatest per-ha deadwood levels are observed for EFTs associated with mountain

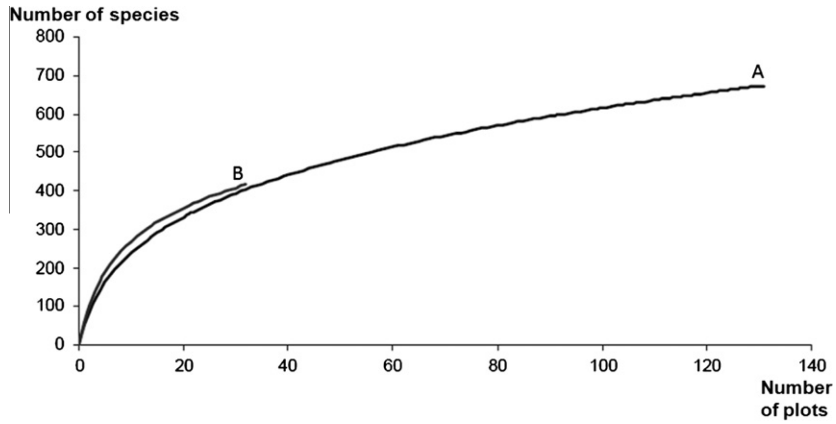

Fig. 1. Rarefaction curves (Section 2.1) resulting from two inventory test surveys of vascular plants carried out on the same forest area (from Corona et al., 2010). The A curve is composed from 132 circular sample 1250-m^2 plots located by a tesselated stratified sampling design over the inventoried area, while the B curve is subsample selected by simple random sampling without replacement of 2500-m^2 squared plots centered at each corresponding 1250-m^2 plot. It is at once apparent from the figure that the use of a large number of small plots is more effective (in terms of plant species detection) than the use of a small number of large plots (673 detected species versus 417). Even if the same number of plots were used in both the inventory tests, the survey by 1250-m^2 plots would have performance (404 detected species) quite similar to that achieved by the 2500-m^2 plots.

regions: Alpine (EFT 3) and Mountainous beech forests (EFT 7). This phenomenon cannot be explained solely in terms of favorable ecological growing conditions; rather, it is likely linked to the poor accessibility and thus low intensity of forest harvesting in mountainous areas which, in turn, results in greater deadwood accumulation levels. In contrast, average deadwood levels are less in forest types more subject to intensive management and short rotations (EFTs 1, 2, 4, 5, 8, 14). Thus, referencing a biodiversity indicator such as deadwood to EFTs is particularly relevant from the perspective of monitoring country and/or regional progress in implementing silvicultural practices that promote biodiversity conservation through deadwood preservation.

2.5. Estimating landscape metrics

So far we have dealt with the use of forest attributes recorded at sample plots to estimate biodiversity indicators for entire populations. Information on the relative positions of sample plots (e.g. the spatial distribution of sample plots in terms of proximity to neighbors) is not used to derive such inferences. This type of information can be of interest for biodiversity conservation, e.g. to understand the spatial structure and patterns of forest habitat patches. In fact, a landscape perspective is essential to understand how animal and plant populations are distributed across complex mosaics of forest habitat patches, i.e. different forest habitat types; such knowledge is, in turn, of interest to forest planners and managers (Köhl et al., 2006). Forest landscape spatial pattern can be quantified in terms of area, diversity and spatial pattern of land cover types by specific landscape metrics; examples include measures of diversity of land cover types (proportion, Shannon's diversity, dominance), total area and patchiness of habitat suitable for a particular species (patch density, mean patch size, largest patch index), spatial continuity and connectivity of important habitats (contagion).

Monitoring landscape metrics may help identifying critical changes in key landscape characteristics that might contribute to forest biodiversity loss. For example, the MCPFE–Forest Europe process includes "Landscape level spatial pattern of forest cover" (indicator 4.7) and the Montréal Process includes "Fragmentation of forests" (indicator 1.1a) as indicators for the criterion related to conservation of biological diversity in forest ecosystems.

Landscape metrics are commonly quantified on the basis of land cover thematic maps obtained from remote sensing or by multisource techniques (e.g. Riitters et al., 2004; Luque et al., 2004). Estimation of landscape metrics using sample-based data can be a viable alternative (Ramezani, 2010). This is particularly the case for forest inventory schemes based on multi-phase sampling for which large numbers of first phase samples are classified into land cover and/or forest habitat types.

Various papers demonstrate the potential for estimating landscape metrics from sample-based data; examples include the pioneering work of Hunsaker et al. (1994) and the more recent work of Kleinn (2000), Kleinn and Traub (2003), Corona et al. (2004b), Ramezani and Holm (2010), and Ramezani et al. (2010). These papers show that: (i) sampling is a cost-efficient alternative to wall-to-wall mapping for assessing landscape metrics, (ii) it is possible to derive currently used metrics or to develop new metrics from multi-phase forest inventory, both from first-phases conducted using remotely sensed imagery and from subsequent phases of field survey; (iii) time series of metrics can be derived from field-based NFIs; and (iv) some inventory-derived metrics can accommodate both the two general model approaches to landscape structure, i.e. the patch-mosaic model (Forman, 1995) and the gradient-based model (McGarigal and Cushman, 2005).

However, not all landscape metrics can be estimated from sample data, at least with unbiased estimators. In general, unbiased estimators of landscape metrics also require unbiased estimators of components such as size, number, and edge length of landscape units. Furthermore, a given sampling method may not work well in terms of cost-accuracy for all selected metrics. Hence, it may be recommended to use a combination of sampling methods such as point and line intersect sampling. Point sampling appears to be efficient for metrics involving area proportions (e.g. Shannon diversity of forest types), whereas line intersect sampling (to be centered on forest inventory sample points) is efficient for perimeter–dependent metrics (e.g. forest edge length) (Ramezani, 2010). Because sample-based assessment of landscape metrics appears to be a promising approach, further studies are needed in this area, with particular emphasis on the assessment of statistical properties of sample-based estimators of landscape metrics and the development of new metrics suitable for estimation using data from standard forest inventories.

3. Discussion

Forest ecosystems have the potential to harbor greater levels of biological diversity than any other terrestrial ecosystem (Lindenmayer and Franklin, 2002). However, forest diversity is

Table 3
Volume of deadwood by European Forest Types for selected countries. (Source: Elaboration from NFIs data collected in MCPFE/Forest Europe reporting; Reference year: 2005).

Country	Total volume of deadwood (m^3 ha^{-1}) by European Forest Types													
	1. Boreal forest	2. Hemiboreal and nemoral coniferous and mixed broadleaved-coniferous forest	3. Alpine forest	4. Acidophilous oak and oak-birch forest	5. Mesophytic deciduous forest	6. Beech forest	7. Mountainous beech forest	8. Thermophilous deciduous forest	9. Broadleaved evergreen forest	10. Coniferous forests of the Mediterranean, Anatolian and Macaronesian regions	11. Mire and swamp forest	12. Floodplain forest	13. Non-riverine alder, birch or aspen forest	14. Introduced tree species forest
Czech Republic	–	8.5	22.6	11.4	9.7	9.1	16.4	na	–	–	21.3	13.9	11.6	10.0
Slovakia	–	28.0	31.7	19.2	14.8	24.5	51.1	7.3	–	–	9.9	12.2	14.0	17.6
Denmark	–	4.6	–	4.6	5.5	4.7	–	–	–	–	11.7	8.6	–	4.5
Estonia	–	13.3	23.2	12.0	12.8	13.5	24.0	18.8	–	–	12.6	8.3	7.2	13.0
Sweden	9.0	7.2	na	8.5	13.5	14.4	–	–	–	–	4.2	9.1	7.5	4.8
Slovenia	–	8.6	16.0	14.2	11.6	11.6	13.2	11.1	–	–	–	10.9	–	–

–, EFTs not present in the country; na, EFT present in the country, data not available.

increasingly threatened with at least one tree species at risk in each country of the world (FAO, 2005). Given the urgent need for halting biodiversity loss and forest degradation, increasing efforts to implement effective programs to monitor the state of forest biodiversity is a high priority. The limited data available for biodiversity assessments is recognized as one of the major gaps that constrain selection of robust and relevant biodiversity targets, at least in Europe (EEA, 2010).

Large-scale forest inventories can contribute to better knowledge and monitoring of forest biodiversity by providing statistically sound periodic assessments of key baseline variables over large areas. According to Barrett and Gray (2010), the primary strength of such inventories is a scientifically rigorous, design-based, statistical estimation method that produces estimates of forest attributes with known sampling variability and quantifiable measurement error. This aspect is neither trivial nor negligible for biodiversity assessments that frequently rely on floristic data collected using preferential sampling schemes that do not permit quantitative comparisons of floristic diversity.

On the other hand, the weakness of forest inventories for biodiversity monitoring may include low statistical precision for small area estimates. Further, the probabilistic sampling schemes adopted by NFIs are often not appropriate to quantify changes in the abundance of rare species that are typically of interest for biological conservation (e.g. threatened or endemic species). Finally, based on the commentary discussion presented in this paper, greater attention should be paid to two additional issues:

(a) current standard forest inventories mainly provide information on tree-related compositional and structural biodiversity indicators; methods for assessing these variables are available or can be accommodated within standard forest inventory approaches (Newton and Kapos, 2002). However, estimation of tree-related diversity indices is greatly affected by the survey techniques. As a first example, forest inventories based on angle-count sampling, which are still rather widespread (Tomppo et al., 2010), are less suitable for assessing the majority of tree diversity indices; as a second example, adoption of large values for minimum dbh thresholds in fixed-radius plots has a large impact on the assessment of species (McRoberts et al., 2009);

(b) for purposes of biodiversity monitoring, use of permanent plots is crucial so that change and trends can be estimated with sufficient precision. In this regard, some consideration must be given to the manner in which stratified or first-phase sampling designs are constructed. In particular, construction of strata or first-phase sampling schemes based on themes such as forest type that may change between successive measurements should be avoided. Otherwise, when sampling intensities vary, some plots that were previously assigned to a high intensity sampling stratum but are now assigned to a lower intensity stratum may have to be dropped; similarly, plots may have to be added to strata with greater sampling intensities if strata sizes increase. In addition, if the strata change, then to which stratum should a plot be assigned for change or trend purposes, the stratum for the previous measurement or the stratum for the current measurement?

4. Final remarks and prospects for the future

From the perspective of fully acknowledging and/or further expanding the capabilities of forest inventories for biodiversity assessment and monitoring, the following points must be considered.

First, simultaneous assessment of both forestry and biodiversity related variables clearly enables forest monitoring with comparatively little additional costs (Sterba, 2008), as opposed to the considerable costs of designing and implementing separate biodiversity inventories. It is clearly beneficial to sample characteristics belonging to different aspects of forest ecosystems at the same sample points and at the same time so that the information for different attributes can be related to each other (Motz et al., 2010). For instance, this would allow relating the biodiversity dynamics of natural regeneration to changes in landscape metrics as assessed by remote sensing or in the main canopy as assessed by dendrometric field measurements. This is an important synergistic gain which cannot be emphasized enough (Motz et al., 2010).

Indeed, the improvement of forest surveys through simultaneous monitoring of forestry and biodiversity related variables is a topic of increasing interest and is regarded more favorably by the stakeholders than establishment and maintenance of separate monitoring and assessment programs. In particular, such a strategy may alleviate the expected impacts of the anticipated national austerity programs resulting from national budgetary and financial problems in many countries.

Second, sampling strategies that are typical of standard large-scale forest inventories are well developed and readily available to users. However disciplines such as conservation biology, landscape design, recreation planning, and environmental impact assessment which influence land use decisions and use forest inventory data, often still have relatively modest experience with statistical sampling; rather, they are more accustomed to case studies or the examination of purposively selected typical areas. Familiarity with only the latter approaches may lead to problems with the correct interpretation of the results arising from surveys based on probability sampling. On the other hand, exploitation of techniques such as probability sampling will bring unique benefits, facilitate effective integration among different survey sectors, and contribute to comprehensive environmental monitoring and assessment. Good statistical design is an inherently critical component of any successful monitoring and assessment program. Further, a good study design, coupled with the rigor of subsequent statistical analyses of high-quality data, emphasizes the point by Lindenmayer and Likens (2010) that biodiversity monitoring and assessment needs to be good science.

Mandated monitoring and assessment programs such as NFIs are typically large-scale and useful for producing coarse-level snapshots of forest resources conditions. From the perspective of biodiversity conservation and management issues, a key challenge is to introduce into such programs question-driven monitoring approaches as a means of eventually identifying and assessing mechanisms that influence ecosystem changes (Lindenmayer and Likens, 2010). This is far from a trivial task. Good question-setting must result in quantifiable objectives that offer unambiguous signposts for measuring progress and require a well-developed partnership among ecologists, nature conservationists, statisticians, resource managers and policy-makers (Lindenmayer et al., 2008; Gibbons et al., 2008). Related key ingredients are transparency and the potential for unrestricted use of raw data, which would likely bring new findings and stimulate research and management questions, and may also be one of the primary ways for effectively uncovering errors and data artefacts (Lindenmayer and Likens, 2010).

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