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Indicators of biodiversity and ecosystem services: a synthesis across ecosystems and spatial scales

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According to the Millennium Ecosystem Assessment, common indicators are needed to monitor the loss of biodiversity and the implications for the sustainable provision of ecosystem services. However, a variety of indicators are already being used resulting in many, mostly incompatible, monitoring systems. In order to synthesise the different indicator approaches and to detect gaps in the development of common indicator systems, we examined 531 indicators that have been reported in 617 peer-reviewed journal articles between 1997 and 2007. Special emphasis was placed on comparing indicators of biodiversity and ecosystem services across ecosystems (forests, grass- and shrublands, wetlands, rivers, lakes, soils and agro-ecosystems) and spatial scales (from patch to global scale). The application of biological indicators was found most often focused on regional and finer spatial scales with few indicators applied across ecosystem types. Abiotic indicators, such as physico-chemical parameters and measures of area and fragmentation, are most frequently used at broader (regional to continental) scales. Despite its multiple dimensions, biodiversity is usually equated with species richness only. The functional, structural and genetic components of biodiversity are poorly addressed despite their potential value across habitats and scales. Ecosystem service indicators are mostly used to estimate regulating and supporting services but generally differ between ecosystem types as they reflect ecosystem-specific services. Despite great effort to develop indicator systems over the past decade, there is still a considerable gap in the widespread use of indicators for many of the multiple components of biodiversity and ecosystem services, and a need to develop common monitoring schemes within and across habitats. Filling these gaps is a prerequisite for linking biodiversity dynamics with ecosystem service delivery and to achieving the goals of global and sub-global initiatives to halt the loss of biodiversity.

"Over the past 50 years, humans have changed ecosystems more rapidly and extensively than in any comparable period of time in human history, largely to meet rapidly growing demands for food, fresh water, timber, fibre and fuel. This has resulted in a substantial and largely irreversible loss in the diversity of life on earth". This first major finding of the Millennium Ecosystem Assessment (MA 2005a) clearly links the substantial and widespread loss of biodiversity on Earth to the growing intensity of many anthropogenic pressures on biodiversity. According to the MA's Biodiversity Synthesis (MA 2005b), the most important direct drivers of biodiversity loss and ecosystem service changes are habitat change (such as land use changes or physical modification of rivers), climate change, invasive alien species, overexploitation, and pollution. Hence, biodiversity loss is linked to "the degradation of many ecosystem services [and] could grow significantly worse during the first half of this century [. . .]" (MA 2005b).

In 2004, a pan-European initiative called 'Streamlining European 2010 Biodiversity Indicators' SEBI 2010 was launched to develop a European set of biodiversity indicators (Balmford et al. 2005, EEA 2007) for monitoring the status and trends in biodiversity, namely its rate of loss. The authors clearly stated the need for indicators "[. . .] of biodiversity and ecosystem functions and services that are rigorous, repeatable, widely accepted and easily understood". The initiative is linked to the global Convention on Biological Diversity (CBD), which already listed eleven 'global indicators for assessing progress towards the 2010 target' (UNEP/CBD/COP7 2003). Both the CBD and SEBI 2010 have created global awareness for the need of (novel) multiple biodiversity indicators that can easily be communicated to decision makers and practitioners, and both initiatives have already suggested sets of current and potential biodiversity indicators at regional to global spatial scales with respect to the goal 'to halt the loss of biodiversity by 2010'.

Despite these considerable achievements, the question remains whether these indicator sets will suffice to achieve the goal of developing efficient monitoring of biodiversity schemes. In common with other indicator systems (e.g. listing of threatened species), the different proposed sets mainly comprise biotic indicators tracking trends of biodiversity components in time (status and trend indicators). However, if a negative trend becomes obvious indicators should help to associate this trend with the potential causes (drivers, pressures). Once the causes are identified, specific actions and policies (response) should aim at adaptive habitat management concepts. This sequence follows the driver–pressure–state–impact–response (DPSIR) scheme (EEA 1999). Although the scheme represents a rather simplified and linear model of the linkages of socio-economic activities and their environmental impact, it can be helpful to identify and classify indicators that link between societal and environmental aspects of biodiversity. With a prevailing focus on status and trend indicators, however, indicator systems do not integrate both aspects appropriately, which is likely to significantly hinder a timely policy response to reverse negative biodiversity trends.

In their comprehensive analysis of mistakes frequently encountered with respect to biodiversity indicators in forest ecosystems, Failing and Gregory (2003) stressed also the fundamental need to define 'endpoints' of indication, i.e. the ultimate purpose (or goal) and information provided by the indicators. In other words: the purpose of indication strongly determines the type of indicator needed to address a problem and the spatial scale of application. If, for example, the vitality of a population of a threatened species at the landscape scale is the aim of monitoring, genetic indicators (e.g. the level of heterozygosity) and landscape patterns should be monitored. Moreover, preserving biodiversity at a level needed to sustain ecosystem services is likely to be very different from halting the loss of biodiversity and probably requires different 'toolboxes' of indicators. However, the suitability and coverage of present indicators for ecosystem service evaluation is largely unknown.

In this context, our study reviews and compares the purpose and application of indicators across different terrestrial and aquatic ecosystem types comprising natural and managed systems. Based on an extensive literature review of more than 600 peer-reviewed papers published between 1997 and 2007, we compare the availability and characteristics of indicators in forests, grass- and shrublands, wetlands, rivers, lakes, agro-ecosystems and soils. Furthermore, we assess the degree to which indicators and their application tend towards specific ecosystems and spatial scales. Finally, we assess the degree to which indicators focus on biodiversity and/or capture ecosystem services. From this analysis, we identify and discuss components related to biodiversity and ecosystem services that are largely underrepresented by current indication approaches. Hence, our analysis represents an attempt to synthesise the different efforts towards the development of a broad range of indication systems across multiple ecosystems.

Analytical framework and data analysis

An analytical framework was developed to define systematic criteria for the analysis and comparison of a wide range of indicators across organism groups and habitats. We considered five criteria to characterise and classify each indicator described in the literature: purpose of indication, indicator type, spatial scale, biodiversity component(s) addressed and, eventually, ecosystem service(s) addressed. These criteria are summarised in Table 1 together with the rationale and categories used in this study.

The general purpose of indication was classified into broad categories (e.g. ecosystem quality assessment, monitoring, biodiversity evaluation) given the broad range of indicators and ecosystem types covered by our study. The classification of indicator types was defined according to Noss (1990) and Pioani et al. (2000), and was based principally on the discrimination of biotic and abiotic parameters. Biotic indicators (i.e. indicators referring to organisms: indicator species, species traits, etc.) are per se required to assess the status and trends of biodiversity, while abiotic measures (i.e. indicators referring to the non-organismic environment: physical, chemical, area, etc.) are potentially useful for detecting and quantifying the level of environmental stress or disturbance impacting the ecosystem/habitat. Spatial scale refers to the geographical area in which the indicator is measured in general, ranging from the (fine) patch scale to the global scale. The measurement scale, however, may be different from an indicator's potential scale of application (not considered in this study), which is predominantly controlled by the scale of data coverage. For instance, a richness measure at the patch scale (single farm) may be statistically upscaled to a regional mean richness value, if sufficient data is available. Biodiversity (=biotic) indicators were further distinguished, if they referred to structural (e.g. canopy cover and the amount of dead wood in forests), functional (e.g. trait richness and abundance, functional diversity) or genetic (e.g. heterozygosity) components of diversity. Finally, for attributing ecosystem services we followed the Millennium Ecosystem Assessment classification (MA 2005a) into 'provisional', 'regulating', 'cultural' and 'supporting' services (see Table 1 for a definition of these categories).

As the purpose of our study was to review existing indicators (which presumes the 'purpose to indicate' in the reviewed body of literature), we aimed at references of indicators sensu stricto, i.e. studies that clearly involved a purpose to indicate and ideally considered criteria of indicator suitability, such as reference to ecological theory, monotonic response to environmental impact and transparency to policy and decision makers (McGeoch 1998, Fairweather 1999). Judgement on indicator suitability,

Table 1. Criteria and categories defined to set up the review database and to analyse the indicators across ecosystem types.

Criteria	Rationale	Categories considered
Purpose or field of application	The motivation, or endpoints, for indicator development.	Habitat quality assessment, monitoring, biodiversity, water quality assessment, ecological status assessment, restoration/management evaluation, ecosystem function assessment, response to natural disturbance (e.g. fire, flood) assessment, predictive modelling, economic valuation.
Type of indicator	Indicators may be abiotic (e.g. soil properties and landscape structure) or biotic (e.g. species richness and functional traits composition).	Biotic: genetic, species, community, function, structure, sensitivity, traits. Abiotic: physical, chemical, biochemical, habitat area, fragmentation.
Spatial scale	Scale(s) at which the indicator is used. Potential scale of application may differ.	Patch ('farm'), local, regional, national, sub-global and global scale. (Fine-scale indicators may be statistically scaled up to broader scales, which depends on statistical scalability of the indicator and the scale of data coverage.)
Component of biodiversity assessed	Biodiversity is constituted of different components.	Genetic, structural, functional biodiversity (unlike the indicator type category, we specifically looked at the biodiversity component estimated by an indicator).
Ecosystem services assessed	The Millennium Ecosystem Assessment (MA 2005a) defines four different classes of ecosystem services: Provisioning: products obtained from ecosystems (e.g. food, fibre, fresh water) Regulating: benefits obtained from the regulation of ecosystem processes (e.g. air quality, climate and water regulation) Cultural: nonmaterial benefits people obtain from ecosystems (e.g. spiritual enrichment, recreation) Supporting: services necessary for the production of other ecosystem services (e.g. nutrient and water cycling, photosynthesis)	Food/water supply, fibre/energy supply, genetic/ornamental resources, air/water/climate/erosion regulation, pest/disease/hazard regulation, pollination, waste treatment/self-purification, primary production, soil formation, decomposition, nutrient/water cycling

however, was beyond the purpose of this study. Hence, our analytical framework considered references that 1) explicitly involved the purpose to indicate and that 2) provided the information on the criteria listed in Table 1. The analytical framework was applied to publications listed in the Science Citation Index Expanded (SCIE) between January 1997 and May 2007. Altogether, 617 references on 531 indicators were reported back from the SCIE on the seven ecosystem types (Fig. 1a). The relevant information on the five criteria mentioned above was collated in an indicator-by-criteria matrix. Each indicator was added as an entry to the database. As several ecosystem types, however, shared common indicators, the total number of entries exceeds the total number of indicators in the database. It is obvious that the restriction to the SCIE inevitably excluded all non-peer-reviewed literature but scientific validations of the indicators was considered an important pre-requisite for the development of sound monitoring schemes. This was partly solved by an attempt to account for studies and reports at regional and broader scales (UNEP/CBD/COP7 2003, EEA 2007). Similarly, the set of keywords used for the literature survey might have influenced our survey. Nevertheless, we decided to follow a systematic and repeatable approach that can be compared and improved depending on the research question. In this sense, as the purpose of this study was to compare indicators across ecosystem types, a standardised assessment with a common set of keywords was considered the most appropriate approach.

Criteria that apply to an indicator were coded '1' and all others '0' in the final binary indicator-by-criteria matrix. Multivariate analysis was used to summarise and visualise the general structure of relationships among multiple indicators, indicator criteria and ecosystem types, as ordination plots were considered a useful tool to display the multiple relationships. We applied detrended correspondence analysis (DCA) to the binary matrix of indicator-by-criteria and principal component analysis (PCA) to another matrix of relative proportion (% values) of indicators that meet a specific criterion. All multivariate analyses were run using the software package CANOCO (ver. 4.5, ter Braak and Smilauer 2003).

Indicator patterns

Our database revealed different patterns in the use of indicators and indication approaches as reported in the literature. Regarding the ecosystem type, indicators of river and soil ecosystems are more frequent (Fig. 1a). Both are represented with >150 entries each and, together, account for 289 indicators (54.4%). The general purpose of indicator development and application was habitat and ecosystem quality assessment, monitoring, and biodiversity assessment (65.5, 43.9 and 32.4% of entries, respectively) (Fig. 1b). Fewer publications address the indicators of natural disturbance (e.g. fire, storm) (8.3%), ecosystem and habitat restoration and management (8%) and prediction of environmental impacts on ecosystem function and structure (3.1%). Only eleven entries (2.1%) refer to the valuation of ecosystem services.

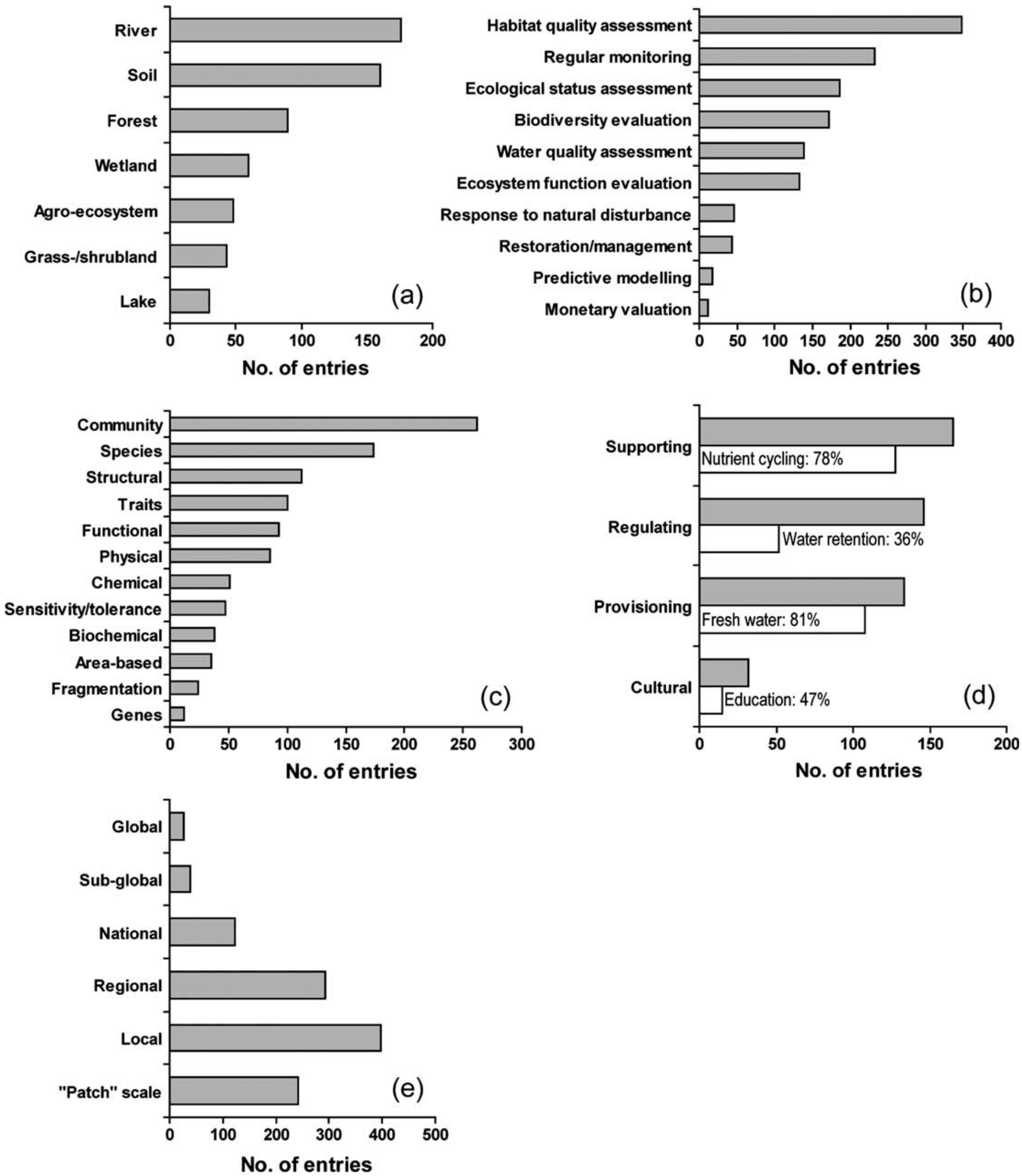

Figure 1. Number of indicator entries in the database per (a) ecosystem, (b) purpose of indication, (c) indicator type, (d) ecosystem service categories according to the MA (2005a) (a)–(d in decreasing order) and (e) spatial scale (ordered along a spatial gradient). Panel (d) also contains the no. of entries of the most dominant service per service category (empty bars). The sum of entries may exceed the total number of indicators in the database (531) due to multiple selections.

Regarding indicator type, the vast majority of indicators represent biological attributes either based on taxonomic identity of species or communities (32.8% and 49.3 respectively) (Fig. 1c). Jointly, these two indicator types are frequently reported across all ecosystem types, yet the majority was found for river, soil and forest ecosystems. Frequently used community measures include taxonomic composition, richness and diversity of an assemblage or part of it, while the individual species level is most often addressed by particular sensitive or tolerant indicator species, and umbrella and keystone species. Among biotic indicators about 30% of indicators represent structural and functional measures. Structural measures refer to the temporal (e.g. age structure) and spatial organisation of an assemblage. These were most frequently reported in forest, agro- and river ecosystems (17.6, 16.0 and 13.2%, respectively). Functional indicators are most frequently addressed in forest, soil and river ecosystems (13.7, 13.6 and 11.6%, respectively) and are often linked to the application of ecological traits (direct measures of an biological or ecological characteristic, e.g. growth form, size, life history, reproduction measures and feeding types). In some cases indicator types overlap in the sense that references do not provide sufficient information for a clear assignment. This particularly applies to the group of multimetric indices, which combine the results of several single metrics into a composite measure. They may incorporate measures at the species, community and functional level and have been widely applied in freshwater ecosystems for decades (Karr 1981).

Among the non-biotic indicators, physical measures constitute 16% of total indicators and are particularly applied in agro-ecosystems and rivers (14.8 and 11.6%, respectively). We found only a few chemical, biochemical and landscape indicators that were reported to be indicative of biodiversity. It should be stressed, however, that several hundreds of landscape-level metrics that are readily available with software tools, such as FRAGSTATS, and some of which may be potentially useful indicators, were not considered for this study. The smallest number of indicators was reported for the genetic component of biodiversity.

The vast majority of indicators reviewed for this study range between the (finest) patch scale and the (intermediate) regional scale (200–400 entries; Fig. 1e). Sub-global and global scales are addressed by only 48 indicators (9% of total indicators). Overall, biotic and abiotic indicators differ in the spatial scale at which they are measured and to which they are applied, respectively. Biotic indicators are often measured and applied at local to regional scales, whereas abiotic indicators, such as physical, area and fragmentation measures, rather refer to regional (landscape) and broader scales. The inherent association between indicators, indicator type, scale, and ecosystem type is illustrated with Fig. 2. Along axis 1, biotic indicators (left hand side) separate from abiotic indicators (right hand side) based on their indicator type characteristics and usage in different ecosystems. Abiotic indicators are frequently used in agro-ecosystems at the broad scale, while biotic indicators dominate in the other ecosystem types at finer scales. A further separation of biotic indicators into rather functional (including biochemical and physiological sensitivity measures) and rather structural measures is evident along axis 2 of Fig. 2. Ecosystem type centroids are located in the lower half of the plot and illustrate that – according to our literature survey – structural indicators dominate over functional ones in all ecosystems.

Roughly one third of the indicators address biodiversity assessment, while a general lack of indicators is obvious for genetic diversity; less than 5% of all biodiversity indicators referred to the genetic component (Table 2). The same is true for the functional component, which is rarely addressed except for rivers and forests. Species and community diversity are frequently reported by studies on all ecosystems and ranged between 21 and 88%, except for soils (6%). To a lesser extent, this was also true for structural biodiversity measures (3–45% of the total biodiversity measures) with the exception of wetland and soil ecosystems, where they are particularly rare. A detailed examination of indicators of biodiversity revealed that only slightly more than 40% of 'biodiversity' indicators directly address biodiversity, i.e. the diversity of biological attributes, such as species, groups of species, community structure or function. The majority tend to refer indirectly to biodiversity and often apply landscape-scale surrogate measures, such as habitat area, management parameters and fragmentation measures.

Indicators for ecosystem services are numerous with the exception of the cultural services category (Fig. 1d). The ecosystem-specific analysis revealed that different services and groups of services are closely associated with particular ecosystem types (Table 3). For instance, nearly 80% of the indicators referring to provisioning services (all ecosystem types) address the provision of fresh water. This finding was probably largely a consequence of the high number of indicators for river, lake and wetland assessment and monitoring, most of which ultimately address the supply of fresh water. The association of services and ecosystem types is illustrated by a PCA based on the relative proportions of service indicators per service category and ecosystem type (cultural services omitted). The ordination biplot (Fig. 3) in particular reveals the relevance of indicators of regulating and supporting services across ecosystems. Among the provisioning services, the provision of (fuel) wood in forests and fresh water in lakes and rivers are frequently addressed. However, it can be assumed that provisioning services with direct market values are rather

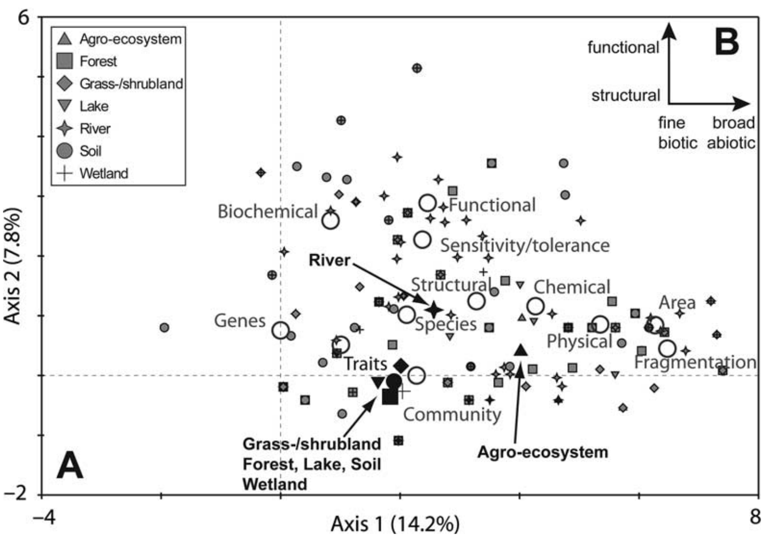

Figure 2. (A) Detrended correspondence analysis (DCA) of 531 indicators classified into seven ecosystems (see legend for symbols) and 12 indicator types (○). Arrows point at ecosystem centroids (larger black symbols). (B) DCA axis 1 synthesises indicator types and spatial scales, while functional and structural indicators distinguish along axis 2 (explained variance given in brackets).

Table 2. Overview of the status of biodiversity indicators in the database. The first column refers to the number of biodiversity indicators per ecosystem type, which is then divided into five components of diversity in the columns on the right. Total no. of indicators = 531. Genetic indicators refer to any measures that address single genes or alleles. Typical measures are the number of genotypes or the level of heterozygosity. Species' diversity is frequently referred to as species richness and reflects the diversity of the taxonomic composition. In contrast to community measures, species' diversity is often related to a limited group of species of interest. Community diversity measures account for an entire community, which is frequently addressed by diversity indices such as the Shannon–Wiener and Simpson diversity index. Another subdivision allows for the separation of structural and functional components of biodiversity. The structural component is reflected by any measure that refers to the spatio-temporal structure of a community or part of it. For instance, the growth forms of different plant species in forest and grassland ecosystems address spatial structuring, while the different life cycles reflect the temporal structure. Finally, the functional component of biodiversity is related to the diversity of ecosystem functions that are covered by, for instance, a community. This might be the diversity of feeding types within the community of leaf litter decomposers in a river ecosystem. As multiple entries were possible for the allocation of indicator types, the values of column 2 may deviate from the sum of columns 3–7.

Ecosystem	No. of biodiversity indicators (% of total indicators)	No. (%) of biodiversity indicators per indicator type				
		Genes	Species	Community	Structural	Functional
Forest	53 (58.8)	1 (1.9)	31 (58.5)	5 (9.4)	12 (22.6)	4 (7.5)
Grass-/shrubland	27 (62.8)	1 (3.7)	13 (48.1)	16 (59.3)	4 (14.8)	1 (3.7)
Wetland	24 (40.0)	1 (4.2)	12 (50.0)	16 (66.7)	– (0)	– (0)
River	40 (22.7)	2 (5.0)	30 (75.0)	28 (70.0)	18 (45.0)	10 (25.0)
Lake	15 (50.0)	1 (6.7)	10 (66.7)	11 (73.3)	2 (13.3)	– (0)
Soil	34 (21.3)	5 (14.7)	2 (5.9)	30 (88.2)	1 (2.9)	3 (8.8)
Agro-ecosystem	28 (58.3)	1 (3.6)	9 (32.1)	6 (21.4)	9 (32.1)	– (0)

addressed by direct measures, such as the number, mass or volume of food or fuel that is provided. These metrics are not likely to be well represented in this study due to the focus on direct references to indicators in the peer-reviewed body of literature.

The predominant regulating service is water retention (35% of all entries in that category) and is frequently addressed by studies on forest and grass-/shrubland ecosystems. The majority of soil and wetland indicators refer to the supporting service of nutrient cycling (78% of indicators in this service category; 86% together with water cycling). Decomposition, a function that is intrinsically linked to the supporting service of nutrient cycling, is also frequently considered by studies on soils and wetlands (Fig. 3): 76% of indicators of decomposition originate from soil ecosystems. The provision of fibre and fuel (energy) is particularly linked to indicators in forest and grass-/shrubland ecosystems. The total number of indicators of this service, however, was low (14 = 10% of indicators of provisioning services). Among the cultural services (not considered in Fig. 3), most indicators relate to education and knowledge systems, such as some keystone and umbrella species or the amount of dead wood in forests and rivers (both are widely used for environmental education purposes), ancient grassland species or physical structure of rivers (47% of indicators in that category), followed by recreation (31%). However, taking all indicators into consideration, only 6% refer to cultural services.

Discussion

Our synthesis reveals that the purposes, or endpoints sensu Failing and Gregory (2003), for the use of indicators can be as manifold as are the different policy demands on monitoring and indicator types. In order to streamline future indicator development and application, some effort should be spent on the identification of major endpoints regarding large-scale initiatives to 'halt the loss of biodiversity' and 'to maintain ecosystem services'. In this sense, a clearer definition of endpoints at relevant spatial scales is likely to stimulate further indicator research. Moreover, well-defined endpoints would help to set quality criteria for data collection and compilation. Nevertheless, an overwhelming amount of indicators have already been developed for the purposes of ecosystem (habitat and quality) assessment and monitoring and for biodiversity assessment. In other words, many existing indicators aim to assess status and trends in biodiversity and ecosystem integrity, but not ecosystem services directly. A clear and demonstrated linkage between biodiversity, system integrity and ecosystem services, however, is being required to render indicators suitable for service status and trend assessment.

Whether simple biodiversity measures (e.g. taxonomic richness) are sufficient to measure or predict the complexity and multifaceted components of biodiversity (Diaz and Cabido 2001) remains an area of great concern (MA 2005a). Our results clearly reveal, despite the knowledge

Table 3. Number of indicator entries in the database per ecosystem service category (% of total number of service indicators per ecosystem given in brackets).

Ecosystem	Provisioning	Regulating	Cultural	Supporting
Forest	12 (16.9)	25 (35.2)	19 (26.8)	15 (21.1)
Grass-/shrubland	12 (19.7)	24 (39.3)	8 (13.1)	17 (27.9)
Wetland	21 (24.7)	32 (37.7)	3 (3.5)	29 (34.1)
River	84 (78.5)	8 (7.5)	7 (6.5)	8 (7.5)
Lake	18 (58.1)	4 (12.9)	3 (9.7)	6 (19.3)
Soil	4 (2.1)	76 (40.0)	– (0)	110 (57.9)
Agro-ecosystem	7 (21.2)	8 (24.2)	5 (15.2)	13 (39.4)

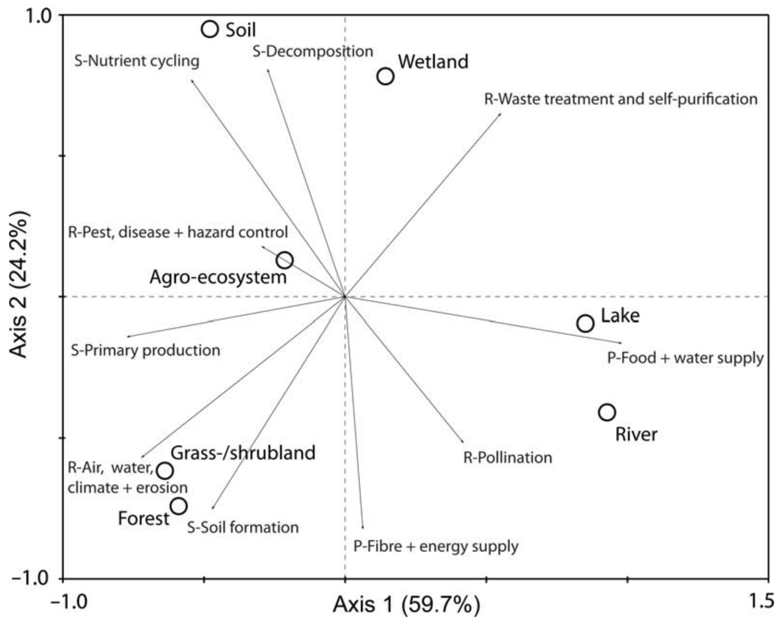

Figure 3. Principle components analysis (PCA) of ten groups of ecosystem services and seven ecosystems using the relative proportion of classified indicators per group of services and ecosystem (sum of all groups of services = 100% per ecosystem). Groups of services refer to similar single services and start with 'P-' for provisioning, 'R-' for regulating and 'S-' for supporting services. Since less than 3% of indicators refer to cultural services, this group is omitted (variance explained by axes given in brackets).

that biodiversity has multiple components, the still prevailing role of richness measures in biodiversity assessment at local to regional spatial scales. In contrast, functional and structural aspects remain almost unstudied, irrespective of spatial scale and ecosystem type. This reflects the fact that biodiversity assessment and monitoring until recently has been mainly driven by conservation biologists. Moreover, functional and structural indicators are often considered difficult to measure and to interpret over broad areas (Gustafson 1998, Turner 2005, but see Lavorel et al. 2008).

While important (valuable) provisional services (e.g. water, food and energy supply) are frequently addressed in the reviewed body of literature, other services (e.g. self-purification in rivers, aesthetic values in grasslands, recreation at freshwater sites, or nutrient retention in wetlands) are notably rarely mentioned – although they possess a considerable economic value (Gren et al. 1995, Costanza et al. 1997, Bolund and Hunhammar 1999, Everard 2004). There is a clear lack of formal investigation and application of indicators to detect, or even measure, status and trends in these services (but see Quétier et al. 2007, Diaz et al. 2007a). Moreover, measures should be developed and applied to indicate the environmental (and human) impact on such often non-market ecosystem services. On the response side indicators are required to indicate whether policies and related management actions have the desired effect on both service maintenance and biodiversity.

The significance of biodiversity and ecosystem service indicators could be enhanced if they could be related to ecosystem-specific or regional reference values and be expressed as the deviance from this benchmark (O'Connell et al. 2000, Carlisle et al. 2008). Such a relative approach would improve comparability across ecosystem types and regions, although little is known up to now on the identification and setting of these threshold values (but see Carey et al. 2002 for an example). A better understanding of the linkages between biodiversity and ecosystem functions and processes would be necessary to define such thresholds (Srivastava 2002, Hooper et al. 2005, Diaz et al. 2006).

To what extent can present indication approaches help to implement biodiversity policies in different ecosystems? The European Environment Agency has recently proposed a first set of indicators to monitor progress with respect to SEBI 2010 (EEA 2007). Although the set comprises as many as 26 indicators, only a few directly refer to biotic measures, while the majority address landscape area and fragmentation, usable stocks and monetary values, for which data is widely available at national to regional levels. This illustrates the general trend in large-scale biodiversity assessment (UNEP/CBD/SBSTTA10 2004) to use aerial and other surrogate measures instead of direct measures of biological diversity (Levrel et al. 2007). Yet, the applicability of those surrogates to account for biodiversity and to address the different components of biodiversity – and ultimately to address ecosystem services – remains questionable as long as their linkage to actual biodiversity levels is not well-grounded and validated. In a recent European study on indicators of biodiversity in agricultural landscapes, Billeter et al. (2008) focused on species richness in several taxa and related them to landscape structure and management in agro-ecosystems. The authors conclude "[. . .] that indicator taxa are unlikely to provide an effective means of predicting biodiversity at large spatial scales [. . .]", although they report a clear link between total species numbers, landscape structure and land use intensity.

Consistent with these observations our review shows a pronounced scale-dependent difference in indicator availability and application. More than three fifths of indicators directly account for biological attributes or components of biodiversity (e.g. indicator species, taxonomic richness and diversity) at a fine spatial scale (Lawton 1999). Biotic indicators were more frequent in patch-scale than in regional-scale studies, while abiotic measures dominated studies at the broader sub-global and global scale (Fig. 2). With present indication approaches, it might often prove difficult to upscale biotic indicators and relate them to broader-scale biophysical measurements. This significantly limits the ability to assess biotic conditions, processes and ecosystem functioning across national and continental scales and is likely to affect the feasibility of the objectives of sub-global and global biodiversity policies. However, our survey also identified comparatively new indication approaches that are potentially suitable to fill this gap in broad-scale indication.

First, the application of population genetics in landscape ecology is certainly a promising approach, since population genetics offers a significant potential to expand the scale of biological indicators and to address broad-scale questions – including those related to habitat fragmentation and landscape pattern (Mech and Hallett 2001, Michels et al. 2001, Manel et al. 2003). This approach may better link biotic to abiotic indicators such as habitat composition, connectivity and area and fragmentation, which can be estimated through remote sensing (Hagan and Whitman 2006, Ponge and Chevalier 2006). Moreover, according to Larsson (2001) several biotic indicators based on structural and functional measures (e.g. tree species richness, tree stand structural complexity, amount of dead wood and litter) are scaleable across different spatial scales. Most often, scaleable indicators are measured at finer scales (resolution) and then integrated and projected to larger scales (application). Brotons et al. (2004), Helm et al. (2006) and Petit and Firbank (2006) provide examples for grassland and shrubland ecosystems, Brown and Vivas (2005) and Dupigny-Giroux (2007) for wetlands, Svoray and Shoshany (2004) and Ludwig et al. (2007) for soils and Statzner et al. (2007) for rivers. All these results suggest that a set of carefully selected biotic indicators (including genetic, structural and functional measures) could potentially be developed further to fill the gaps in the spatial scales of common indication systems. The limited application of biological indicators at broad scales may further result from the lack of well-defined indication endpoints at national to continental scales, and the lack of consistent sampling protocols. Therefore, national and continental assessments of biological diversity might further require concerted indication approaches and the application of spatially extensive monitoring or survey designs (Hunsaker et al. 1990, Parr et al. 2003, Larsen et al. 2007). The development and application of biotic indicators applicable across different spatial scales or of sets of indicators connectable across spatial scales will be an important task for the future.

Another approach gaining increasing interest for the indication of biodiversity and ecosystem functioning across habitats and spatial scales is the use of functional traits (Bady et al. 2005, Statzner et al. 2005, de Bello et al. 2006, Petchey and Gaston 2007). The advantages in using traits are manifold, since they often directly refer to ecosystem functions and processes and to the underlying biodiversity patterns (Hodgson et al. 2005, Diaz et al. 2007a).

Much effort has been spent on the development and application of functional approaches for the assessment of biodiversity changes and ecosystem effects in grassland (Tilman et al. 2001, Diaz et al. 2007b, Quétier et al. 2007) and other managed ecosystems (Decocq et al. 2004, Balvanera et al. 2005, Statzner et al. 2007, Kremen et al. 2007). Traits are frequently used to assess the impact of different management schemes, such as fire, grazing and mowing in grassland ecosystems (Kahmen and Poschlod 2004, Díaz et al. 2007b, Quétier et al. 2007), forest ecosystems (Graae and Sunde 2000, Myking 2002) and agro-ecosystems (Döring and Kromp 2003, Balvanera et al. 2005). The use of traits is not spatially limited since some traits, such as growth form, vegetation height, and cover or leaf area and nitrogen content, can be derived from remote sensing. The benefit of traits in other ecosystems is now being assessed, as for example, in large-scale river quality assessments, which have recently been reported by Feld and Hering (2007) and Dolédec and Statzner (2008). Unlike taxonomic entities, traits are largely independent of biogeographic regions (Baird et al. 2008). Similar to any broad scale application of indicators, a systematic sampling design and consistent set of measures is critical to the application of traits (Cornelissen et al. 2003, Lepš et al. 2006). Nevertheless, an increasing effort on future trait research is unlikely to replace taxon-based approaches (e.g. species' records), as both provide complementary information on impacts of environmental change and consequences for ecosystem services (Flynn et al. 2009).

Conclusions

The loss of biodiversity on Earth is considered to be a major threat to ecosystems and human well-being (MA 2005b). To halt the loss of biodiversity, international and national policies have been launched at global, continental and regional scales. However, these policies do not yet seem to have adequately stimulated the development of comprehensive indicator systems suited to detect and measure the state and trends in biodiversity and their implication on ecosystem service provision. To close the gaps, considerable effort will be required to streamline future ecosystem indication. A comprehensive and standardised design for sampling and data generation will be needed to permit comparisons across different areas and ecosystems. These data would help to develop biodiversity indicators that are directly linked to the genetic, species, population and community level, and that cover the different components of diversity at these organisational levels. Abiotic surrogate measures derived from remote sensing and spatial analyses must be validated in their linkage to the biota by ground truthing. The identification and setting of reliable thresholds for ecosystem services could help to maintain the services at a level required by society. Therefore, better knowledge is needed on the linkage of biotic and landscape features. This particularly refers to the functional component of biodiversity, which is supposed to be fundamental to the provision of ecosystem services (Díaz et al. 2007a, but see Srivastava and Vellend 2005 for a critical review).

At the European scale, two policies already showed impressively their stimulating and streamlining effect on the development of indicators. Since 2000, the Water Framework Directive (2000/60/EC) has led to the development of hundreds of new indicators to assess the ecological quality of surface water bodies in Europe. Tens of billions of Euro have been spent in the European Framework Programmes 5 and 6 to fund related projects. A similar stimulus in the field of soil assessment and monitoring can be anticipated from a European Soil Protection Directive (EC 2006). Accordingly, a 'Biodiversity and Ecosystem Service Directive' would provide an appropriate framework to fill the gaps in biodiversity and ecosystem service assessment outlined above.

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