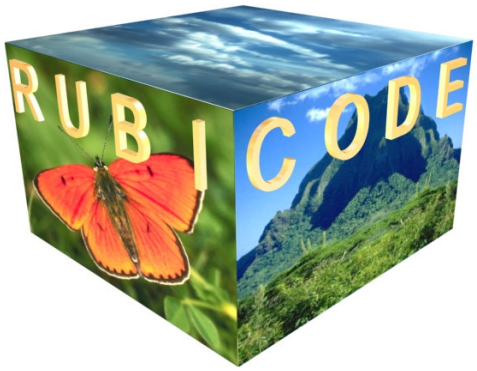

The **RUBICODE** Project

Rationalising Biodiversity Conservation in Dynamic Ecosystems

Funded under the European Commission Sixth Framework Programme (Contract Number: 036890)

Assessing and monitoring ecosystems – indicators, concepts and their linkage to biodiversity and ecosystem services

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Preliminary note

This deliverable is a draft journal manuscript, which compares indication and assessment approaches between ecosystems. It will be submitted by the end of 2007. The final manuscript will cover the following ecosystem types: forests, grass- and shrublands, wetlands, rivers, lakes, agro-systems, soils and landscapes. This deliverable includes all ecosystems except lakes, which will be added later to the final manuscript. This will include further contributions from other workpackages as well as from external experts. The basic data for all eight ecosystems have been collected and references are listed in Annex 2.

At present, this review provides and inventory of indication approaches and indicators developed and applied in the reviewed ecosystems. This includes also the comparison of indicators, motivations and approaches among the ecosystems. Yet, at present, the draft review does not provide deductive remarks and recommendations for the future development and application of indicators. This will be added to the final manuscript.

Document history

The deliverable is based on discussions and preliminary data evaluations of RUBICODE's Workshop 4, which was held in Essen (Germany) from 27 Feb to 1 Mar 2007. Following the agreement of workshop attendees, the criteria for an extensive literature research in the Web of Science, Science Citation Index Expanded (SCIE) were defined and used to identify suited references. The references were transferred into a database and currently sum up to more than 630 publications referring to more than 550 indicators (incl. lakes). While the structure of the database was defined by the workpackage leaders (WP4: Indicators), the population of the database was performed by experts for the ecosystem types included, individually by Christian K. Feld (rivers), Leonard Sandin (lakes), Isabel Pardo and Owen Mountford (wetlands), Rob Bugter (agro-ecosystems), Meelis Partel (grasslands), Francesco de Bello and Sandra Lavorel (shrublands), Ulf Grandin (forests), Pedro Martins, Jörg Römbke and Paulo Sousa (soils) and K. Bruce Jones (landscapes).

The evaluation and final analysis of the database has been carried out by the workpackage leaders, whereas the ecosystem leaders provided and contributed to the ecosystem-specific reviews of Chapter 3.

Daniel Hering and José Paulo Sousa, October 2007

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0. Summary

The Millennium Ecosystem Assessment raised the need for indicators, particularly such that are linked to biodiversity and its role to sustain ecosystem services. According to the MA's Biodiversity Synthesis the most important direct drivers of biodiversity loss and ecosystem service changes are habitat change (such as land use changes, physical modification of rivers or water withdrawal from rivers), climate change, invasive alien species, overexploitation, and pollution.

Current indicators are subject to a couple of drawbacks: i) they do not fully cover the major components of biodiversity, i.e. genetic, structural and functional biodiversity; ii) they do not cover all spatial scales needed; iii) they do not sufficiently address ecosystem function and processes; iv) they do not account for ecosystem specificity; v) they do not sufficiently distinguish between managed and natural ecosystems and biodiversity.

Based on a review of 617 references in the *Science Citation Index Expanded* (SCIE, time span: 1997–May 2007), altogether 534 indicators of six ecosystems (forests, grasslands and shrublands, agro-ecosystems, soils, wetlands and rivers) and landscapes (landscape-level indicators) have been transferred into a database. Seventy-two indicator characteristics, classified into nine groups, are used to analyse and compare indicators among ecosystems. In addition, seven reviews (six ecosystems + landscapes) address the history of indication, indicator types, spatial scale, purposes, policies and standardisation/validation. The reviews provide detailed insight into the development of indicators and indication systems for each ecosystem and at the landscape level.

Many early forest indicator systems were designed to indicate forest stand condition and productivity. At present, forest indicators are available for an array of purposes: measures of forest productivity, indicators for conservation purposes and the evaluation of ecosystem services. Recommendations for forest monitoring suggest using several structural and compositional key indicators, whereas compositional measures should be functionally linked to a broad range of species. Moreover, biodiversity indicators should account for several groups of species, such as keystone species, umbrella species, dispersal-limited species, resource-limited species, process-limited species and flagship species.

A widely-applied example for forest indication is the 'Ellenberg system', which combines indicator values referring to plant species' ecological and physiological optima. Besides, the use of species traits has gained popularity over the last decade. They may contribute to the still patchy understanding of biodiversity and biodiversity indications.

Grasslands and shrublands are semi-natural ecosystems traditionally maintained by low to intermediate management or disturbance events. Many indicators of grasslands and shrublands have

not been exclusively developed for this ecosystem. Bioindication of grasslands and shrublands has been traditionally applied at comparatively small spatial as well as temporal scales, i.e. at the patch or community level. Measures of grazing, mowing and fuel production intensity are among the most relevant indicators of management and disturbance in grassland and shrubland habitats, reflecting different processes that cause biomass loss and regeneration. Climatic trends and nutrient deposition are routinely monitored Europe-wide, as part of standard environmental programs.

The presence and abundance of a given set of functional traits, i.e. morphological, physiological and life history characteristics, is strongly related to particular biodiversity levels and to the functioning of grasslands and shrublands. Overall, plant traits can be used to indicate vegetation response to environmental change and disturbance, and to indicate the consequences of those changes for ecosystem processes. The traditional tool to measure biodiversity in grasslands and shrublands, however, is the direct use of diversity measures.

Agro-ecosystems are ‘functional units, producing agricultural products and providing rural services’. Their spatial extent ranges from a single field to the global scale. Agro-ecosystems are clearly distinct from other ecosystems in that the agricultural production is an integral part of the ecosystem maintenance. Consequently, sustainability of agro-ecosystems needs to be ecologically as well as economically viable. Increasing effort in the development of indicators and assessment methods for this ecosystem is evident as of the beginning of the 1990s, when many national agri-environmental programs at the farm and even field level, as well as at the European level, started.

Indirect indicators for the landscape level can be split up into Land Use Intensity (LUI) and Landscape Structural (LS) measures. The majority of the indicators are single measures addressing species populations, landscape features or land use intensity. Moreover, animal or plant traits prove to be closely linked to human activities and environmental factors and, therefore, offer a good perspective for the development of future indicators. In particular, the development of threshold values for indicators is a significant future challenge at the landscape level.

Some farm and field-scale assessment schemes directly link to farm management and usually contain biotic performance indicators in the shape of indicator species or groups of species. Vascular plants, birds, arthropods in general and carabid beetles in particularly are the most suitable and commonly used indicator groups for biodiversity evaluation. Service indicators are generally available, but there is a considerable lack of quantification of indicator threshold values for service provision maintenance.

Soils indicators are dominated by biotic measures: 70 % of the SCIE references referred to this group as opposed to abiotic indicators (25 %). Soil indication mainly aims at habitat quality assessment and monitoring, which is, due to the limited mobility of many soil organisms, restricted to small spatial scales. This prevalence of indicators at the local scale is closely linked to the soil complex patterns of

variability within small spatial units. Soil indicators are mainly related to supporting services, such as soil formation and nutrient cycling, which comprise almost half of the indicators found in SCIE references. The use of traits is common for life-history and functional characteristics of soil microorganisms.

Historical wetland alteration and destruction, mainly caused by land conversion to agricultural use, led to a loss of nearly 50 % of the extent of this ecosystem around 1900. Thus, wetlands belong to the most threatened ecosystems of the world, whereas the alarming conditions are also due to the fact that the severe threats to wetlands have been acknowledged only very recently by non-scientists and decision makers. However, the functional role of wetlands as invaluable sinks as well as sources of nutrients and retention areas for water is widely known. Basically, it is the retention (and removal) capacity that controls nutrient loading also of adjacent ecosystems, such as rivers and lakes.

Physical indicators discriminate among moisture-driven land use/land cover variations, wetland types and the extent of moisture stress. Linking abiotic and biotic indicators, process-based and biochemical or gene-based assays are becoming more and more important. Species traits in wetlands are used to set up management objectives and assess their success. More than 80 % of wetland indicators are applied at the local to regional scale, which may be due to the comparatively small-scale heterogeneity and diversity in this ecosystem.

The greater sub- and global scales of indication corresponding to landscape indicators are of significance in evaluating large scale management strategies. Recent studies on wetlands revealed the lack of appropriate indicators of wetland biodiversity. This lack of common indicators has also a limiting effect on further comparative studies at sub-global and global scales, which in turn negatively affects conservation efforts and progress. On the other hand, wetland indicators are strongly related to important ecosystem services such as the (regulative) self-purification and water retention.

The application of biotic indices for monitoring rivers dates back more than 100 years, when the first saprobic indices were developed in Germany. Frequently revised and applied in many European countries, this group of indices addresses water pollution through organic matter. Another large group of indicators address river trophy, i.e. the eutrophication with nutrients originating mainly from floodplain land use practices. Both saprobic and trophic indices are based on species tolerance values. Since the early 1980s, the utility of numerous functional metrics, traits and indices became important. Each of the measures potentially represents a different aspect of the community or a different response to environmental stress and, thus, potentially provides both a stable measure and a measure accounting for the impact of multiple stressors. The so-called multimetric indices form the basis of many current assessment and monitoring systems in the USA and Europe.

Similar approaches are in use for abiotic indicators, which combine measures of land use intensity, habitat quality and diversity as well as hydro-morphological and chemical parameters. Both abiotic and biotic indices are also subject to predictive modelling. For instance, a common approach uses environmental predictor variables to model community composition under given and natural environmental conditions. The observed community is then compared to the modelled expected reference; the deviation is equivalent to the degree of impairment. In multimetric indices, species traits already constitute part of the component metrics and provide insight into the functional composition of riverine communities and functional aspects of biodiversity.

Since river assessment is usually based on field sampling at the local scale (water body-related), many indicators also refer to the same scale. Theoretically, however, river bioassessment offers broader application at the regional or even sub-global scale in the near future, since the Water Framework Directive sets the outline for river assessment at a pan-European scale. Biodiversity indicators mainly refer to species richness, but more recently also to functional diversity.

Theoretically, landscapes exist at several spatial and temporal scales, depending on the process or organism being studied. Relationships between landscape patterns and species traits, populations and entire biotic communities have been described at a range of scales, ranging from beetles in very small landscapes to birds, mammals and reptiles at regional and continental scales. With the advent of new imagery from aircraft and earth observing satellites (e.g., Landsat) and advances in computing, landscape ecology and associated landscape indicators have exploded since the early 1990s.

Landscape metrics include a wide variety of measures of spatial composition and pattern of habitats, land-cover and land-use, ecosystems and other land-surface features over a given area of interest. These include measures of composition, shape complexity, patch size, connectivity, distance measures and attribute diversity and/or complexity. A landscape metric becomes an indicator when qualitative and quantitative relationships are established, although the terms “landscape metrics,” landscape indicators,”, and “landscape indices” have often been used interchangeably.

Landscape metrics, indicators, and models have been used to evaluate individual species distributions, as well as patterns of species richness and diversity. New remote sensing data and analytical approaches offer significant potential to improve landscape assessments of habitat and species distributions. These include application of LIDAR to map vertical structure and composition of habitat. However, landscape metrics are often applied without consideration for scale-dependency of the processes and patterns they are attempting to capture. Landscape analyses offer the potential to link ecosystem services through space and time. As such, it may be possible to conduct a full cost-accounting of how optimisation of one ecosystem service (e.g., flood abatement through dam construction), might affect other services. From a species and community traits standpoint, these linkages should enhance valuation of these traits at a landscape scale.

1. Introduction

"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 loss of biodiversity to the growing demand for ecosystem services. However, biodiversity loss in turn leads to "the degradation of many ecosystem services [and] could grow significantly worse during the first half of this century […]". The Millennium Ecosystem Assessment raised the need for indicators, particularly such that are linked to biodiversity and its role to sustain ecosystem services. 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, physical modification of rivers or water withdrawal from rivers), climate change, invasive alien species, overexploitation, and pollution. As many services, for instance, food and fuel provision or nutrient cycling, are related to ecosystem functions and processes, appropriate indicators need to account for both functions and processes, too. This may be one of the future challenges for indicator development, because current indicators often focus on rare and threatened species and habitats.

Facing the dramatic loss of both biodiversity and related ecosystem services, huge and concerted effort is required to halt the loss of biodiversity. A fundamental requirement for monitoring the effectiveness of such actions is indicators. Regardless on how the term 'biodiversity' is defined, it is a complex concept, involving either species numbers, species numbers and abundances, functional traits and their abundances, or complex ecosystem processes, none of which can be routinely monitored. For practical purposes in planning, conservation and management, the concept of biodiversity needs to be simplified and made operational through the use of indicators: simple parameters, which nevertheless reflect the complexity of an ecosystem.

Indicators are widely used in applied ecology, since for many purposes complex ecosystem characteristics need to be described by simple values. Most commonly, occurrence or abundance of species or functional groups is used as a proxy for the intensity of a stressor, thus enabling a simple assessment. But also functions, processes and related services are often difficult to measure and indicators are useful. Most complex, however, is the indication of biodiversity.

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). The authors clearly state 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/COP/7, 2003). Both CBD and SEBI 2010 have largely created global awareness for the need of novel biodiversity indicators that can be easily communicated to decision makers. And both initiatives have already suggested a set of global to regional indicators of biodiversity, mainly measures to assess the status of selected rare or threatened species and habitats. Altogether, eight CBD 'focal areas' and 16 EU Headline Indicators have been selected so far, the latter of which contain as many as 26 single indicators.

However, there are a couple of drawbacks in their approaches for indicator selection. First, they do not fully cover the major components of biodiversity, i.e. genetic, structural and functional biodiversity (Noss, 1990, Pioani et al., 2000). Being mainly limited to selected species, a fundamental portion of biodiversity is likely to remain undetected: functions, processes and services, which are important attributes of biodiversity, are not or only indirectly regarded. Second, indicators need to be applied at suitable spatial scales (e. g. Araujo, 2004, EEA, 2007) and are often limited to narrow geographical ranges. For example, a threatened species in the Mediterranean region may be naturally absent in Scandinavia, thus, limiting its suitability to a specific region. Upscaling of biodiversity indicators, such as selected rare or threatened species, needs, therefore, to be related to the species pool at the respective spatial scale. Third, there is much evidence that ecosystem services are dependent on functions and processes rather than single species (e.g., Gren, 1995; Bolund & Hunhammar, 1999; Strange et al., 1999; MA, 2005a; Diaz et al, 2006). Providing fodder for cattle or sheep is related to the community of palatable and nutritious grassland, which might be composed of very different species; the regulation of self-purification in rivers is controlled by a multitude of organisms processing carbon components, which vary in species composition with region and river type. Fourth, indicator suitability may differ between ecosystems: habitat area measures are rarely applied (and maybe less useful) for river ecosystems, while the protected area is likely not suited for application in agro-ecosystems. Fifth, there is a fundamental difference between semi-natural (= managed) and natural (unmanaged) ecosystems. While an appropriate management is crucial to sustain ecosystem services in the former, the latter usually provides the services without management, if still in natural condition. Biodiversity and service indicators in semi-natural ecosystems, such as grass- and shrublands, agro-systems or managed wetlands, need to take the management practices into account.

Indicators and indication approaches differ, therefore, between ecosystem types, but have rarely been compared. A comparison may enable cross-fertilisation, and may lead to common principles, which can be applied across ecosystems. Based on an extensive literature review of more than 600 peer-reviewed papers published between 1997 and 2007, we summarise the state-of-the-art of indication in six ecosystems (forests, grass- and shrublands, wetlands, rivers, agro-systems and soils) and at the generic landscape-level. Special emphasis is placed on ecological concepts behind indication, the

approaches to address biodiversity and ecosystem services, and on the five drawbacks listed above. Individually, we aim at answering the following questions:

- Which types of indicators have been developed for the individual ecosystems?
- Do indicators and indication systems refer to biodiversity and ecosystem services?
- Which purpose stimulated the development and application of the indicators?
- At what spatial scale do the indicators work in individual ecosystems?
- Is there a policy driving indicator development and application?
- Have indicators and indication systems been standardised and validated?

1.1 Terms and definitions

Indicator

Although ecological assessment using indicators has a long tradition in aquatic ecosystems (e.g., Kolkwitz & Marsson, 1902; 1908; Pantle & Buck, 1955, Friedrich, 1990) and terrestrial ecosystems (e.g., Holloway, 1980; Dufrêne & Legendre, 1997; McGeoch, 1998), there is no common definition of the term 'indicator' that is i) applicable over a wide range of ecosystems, ii) useful for both abiotic and biotic indicators and iii) widely accepted and applied by the scientific community. McGeoch (1998) provided a comprehensive review on terrestrial insects as biological indicators and suggested a classification into environmental, ecological and biodiversity indicators. For the latter, the SEBI 2010 (Streamlining Ecosystem Biodiversity Indication by 2010) coordination team and expert groups recently provided a standard terminological framework (EEA, 2007). Accordingly, a (biodiversity) indicator serves four basic functions: 1) simplification as it summarises often complex and disparate data, 2) quantification as statistically sound and comparable measures are related to a reference or baseline value, 3) standardisation as they are based on comparable scientific observations and 4) communication as they provide a clear message that can be communicated. We basically follow the criteria of the EEA Technical Report, but extended the definition to environmental and ecological indication. The following definition is applied: *An indicator is a simple, measurable and quantifiable characteristic responding in a known and easily understandable way to a changing i) environmental condition, ii) ecological process or function, or iii) element of biodiversity*.

Moreover, the terms 'biodiversity', 'ecosystem function' and 'ecosystem service' are applied here as follows:

Biodiversity

Biodiversity is the variety of life forms (Wallace, 2007), i.e. the variety of taxonomic, functional and genetic characteristics of life.

Ecosystem function

The interactions (events, reactions or operations) among biotic and abiotic elements of ecosystems that lead to a definite result (Tirri et al., 1998; Wallace, 2007). In broad terms, these processes involve the transfer of energy and materials (Lyons et al., 2005; Wallace, 2007). Here, we follow Wallace (2007) and define ecosystem functions synonymous with ecosystem processes.

Ecosystem services

Ecosystem services are benefits that humans obtain from ecosystems that support, directly or indirectly, their survival and quality of life. These include provisioning, regulating and cultural services that directly affect people, and the supporting services needed to maintain the direct services. They are a subset of ecosystem processes, which include roles that are not easily definable in terms of human needs (enlarged from MA, 2005a).

2. Review procedure and database

During a workshop on indicators and indication approaches in terrestrial and aquatic ecosystems held in February 2007 in Essen, Germany, a set of 72 indicator characteristics were defined and categorised into nine groups to compare indicators and indication across ecosystems (see Annex 1):

- *Ecosystem relevance*: Characteristics describing the field of application of an indicator. It can be limited to a single ecosystem (e.g., Secchi depth in lakes), but might also be applied to all ecosystems, such as the number of rare species of a certain taxonomic group.

- *Numerical type of measurement*: Characteristics describing the scaling of a measure, i.e. whether it is a plain number, proportion, index, etc.

- *Purpose of indication*: Characteristics describing the field of application of an indicator, for example, assessment, monitoring, or prediction. The indicator type was further divided into abiotic, biotic and landscape indicators. Physical and chemical measures are typical abiotic indicators, while biotic indicators usually encompass species number and abundance, guilds and traits, biodiversity measures and bio-indices.

- *Driving force behind indication*: Characteristics describing the motivation to develop an indicator. Apart from specific policy demands, indicator development is often driven by science.

- *Spatial scale*: Characteristics describing the geographical extent of application of an indicator.

- *Relation to ecosystem services*: Characteristics describing if individual ecosystem services as defined by the Millennium Ecosystem Assessment (MA, 2005a) are targeted by an indicator.

- *Standardisation*: Characteristic describing if an indicator has been subject to official national and/or international harmonisation and approval.

- *Validation*: Characteristics describing if a test of performance has been conducted, which is usually done with external data not used for indicator development.

The database was filled with the results of an extensive literature research. A standard set of keywords and keyword combinations was applied to the Science Citation Index Expanded (SCIE; time period from 1997 to May 2007) to gain comparable and quantifiable results for all ecosystems (Table 1).

A total number of 617 references on 534 indicators were reported back by the SCIE, the allocation to ecosystems of which is illustrated in Figure 1.

Table 1: Keywords and combination of keywords used for the literature review in the *Science Citation Index Expanded*. Numbers indicate the hits per keyword combination and ecosystem.

Keyword 1	plus	Keyword 2	plus	Keyword 3	plus	Keyword 4	Forests	Grass-/Shrublands	Agro-systems	Soils	Wetlands	Rivers	
'Ecosystem'	+	biodiversity					3,646	1,314	53	n.a.	352	1,180	
'Ecosystem'	+	assess*					9,038	1,938	109	n.a.	1,199	8,811	
'Ecosystem'	+	service*					1,519	134	16	n.a.	127	618	
'Ecosystem'	+	monitoring					2,688	469	18	n.a.	478	3,527	
'Ecosystem'	+	environm*					12,114	2,990	141	n.a.	1,773	14,119	
'Ecosystem'	+	indicat*					15,723	3,553	126	n.a.	2,144	15,532	
'Ecosystem'	+	indicat*	+	review			312	51	4	n.a.	48	223	
'Ecosystem'	+	indicat*	+	biodiversity			932	339	20	n.a.	95	297	
'Ecosystem'	+	indicat*	+	assess*			2,529	577	44	n.a.	433	2,664	
'Ecosystem'	+	indicat*	+	service*			179	13	5	n.a.	16	117	
'Ecosystem'	+	indicat*	+	monitoring			828	158	4	n.a.	168	1,388	
'Ecosystem'	+	indicat*	+	environm*			3215	883	41	n.a.	571	4,130	
'Ecosystem'	+	indicat*	+	environm*	+	assess*	613	166	20	n.a.	128	933	
'Ecosystem'	+	indicat*	+	biodiversity	+	service*	12	0	2	n.a.	2	3	
'Ecosystem'	+	indicat*	+	biodiversity	+	assess*	258	75	7	n.a.	26	80	

* truncation

Ecosystem reviews and the final comparison of results followed a two-tier process. In a first step, the database was used to illustrate indicator development and indicator application during the past ten years. Thus, the database provided a means for simple descriptive analyses focussing on the questions outlined in the introduction. The analysis was limited to the comparison between ecosystems; relative proportions of indicators of a specific characteristic were calculated to account for the difference in total number of indicators among ecosystems. In a second step, ecosystem reviews were supplemented by references before 1997 and non-peer reviewed journals, books and reports.

The ecosystem co-authors referred to a common structure to provide comparable insight into the history of indication, indicator types (e.g., numerical types, abiotic, biotic and landscape indicators), spatial scales of indication, management (if applicable), and the relation to biodiversity and ecosystem services.

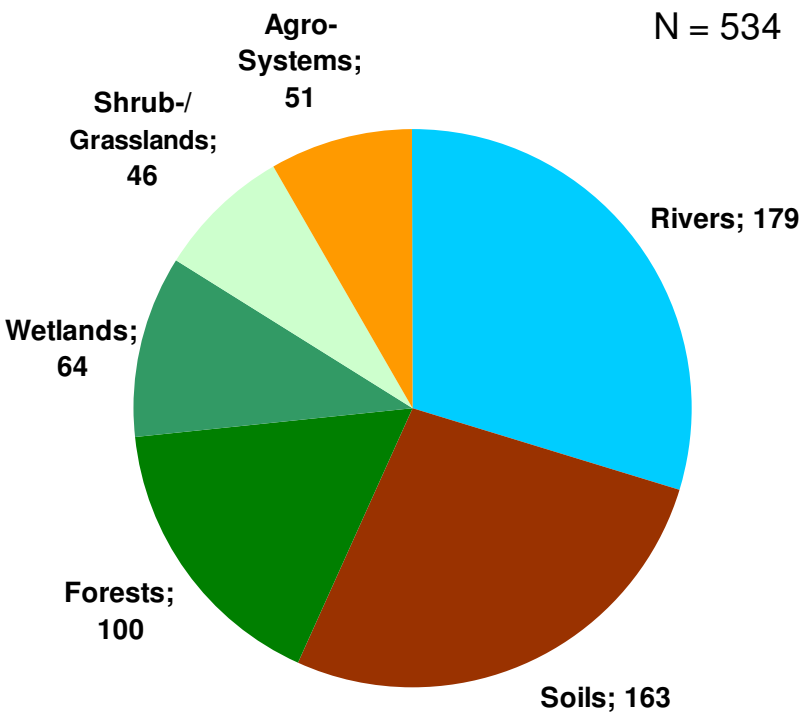

Figure 1: Allocation of 617 references of 534 indicators referring to six ecosystems (except landscapes) reviewed in this study.

3. Ecosystem reviews

3.1 Forest ecosystems

History

One of the first publications addressing the indicator concept dates back to 1919 (Hall & Grinnell, 1919). The authors associated plant and animal species to different habitat types, called life-zones. Contrary to this early ecological classification by Hall & Grinell, many early forest indicator systems were designed to indicate forest stand condition and productivity (e.g., Jonson, 1914). The earliest forest inventories used direct measures on individual trees for extrapolation of growth and stock at the stand level. In Sweden, the first discussion about forest inventories dates back to 1735 (von Segebaden, 1998), but was not realised until the early twentieth century and included coarse descriptions of the understorey plant community. Based on the early inventories, forest indication was developed and refined during the following century (e.g., Hägglund & Lundmark, 1981). At present, forest indicators are available for an array of purposes, including measures of forest productivity, indicators for conservation purposes and the evaluation of ecosystem services.

The natural forest ecosystem comprises a large variety of forest types, determined by geographical and climatic factors. But a major part of forests today encompass a range of semi-natural to entirely artificial tree plantations. Irrespective of the degree of naturalness of a forest stand, forest ecosystems provide a set of unique characteristics that are key factors for biological diversity, which have been conceptualised and described by Noss (1990).

Indicator types

The theoretical framework of Noss (1990) divides forest compositional key factors for diversity into structural, compositional and functional components. In a comprehensive review, Larsson (2001) expands the framework by Noss and gives explicit examples of forest indicators for each of the components, at different spatial scales. Indicators for compositional key factors on a national level should include native, non-native and ‘not-site-original’ species, with the addition of species with specific landscape scale requirements at the landscape scale. Suggested indicators for structural key factors vary with spatial scale. At the national level the indicators include, for example, the total area of: forest types, utilised and protected forests, old growth forests, forests left for free development, afforested/deforested stands and productive forest with respects to tree species and age. At the landscape level, Larsson (2001) suggested habitat composition, lakes and rivers, spatial continuity and connectivity of important habitats, and fragmentation and history of landscape use. At the stand-scale, tree species, stand size, edge/shape and history, habitat type tree stand structural complexity, dead wood and litter are suggested as indicator types for structural key factors. Functional indicators are not scale-dependent, yet are divided into natural and anthropogenic. Natural indicator types comprise fire,

wind and snow, and biological disturbance, while anthropogenic influence is indicated by forestry, agriculture and grazing, other land-use, and pollution.

It has been suggested that two to three compositional and structural key indicators should be selected for effective monitoring (Ferris & Humphrey, 1999). The compositional indicators should be functionally linked to a broad range of other species, such as the extent and species composition of the broadleaved component in conifer forests, and the structural indicators should act as surrogates for general species richness or diversity, for example, the quantity and quality of dead wood.

Indicator types: Single and complex indicators

Forest biodiversity has been traditionally indicated by single indices, as opposed to the multi-metric indices widely applied in river ecosystems (Karr, 1991; Dahl & Johnson, 2004, see also Chapter 3.6 Rivers). There are few examples of weighted averages available over a set of abiotic indicators (e.g., the Ellenberg indices), but no true multi-metric index. In a critical review, Failing & Gregory (2003) commented on the lack of complex indices in forest monitoring, and argue that carefully-designed multi-metric indices might lead to better decisions regarding forest management, compared to single-metrics.

Indicator types: Biotic and abiotic indicators

Biotic indicators of forest biodiversity comprise both compositional and structural components. General descriptions and suggestions have been given by Lambeck (1997), Noss (1999) and Lindenmayer et al. (2000). In a review, Carnican & Villard (2002) suggested the following indicators for forest biodiversity:

- Keystone species: species that generate large ecosystem effects by their interaction with other species (e.g., some woodpeckers).

- Umbrella species: species that require large areas of suitable habitat to maintain viable populations and whose requirements for persistence are believed to encapsulate those of an array of associated species. These species usually have very large home ranges (e.g., bears, wolves).

- Dispersal-limited species: species that are limited in their ability to move from patch to patch or that face a high mortality risk in trying to do so (e.g., wingless insects, species restricted to humid microhabitats, such as most amphibians).

- Resource-limited species: species requiring specific resources that may be critically short, either temporally or spatially. These resources may include snags, nectar sources, fruits, etc.

(e.g., woodpeckers restricted to large-diameter snags for nesting and foraging; oligophagous insects).

- Process-limited species: species sensitive to the level, rate, spatial characteristics or timing of some ecological processes such as fire, flood, grazing, competition with exotic species, or predation (e.g., flora and insect community in grazed forests).

- Flagship species: species that can easily attract public support for conservation (e.g., Giant Panda, *Ailuropoda melanoleuca*).

Abiotic forest biodiversity indicators include many different components, from direct measurements of physical structures, to indirect measures. Indicators that can be measured directly or by using GIS and remote sensing include soil quality (Ponge & Chevalier, 2006), mosaic of landscape types or hydrological regime (Hagan & Whitman, 2006). Indirect indicators may be factors such as frequency and intensity of fires and storms, or related economic and socio-cultural assets (Kotwal et al., 2008).

Another kind of indicator combines abiotic and biotic factors into so-called indicator values that were developed to predict abiotic conditions from the extant plant community. A widely-applied example is the 'Ellenberg system' of indicator values ('*Zeigerwerte*'; Ellenberg et al., 1992), refined and localised by, for example, Diekmann (1995), Wamelink et al. (1998), Hill et al. (2000) and Schaffers & Sýkora (2000). The Ellenberg values refer to plant species' ecological and physiological optima. Common applications are to assess changes in the environment using data from ancient plant inventories, or to model abiotic conditions required to maintain a specific plant community.

Indicator types: Traits

Although the traditional way of looking at forest biodiversity has been from a taxonomical point of view, the use of species traits (McIntyre et al., 1999) instead of phylogenetics has gained popularity over the last decade. The ability for a plant to survive a disturbance depends to a large extent on vegetative or seed-related attributes of the species. As forest management is a form of severe disturbance, there have been some suggestions to use species traits as indicators of forest biodiversity and continuity (e.g., Graae & Sunde, 2000; Myking, 2002). Another advantage of using attributes rather than taxonomy is that it allows comparisons across countries and ecoregions. However, there seem to be very few forest indicators based on traits. For instance, the meta analysis used for this study returned only one out of 270 studies that referred to traits.

Spatial scales

The intensity and frequency of disturbances is one of the major natural factors determining species composition and diversity in the forest landscape (Spies & Turner, 1999). At present, however, the most important factor is forestry and forest management. A disturbance may be anything, from the

effect of trampling at the centimetre scale, to regional storm damages of some hundreds or even thousands of square kilometres. Few, if any, forest indicators cover all spatial scales (but see Gray & Azuma, 2005). Commonly, forest indicators are divided into national or ecosystem level, regional, and local scale (Larsson, 2001). Local scale biodiversity indicators generally describe the alpha and beta diversity, or forest productivity parameters. At the stand level, changes in tree diameter and age class, soil nutrient levels, and stand growth can indicate forest degradation (Lundquist & Beatty, 1999). Biodiversity at stand level is often determined by so-called matrix effects, which are given by the composition of the surrounding landscape (Sverdrup-Thygeson & Lindenmayer, 2003). At the landscape level variables like changes in land cover, habitat fragmentation and management practices are important for local-scale biodiversity and may, thus, be utilised as indicators. Indicators at the national or ecosystem level include interval and frequency of fires, nutrient cycling, and hydrological regime (Harwell et al., 1999). However, the relationship between stand-level diversity and factors at landscape or national levels is complex and difficult to assess (Waldhardt, 2003).

Relation to management

During the 1990's, a number of action plans for sustainable forest management were issued, for example the Swedish Action Plan for biological diversity and sustainable forestry (Wallin et al., 1995), The U.K. Forest Standard (Forestry Commission, 1998), and Criteria and indicators for sustainable forest management in Finland (Eeronheimo et al., 1997). The common theme for these action plans is the aim to combine sustainable forestry with high ecological assets. Along with such aims comes a need for methodologies for rapid and effective biodiversity assessment. However, the relationship between biological diversity and many suggested indicators for biodiversity have not been fully understood (Stephens & Wagner, 2007). It has been shown to be difficult to determine how many of the suggested indicators should be measured, interpreted or monitored, partly as ecological knowledge lags policy initiatives (Ferris & Humphrey, 1999; Lindenmayer et al., 2000; Hagan & Whitman, 2006). Some of the indicator systems suggested during the last decade entirely rely on taxon-based indicators (e.g., Nitare, 2000). Others follow a more holistic approach, i.e. they aim to include all aspects of sustainability in the indicator framework: ecological, economic and socio-cultural assets (e.g., Kotwal et al., 2008). The definition of desired endpoints has been one approach to quantify the deviation from a desired state (Frego, 2007). The Swedish system of environmental quality criteria for the forested landscape (Anonymous, 1999) is one example of a system that uses a set of endpoints or indicators that represent forest integrity, and that are easy to measure.

Indicators of biodiversity and ecosystem services

Indicators of forest biodiversity are also partly described above (biotic *vs.* abiotic indicators). Forest ecosystem services can be divided into provisional, regulating, supporting and cultural. The provisional services include: timber, fuel-wood, drinking and irrigation water, fodder, food, medicine, and genetic resources. Regulating and supporting services include a large array of ecosystem functions

such as mitigation of air pollution, oxygen emission, nutrient cycling, water regulation, biodiversity, carbon sequestration, mediating weather extremes and soil formation. Finally, there are multiple cultural (societal) services, such as employment, aesthetic values, and recreation (Pattanayak & Butry, 2003). For some of these services, indicators are already available: e.g., timber production (either wood volume or monetary value), number of visitors to protected forests or water quality indices. There have also been attempts to quantify the relationship between ecological function and species diversity (e.g., Loreau et al., 2002). Balvanera et al. (2005) reported an example of how species composition and richness is related to carbon cycling, productivity and pollination.

3.2 Semi-natural grassland and shrubland ecosystems

Grasslands and shrublands are semi-natural ecosystems traditionally maintained by low to intermediate management or disturbance events. In Europe, and similarly in the whole Mediterranean basin, grasslands and shrublands have been coevolving for millennia with human societies by providing mainly fodder for livestock production, turfs and fuel for burning (Perevolotsky & Seligmann, 1998; Wessel et al., 2004; Quétier et al., 2007; Partel et al., 2007). Semi-natural grasslands are typically dominated by a diverse herbaceous plant community that is maintained by livestock grazing or mowing. Shrublands are typically characterised by woody species managed, for example, for fuel production. Traditional management aims at reducing shrub dominance (e.g., by prescribed burning) in order to regenerate herbs for fodder (Perevolostky & Seligman, 1998; Papanastasis, 2004). In humid regions, shrublands may convert to heathlands dominated by, for example, *Calluna vulgaris.*

Indicator types

The use of indicators for grassland and shrubland biodiversity can be based on a considerable array of different possible measures. Altogether, more than 100 papers were reviewed resulting in, at least, 38 different indicators, half of which describe the development or application of single indices (or lists of single indicators). Although many indicators are not ecosystem-specific, a number of indicators have been specifically designed for grasslands and shrublands, such as % bare soil, grazing intensity, and vegetation patchiness (Milchunas et al., 1988; Alados et al., 2004; Pueyo et al., 2006). At present, however, common indicator systems in Europe do not make use of specific indicators for grasslands and shrublands. Yet, ecosystem-specific indication has an extensive tradition in North America and Australia (Hof et al., 1999; Pyke et al., 2002; Briske et al., 2005); the indication systems are mostly based on the concept of pasture 'health' or 'condition' for management practices (Pyke et al., 2002; Briske et al., 2005), i.e. focusing on the maintenance of sustained production. These systems are commonly based on the selection of a synthetic list of single indices that, by their evaluation, give an estimation of the overall habitat condition. Indicators commonly used in these systems are biotic

indicators (see below). The validation of indicators for grasslands and shrublands has been traditionally limited, especially in Europe.

Indicator types: Biotic and abiotic indicators

Non-biotic indication in grasslands and shrublands cover environmental and anthropogenic forces that control and determine local biodiversity patterns and habitat conditions: i) the management/ disturbance regime, ii) environmental conditions and iii) landscape patterns. Grazing, mowing and fuel management are the most relevant indicators of management and disturbance in grassland and shrubland habitats (Milchunas et al., 1988; Pons et al., 2003; Delitti et al., 2005; Diaz et al., 2007), reflecting different processes that cause biomass loss and regeneration. But management indicators are also subject to substantial natural variation, for example through wild grazing and fires, and to human impact, such as fertilisation, fragmentation, pollution, and drainage. Environmental conditions are determined by soil parameters, such as moisture, pH, nutrients, Soil Organic Matter (SOM) and other physical and climatic parameters, for instance light at the ground layer, rainfall and microclimate (Partel et al., 2004; Sebastaia, 2004; Pueyo, et al., 2006; Briton & Fischer, 2007; EASAC, 2005; DEFRA, 2007). Climatic trends and nutrient deposition are routinely monitored Europe-wide, as part of standard environmental programs (European Monitoring and Evaluation Programme, EMEP) or estimated using biotic indicators (EASAC, 2005), for example, the spring index (UKPN, 2007) and Ellenberg values (Ellenberg et al., 1992). Landscape patterns are described with parameters such as habitat area, habitat connectivity/fragmentation and heterogeneity/diversity of the landscape (Broton et al., 2004; Helm et al., 2006; Petit & Firbank, 2006).

Indicator types: Traits

The presence and abundance of a given set of functional traits (e.g., morphological, physiological and life history characteristics; Diaz et al., 2007) is strongly related to particular biodiversity levels and to the functioning of grasslands and shrublands (Quetier et al., 2007). For example, Hodgson et al. (2005) demonstrated that leaf characteristics and plant height could be used as indicators for biodiversity conservation and productivity in different European grassland systems. Plant traits linked to leaf chemistry can be very effective to indicate biogeochemistry patterns (Eviner & Chapin, 2003) and animal nutrition (Hoste et al., 2006), while flower type and density can be very valuable in assessing the diversity of pollinators and their services (Hegland & Boeke, 2006; Potts et al., 2006). Growth forms (e.g., shrubs, grasses, herbs) and life cycle (short *vs.* long living species) are very effective indicators for monitoring biodiversity, management and fire regimes, and habitat condition (Bardgett et al., 1995; Whitford et al., 1998; Schwab et al., 2002; Pons et al., 2003; Tasser et al., 2003; Pueyo et al., 2006; Diaz et al., 2007), especially because they can, in many cases, be monitored by modern remote sensing tools.

The canopy structure of the vegetation provides a useful set of indicators as it is characterised by vertical and horizontal components, for instance, vegetation height, density per strata, Leaf Area Index (LAI), vegetation cover, % bare ground and patchiness (Schwab et al., 2002; Hodgson et al., 2005; Paruelo et al., 2005). Different remote sensing tools can also help defining these parameters (Paruelo et al., 2005). Vegetation patchiness relates to important patterns in biodiversity and habitat condition, especially in dry habitats (Whitford et al., 1998; Maestre, 2004; Ludwig et al., 2005; Pueyo et al., 2006, de Bello et al., 2007).

Spatial scales

Biotic indicators of grasslands and shrublands have been traditionally measured at fine spatial and temporal scales, i.e. at the patch or community level. Indicators, such as the number of rare species, can be easily extrapolated to the regional level. Species-area curves, for example, express the linkage of species richness and sampling area and enable an estimation of diversity patterns and vegetation structure over different spatial scales (Magurran, 2004; de Bello et al., 2007). At the broadest scale, landscape patterns are assessed to measure and monitor the trend in grassland area in a landscape or region; Helm et al., 2006) and information is required over long time scales to indicate the actual diversity (even centuries; Partel et al., 2007). Several biotic indicators can be derived from remote sensing tools, if the appropriate resolution for reliable estimations is available (Rocchini et al., 2004; Paruelo et al., 2005). By these tools, we can estimate the biotic component of grassland and shrubland habitats, such as growth form composition, LAI, patchiness, % bare ground, productivity or fire regime.

Relation to management

Overall, plant traits can be used as indicators of vegetation response to environmental changes and disturbance (Kahmen & Poschlod, 2004; Poschlod et al., 2005; Diaz et al., 2007) and as an indication of the consequences of these changes on ecosystem processes (Tasser et al., 2003; Quetier et al., 2007). It is important to notice, however that the predictive value of plant traits changes in different environmental conditions (de Bello et al., 2005). Thus, traits as indicators need to be specified differently in regions with different climate and herbivory history (see Diaz et al. 2007 for a review).

Despite the fact that precise information on these indicators can offer an important indication system on its own, such information is often unknown or, generally, scarce. The trends of these factors need also to be tracked to provide suitable indication systems for grasslands and shrublands. For example, since biodiversity slowly responds to changes in management and landscape structure, it is of importance to consider the history and legacy past (e.g., historical area and connectivity: Helm et al., 2006; historical grazing pressure: Milchunas et al., 1988).

Indicators of biodiversity and ecosystem services

The use of a particular (sensitive or key) species or set of species as an indicator is widespread and common. For instance, a selected 'synthetic list of taxa', umbrella species, red list species or invasive species potentially indicates particular levels of biodiversity and habitat condition (Niemela & Baur, 1998; Duelli & Obrist, 2003; Rosenthal, 2003; Lindborg, et al. 2005; Maes & Van Dyck, 2005; Wittig et al., 2006). The traditional tool to measure biodiversity in grasslands and shrublands, however, is the direct use of diversity measures (Helm et al., 2006; de Bello et al., 2007). The diversity within a reduced set of taxonomic groups is then used as a proxy for the overall diversity (Magurran, 2004; Briton & Fischer, 2007). In general, the value and applied relevance of most diversity indices needs to be adjusted to the extent and composition of the regional species pool (e.g., local species richness divided by the regional species pool; Ingerpuu et al., 2001; Partel et al., 1996).

3.3 Agro-ecosystems

In this review we follow Xu and Mage (2001), who define agro-ecosystems as 'functional units, producing agricultural products and providing rural services'. They can range in scale from a single field to the whole globe and are clearly distinct from other ecosystems by the fact that agricultural production is an integral part. As a consequence agro-ecosystems need to be ecologically as well as economically viable to be sustainable.

History

Agriculture is the activity that by far lays the largest claim on the surface area of the European Union (77 %, see (Piorr, 2003)), and the intensification of agricultural production due to the 'green revolution' after the second world war led to a growing concern over the increasing pressure on natural resources and environment. In the early 1970s, agricultural production environments started to be referred to as agro-ecosystems (first encounter in web of science search), and at about the same time the first agri-environmental schemes to enhance biodiversity in agricultural areas were launched (Robinson, 2005). In the European context, environmental considerations became part of the new Common Agricultural Policy in the early 1990s, followed by a direction towards sustainable agriculture (Piorr, 2003). Also from the beginning of the 1990s, the wish to evaluate the effects of national agri-environmental programs at the farm and even field scale as well as at the European scale, led to an increasing effort to develop indicators and assessment methods. During the last decade, the increasing interest in sustainability and resilience of ecosystem functions and services has led to the development of more complex indicators.

Indicator types

Agri-environmental indicators have been classified according to various conceptual frameworks for biodiversity (see Xu & Mage, 2001; Duelli & Obrist, 2003; Clergue et al., 2005; Buchs, 2003a for overviews). Besides Noss' classification of diversity (Noss, 1990), another key publication by Duelli & Obrist (2003) approached indicator selection from the angle of three different motivations: 'conservation', 'pest control' and 'ecological resilience'. Also, in the special journal issue of *Agriculture, Ecosystems & Environment* (Buchs 2003b) a multitude of possibilities, requirements and applications for biotic indicators have been described.

Indirect indicators at the landscape scale can be Land Use Intensity (LUI) based or Landscape Structural (LS) based. LUI indicators often have the disadvantage that there are no reliable measurements available at this spatial scale, especially since the LUI of at least the previous decade influences present day environmental quality and therefore has to be known. The same is true for landscape structure, but indicators of past LS are usually easier to obtain from topographic maps and aerial photographs and obtaining information on present day LS is no problem (Bailey et al., 2007). The landscape scale is also the typical scale for reconstruction, recovery or development plans and indicators for future LUI and LS can easily be derived from plans and maps. This field has even led to the development of specific, dedicated indicators (Bar & Loffler, 2007).

Indicator types: Single and complex indicators

The majority of the indicators found (61 % in the database) are single measures for populations (e.g., number of species, abundance), landscape (e.g., field size) or land use intensity (amount of fertiliser, pesticides applied), but simple indices referring to community structure (e.g., % pioneer species), landscape structure (e.g., share of semi-natural elements) or LUI (e.g., share of highly fertilised land) are also quite common (Buchs, 2003a; Herzog et al., 2006). The Shannon-Weaver or Simpson indices are used as indicators for species communities as well as LS and LUI (Bailey et al., 2007, Buchs, 2003a). Practically, single indicators are very seldom used for their own merit. The majority of applications are within assessment systems like the ones reviewed by Braband et al. (2003) and Galan et al. (2007). The increasing interest in biodiversity and ecosystem functions and services (Clergue et al., 2005) is leading to the development of more complex community indicators (e.g., Balvanera et al., 2005) while the increasing interest in easy large-scale assessments using remote sensing imagery is leading to the testing and development of more complex landscape measures (Ares et al., 2001; EEA, 2005).

Indicator types: Traits

Animal or plant traits are closely linked to human activities and environmental factors (Pervanchon, 2004; Clergue et al., 2005) and, therefore, offer a good perspective for the development of indicators. For instance, Balvanera et al. (2005) showed the effect of management on functional species and

Hendrickx et al. (2007) showed that arthropod communities not only decreased in number due to increasing land use intensity and decreasing landscape structure, but also homogenised across the landscape, probably ending up with a limited number of generalists with good dispersal capacities being present.

Spatial scale

Here we take practical use at the relevant assessment levels within the EU as the basic motivation for an indicator overview, and, therefore, mainly discuss the utility at three different spatial scales: field/farm, regional/landscape and above regional (national to European) scale. These scales are distinct in the organisational as well as the spatial sense.

Assessment schemes at the field/farm scale intend to assess environmental impacts of farming practices and monitor ecological achievements. The schemes mostly apply a mix of land use intensity indicators (e.g., pesticide and fertiliser use, live stock density, crop rotation schemes), landscape structure indicators (e.g., area or length of field margins) and biotic indicators (e.g., weed, vascular plant and bird species numbers). Braband et al. (2003) evaluated seven assessment tools used in five different European countries and concluded that, while the assessment systems sufficiently reflected the state of abiotic resources, for biotic resources "…the ecological quality, the state of the cultivated farmland, is not really reflected". Species oriented indicator systems are deemed best suited to this purpose, but complexity and regional specificity are cited as drawbacks. Galan et al. (2007) compared the results of five different French assessment tools applied at the same farms. In all these tools, biotic indicators played only a minor role. Nevertheless, the tools were found sometimes to produce completely different results, which is blamed on the fact that the tools were designed for different regions and farming systems.

Agri-environmental schemes are intended to affect environmental quality and biodiversity at the agricultural landscape scale and are usually assessed at that. Environmental Impact Assessments of agri-environmental schemes have been the subject of a number of review studies (Primdahl et al., 2003; Payraudeau & van der Werf, 2005; van der Werf et al., 2007). However, environmental impact studies only include the effect on biodiversity at a very general level and in none of the reviews are indicators discussed. Also, no studies evaluating or quantifying the effect of agri-environmental schemes on landscape structure (e.g., on the share of semi-natural elements) were found. Remarkably, a review by Kleijn and Sutherland (2003) evaluating the biodiversity effects of agri-environmental schemes, only found 62 evaluation studies originating from just five EU countries and Switzerland, with the majority of the studies (76 %) being carried out in the Netherlands and the UK. The review was limited to the effects measured by direct biotic indicators. The indicators used for that purpose were plants (32 % of studies), insects and spiders (32 %) and birds (47 %), with the exception of a single study that used a mammal (the brown hare).

Indicators at the landscape scale are often used in relation to the sustainable use of agriculture (e.g., Zhen & Routray, 2003). However, objective criteria for sustainability do not yet exist. The development of threshold indicators for sustainability is therefore a significant future challenge.

At the above regional spatial scale, the objective of assessment and monitoring systems is mostly the delivery of environmental impact and biodiversity trend information in a form suitable for comparisons between regions and countries. After a first indicator evaluation by the OECD (2001) several projects have been dedicated to indicator development at this scale (EEA, 2001; EC 2002; DelBaere & Nieto, 2004; EEA, 2005). At this scale, the ease of collecting data for large areas and the comparability, reproducibility and availability are requirements that almost rule out the use of direct biotic indicators. In the EEA's European scale set of agri-environmental indicators (EEA, 2005), only 3 of the 16 indicators used for 'biodiversity and landscape' can be called biotic measurements ('genetic diversity', 'farmland birds' and 'impact on habitats and biodiversity'). The other indicators are indirect LS indicators that can be calculated from satellite imagery and LUI indicators obtained from regional or national statistics. The indicator set is devised to identify trends, threats and policy performance and most of the indicators only have a very tenuous link with biodiversity. The same is true for the two indicators for the sustainable use of agriculture ('nitrogen balance' and 'area under management practises potentially supporting biodiversity') in the recently proposed first set of European headline indicators for biodiversity monitoring (EEA, 2007).

Given the existence of LS measures with good links to biodiversity at the landscape scale (Bailey et al., 2007; Billeter et al., 2008), there certainly is a perspective for the development of agro-ecosystem biodiversity indicators with a much better spatial resolution and sensitivity for the effects of landscape change, especially for the interregional and international comparison applications often used at the EU scale.

Relationship to management

Farm and field-scale assessment schemes such as those reviewed by Braband et al. (2003) and Galan et al. (2007) directly link to farm management and usually include biotic performance indicators in the shape of indicator species or groups. The link between indicators and management is often qualitative in the sense that the objective of indication is only to show if pressures have increased or decreased and if biodiversity has improved or deteriorated. At this spatial scale, some specific management-related indicators (such as vascular plants (Wittig et al., 2006) and insect diversity (Di Giulio et al., 2001)) of impacts of management in grasslands have been described. At the landscape scale, Billeter et al. (2008) provide quantitative information about the relationship between species richness in birds, vascular plants and arthropods and LUI and LS indicators. Agri-environmental schemes frequently use these groups as performance indicators. At the larger spatial scales, the effect of policies is measured through performance indicators, e.g., changes in nitrogen and pesticide use, Farmland Bird Index

(Gregory et al., 2005), changes in landscape structure, as well as policy performance indicators like the change in protected or managed areas (OECD, 2001; EEA, 2005, 2007).

Indicators of biodiversity and ecosystem services

The possible damage that biodiversity loss could pose to ecosystem functioning has led to much research on the subject, especially on the relationship between diversity and ecosystem stability (Loreau et al., 2001; Loreau et al., 2003; see Clergue et al., 2005 for review). Buchs (2003a) reviewed biotic indicators of biodiversity and remarked that biodiversity in this case mostly means species richness. Since biodiversity sampling is basically labour and time-intensive and therefore expensive, the sampling schemes used in monitoring and assessment programs are usually limited to a number of easily sampled species groups and/or surrogate measures under the assumption that these are indicative of general biodiversity/species richness. The representativeness and sensitivity of sampling systems and species groups have therefore regularly been the subjects of research (Duelli et al., 1999; Duelli & Obrist, 2003; Oertli et al., 2005; Bailey et al., 2007; Billeter et al., 2008). Although results very much depend on method and spatial scale of sampling, vascular plants, birds, arthropods in general and carabid beetles in particular are the most suitable and commonly used indicator groups (Buchs, 2003a; Kleijn & Sutherland, 2003).

Agro-ecosystems or, more generally, agricultural landscapes provide a large number of services. Most obvious are provisioning services (e.g., food, fuel), but indicators for those account for production rather than biodiversity, and are, therefore, not discussed here. Sustainability of the provisioning services, however, depends largely on regulating services, the best known and important of which are pollination and pest control. Good indicators for these services are obviously the pollinators and pest controllers themselves, for many of which threats to service delivery and conditions necessary for sustainable service delivery are fairly well known (Allen-Wardell et al., 1998; Stacey, 2003; Klein et al., 2007). Moreover, service delivery is enhanced by agri-environmental schemes (Albrecht et al., 2007). Yet, the literature research in the SCIE did not produce a single reference of a measure quantifying a threshold for service delivery. Hence, in general, the level of knowledge about the relationship between biodiversity, ecosystem functioning and service delivery still seems to be (mainly) qualitative (Jackson et al., 2007). Cultural services delivered by the agro-ecosystem include landscape cultural history, recreation and landscape aesthetics. These services require new types of indicators of, for example, multi-functionality (Rossing et al., 2007) or visual landscape preferences (Dramstad et al., 2006). Indicators for (regulatory) water-related services as well as supporting services related to soil formation are discussed elsewhere.

At the field/farm scale, abiotic and habitat structural indicators are reliable but have the setback that they do not reflect the real effect of measures on local biodiversity. Biotic indicators, especially the ones based on fauna species, are problematic at this scale because they are prone to the influence of

surrounding habitats (meta-population dynamics, larger species like birds) and to the effects of local disasters. Because of the existence of seed banks, indicators based on plant species are to a lesser extent hampered by these problems. However, the results of both Braband et al. (2003) and Galan et al. (2007) show that assessment tools are still devised locally and that even for the abiotic and structural components indicators are still not standardised and calibrated.

Biodiversity effects at the regional/landscape scale can be assessed through direct measurements (species numbers and abundance, species composition: Buchs (2003a)), as well as indirect measurements (changes in landscape structure and intensity of use: Herzog et al. (2006) and Bailey et al. (2007)). Direct biotic measurements are, provided they are used in a correct way to reflect landscape biodiversity levels (Duelli & Obrist, 2003), the best suited for the assessment of biodiversity states and trends, but they also have several drawbacks such as the time lag between the occurrence of environmental change and the reaction of species. Also, biotic measurements are often complex, expensive and labour intensive. Landscape structure measurements have the advantage of being easier and more cost effective, being suitable for making predictions based on planned landscape developments and of easily providing standardised information for interregional or international comparisons. But to be really useful, indirect or surrogate indicators must have a known, quantified relationship with real biodiversity. Not surprisingly, this relationship has received increasing attention over the last decade (Ma et al., 2002; Moser et al., 2002; Dauber et al., 2003; Jeanneret et al., 2003; Zechmeister et al., 2003; Bailey et al., 2007; Billeter et al., 2008).

3.4 Soil ecosystems

Soils are defined as the “upper layer of the earth crust composed of mineral parts, organic substance, water, air and living matter (ISO 11074-1, 1996)”. They provide many natural as well as anthropogenic functions and services such as decomposition of organic matter, nutrient cycles (carbon, nitrogen, sulphur, etc.) or degradation of pollutants (very important for groundwater quality). Moreover soil plays an important role in hydrology, it supports buildings and infrastructure and it is the substratum for agriculture and nature. Soil also contains a huge amount of organisms with a high biodiversity that live in close interaction with the above ground organisms. These soil functions and services are under increasing pressures globally by contamination, erosion, organic matter decline, compaction, salinisation and landslides (EC, 2006).

History

In applied ecology the direct use of organisms as indicators of soil quality has a long tradition (e.g., Volz, 1962) and many ad-hoc working groups have used different soil organisms for this purpose, particularly protozoa (e.g., Foissner, 1999; Bamforth, 2007), nematodes (e.g., Yeates et al., 1999;

Pavao-Zuckerman & Coleman, 2007), enchytraeids (e.g., Graefe & Schmelz, 1999; Beylich et al., 2006), earthworms (e.g., Paoletti, 1999; Brown & James, 2006), mites (e.g., Behan-Pelletier, 1999; Minor & Cianciolo, 2007) and collembolans (e.g., van Straalen, 1998; Sousa et al., 2006; Greenslade, 2007). However, as no single group from the several ad-hoc works is able to cover the huge variety of sites and soils, none of these group-specific concepts became widely or routinely used (Breure et al., 2005). On the other hand, contrasting with the aquatic ecosystems, soil quality indices have been developed quite recently and most still require further comprehensive work to accomplish their validation and standardization before wider implementation in monitoring schemes.

Indicator types

Altogether, 152 soil indicators were recorded based on 356 SCIE references. A distinctive predominance of biotic indicators was found in the literature. 70 % of the references dealt with biotic indicators, while 25 % were about the use of abiotic indicators and soil processes; less than 5 % were about indices comprising abiotic parameters coupled with biotic indicators. Within biotic indicators, more than one third of the references were related to micro-organisms (143 references) and a fifth comprised studies with macrofauna (96 references), including below (e.g., earthworms) and above ground organisms (e.g., carabids). Micro- and mesofauna were relatively less represented in the literature despite the considerable amount of studies focusing on nematodes (48 references) and micro-arthropods (mites and collembolans with 46 and 43 references, respectively).

Two thirds of the indicators reported referred to direct measures such as soil respiration and soil organic matter content (abiotic parameters), microbial biomass measured by phospholipids fatty acids and enzymatic activity (microbial parameters), in addition to abundance, biomass and species composition of organisms from micro, meso and macrofauna. A considerable relative number of indicators found in SCIE references (21 %) were indices calculated to assess soil quality, mostly related to microbial parameters. Main examples are the quotient of microbial carbon in the biomass to organic carbon content (Cmic/Corg) as an indicator for C-dynamics in soil (Emmerling et al., 2001, Anderson 2003, Friedel et al., 2006) and the metabolic quotient (qCO2) as an indicator of energetic efficiency (Kutsch et al., 2001; Chapman et al., 2003; Joergensen & Emmerling, 2006).

Soil quality evaluation traditionally applied the maturity index (e.g., Yeates & Bongers, 1999; Mulder et al., 2005). This life-history index has been useful to ecologists in measuring the impact of stressors, suggesting a more easy functional interpretation in relation to disturbance (Mulder et al., 2005). Other indices based on nematode surveys that were more extensively used were the Channel index - CI (e.g., Ferris et al., 2001; Li et al., 2007) and the Plant-Parasitic index - PPI (e.g., Bongers & Bongers, 1998; Villenave et al., 2001; Tsiafouli et al., 2006). More recently, a few soil quality indices were also developed regarding soil mesofauna, particularly the “Arthropod Acidity Index” (The Netherlands) (van Straalen & Verhoef, 1997, van Straalen, 1998), based on the pH preferences of arthropod

communities, and the 'Qualità Biologica del Suolo' (QBS, Italy; Parisi, 2001; Gardi et al., 2002; Parisi et al., 2005), based on an eco-morphological range of arthropod' edaphic adaptations. Furthermore, since the International Conference on the Assessment and Monitoring of Soil Quality (USA, Rodale Institute, 1991), scientists have sought to develop indices for evaluating soil quality functions by combining and integrating specific soil quality elements, with the main purpose of monitoring sustainable production (Arshad & Martin, 2002). Therefore a few soil quality indices (SQ) were proposed, integrating abiotic (physical and chemical) parameters with productivity components (e.g., Doran & Parkin, 1994), as well as biotic parameters (e.g., Parr et al., 1992). The aim was the implementation of a monitoring framework for land management which could be up-scaled from the farm level to the regional, national or even global levels and it's wide use in soil quality assessments (Arshad & Martin, 2002). However, this specific goal of soil quality indicators is still far from being achieved, since, up to now, there is not a well-defined universal methodology to characterise soil quality and to define a set of clear indicators (Doran & Parkin, 1994; Dumanski & Pieri, 2000; Bouma, 2002).

Indicator types: Biotic and abiotic indicators

Concerning the main processes and abiotic parameters, 21 % of SCIE references regarded "concentrations of major elements" in soil, 15 % soil respiration, 8 % both N mineralization and litter mass loss measured with litter bags, 7 % both pH and bait lamina, 5 % soil organic matter (SOM) content. All other abiotic indicators were reported for less than 5 % of the references.

Almost half of the biotic indicator types encompassed community composition assessments. This result was a consequence of the high diversity of compositional indicators within meso and macrofauna, as more than 60% of the indicator types found in each of these groups comprised compositional indicators. This traditional community approach, based on taxonomical identification of soil invertebrates, had already arisen in the early 1960s (Breure et al., 2005) and has been the main approach used in several soil quality assessments and monitoring schemes.

Indicator types: Traits

Life-traits and other functional indicators are more prevalent than the community composition approach for microfauna and micro-organisms. Trait indicators (e.g., microbial biomass parameters) encompassed far more than one third of the micro-organism indicator types, followed by other functional indicators (e.g., community-level physiological profile – CLPP, biochemical parameters such as enzymatic activity) with more than one fifth, while compositional indicators were represented with only one fifth. Therefore, in contrast to soil animals and plants, micro-organism communities have been traditionally characterized by physiological approaches, i.e., their function and activity, instead of the compositional taxonomic approaches. A large number of methods for assessing microbial function and activity were lately developed, some of which are already internationally

standardized (Winding et al., 2005). Nevertheless, in recent years, much effort has been spent on using structural aspects for characterization of the microbial community. For this purpose, molecular methods (e.g., Fitter et al., 2005; Fierer & Jackson, 2006; Gomez-Alvarez et al., 2007) as well as the determination of single micro-organisms or microbial groups using cell components (e.g., phospholipid fatty acids: Spiegelberger et al., 2006; Chen et al., 2007) have been successfully applied (Winding et al., 2005). Regarding microfauna, namely nematodes, the predominance of trait indicators was even higher, achieving 46 % of all indicator types against the 36 % of compositional indicators. This result was not surprising as within the last decade a great evolution occurred from traditional compositional approaches to nematode life-traits as soil indicators, such as the trophic groups and life-history traits, particularly the so-called maturity index (Bongers, 1990; Bongers & Bongers, 1998; Yeates & Bongers, 1999; Mulder et al., 2005). Life-history traits were also developed for some meso and macrofauna, such as enchytraeids (e.g., Jänsch et al., 2005), earthworms (e.g., Römbke et al., 2005) and mites (e.g., Behan-Pelletier, 2003; Ruf & Beck, 2005). However, the response trait most commonly used within soil fauna is functional composition based on trophic groups (e.g., mesofauna: Zaitsev et al., 2002; Gormsen et al., 2006; macrofauna: Olechowicz, 2004; Nahmani et al., 2006) although its application as indicators was not extensive in soil quality assessments, compared to the traditional approach of soil fauna abundance, biomass and species composition.

Spatial scales

Soil indicators have been used most frequently at local and farm scales (about 40 % each), mainly as a consequence of the predominance of ad-hoc works based on biotic compositional indicators used in direct assessments. In fact, within compositional indicators only 4 % comprised assessments at the regional scale, against the 15 % of regional scale assessments concerning functional indicators. The prevalence of indicators at the local scale is closely linked to the soil complex patterns of variability within small spatial units (e.g., a farm) (Nortcliff, 2002; Svoray & Shoshany, 2004; Bestelmeyer et al., 2006). This is particularly true for indicators based on taxa with small body sizes and narrow home ranges (Chust et al., 2003), such as micro-organisms and soil invertebrates. In fact, species composition, number and biomass of micro-organisms and soil invertebrate groups, are extremely dependent on the local site properties and the land-use form to be classified and assessed (forest, grassland, crop site) (Schloter et al., 2003; Broos et al., 2007). Although the patchiness attribute of many soil indicators (e.g., soil invertebrates) has been a useful tool at narrow scales, allowing the assessment of ecological soil quality at high resolutions (Beck et al., 2005), this feature also makes it more difficult to establish a systematic monitoring framework at regional or national scales. To reach this goal, biological concepts for the classification and assessment of soils, based on reference data for soil organisms, were developed in various countries, such as the United Kingdom (SOILPACKS, Weeks et al., 1998), The Netherlands (BISQ, Schouten et al., 1997) and Germany, (BBSK, Römbke et al., 1997). Most of them are based on similar concepts from limnology and aquatic ecotoxicology that are already routinely applied (Wright, 2000; Breure et al., 2005; Römbke & Breure, 2005). For

instance, the soil biological site classification concept (BBSK) is a monitoring system based on a limited number of "site types" in a certain region, each with a well characterised soil community (the ecotypes). Therefore, the evaluation of a soil is possible by comparing the observed (sampled) community with the expected community for a given site, i.e. under reference conditions (Ruf et al., 2000). Moreover, in the biological indicator for soil quality (BISQ), the combination of biotic and abiotic measurements in the same monitoring program leads to the possibility of deducing response models for individual indicators. With such models based on the dataset obtained, predictions can be made about effects of environmental and human impact scenarios, which are important for soil monitoring from local to national scales (Breure et al., 2005). Multi-scale indicators based on soil process models have been mainly used in drought and erosion risk assessments (e.g., Svoray and Shoshany, 2004; Ludwig et al., 2007) as well as in soil contamination evaluations (e.g., Stein et al., 2001), through remotely sensed data.

Indicators of biodiversity and ecosystem services

Soil habitats comprise one of the most diverse assemblages of living organisms on earth (Giller et al., 1997). For instance, according to recent estimations soil animals may represent as much as 23 % of the total biodiversity that has already been described (Decaëns et al., 2006). The activities and interactions of soil organisms are responsible for the ecological processes in the soil that provide the basic conditions for ecosystem maintenance. However, soil biodiversity has been neglected as an integral part of ecological quality and as a key issue for "sustainable development" despite the adoption of the concept of "the sustainable use of biodiversity" in the United Nations Conference on Environment and Development in Rio de Janeiro (Agenda 21: UNCED, 1992) (Römbke & Breure, 2005). In fact, according to the SCIE references, two thirds of the indicators have been mainly used for habitat quality assessment and monitoring purposes, almost one fifth (24 %) for ecological status and functioning purposes, whereas only 8 % have been used as biodiversity indicators. Moreover, the majority of indicators used for biodiversity evaluations are mainly related with the uppermost soil layers, the soil surface and the litter layer, such as beetles (particularly ground-beetles) and spiders, as well as plants (13 % of each were used as biodiversity indicators), i.e., not strongly related to edaphic biodiversity. Nevertheless, organizations such as the FAO and OECD have already started initiatives towards the sustainable use of soil biodiversity (FAO, 2003a,b; OECD, 2004). Since sustainability assessments require constant monitoring, this need was recently realized by the European Union in its first activities for soil protection (EU, 2002), where the monitoring of soil quality as a habitat for soil organisms is one of the main topics. The protection of soil biodiversity was explicitly highlighted in the report of Task Group 3 'OM and Biodiversity' of the Working Group on Loss of Organic Matter (Van Camp et al., 2004). The implementation of this initiative will also be connected with the protection of soil services as these are highly dependent on the activity of many types of soil organisms (from micro-organisms to a wide variety of invertebrates like earthworms and mites) (Römbke & Breure, 2005). However, according to the most recent draft of the Soil Framework

Directive (SFD; EC, 2006), the assessment of soil biodiversity is hampered by the lack of knowledge on soil organism communities in several European regions. This means that soil biodiversity is not discussed to the same degree as other threats listed in the SFD such as soil erosion or contamination.

According to the service classification of the Millennium Ecosystem Assessment (MA, 2005a), soil indicators are mainly related with supporting services, which comprised almost half of the indicators found in SCIE references, as well as regulatory services, comprising almost 40 % of the indicators. Within supporting services the main processes provided by the overall soil indicators are nutrient cycling (28 %), decomposition of organic matter (24 %), soil formation (21 %) and primary production (14 %). Several soil components, biotic and abiotic, are involved in these services in a complex network of interactions. For example, the mixing of organic and mineral soil particles including the formation of clay-humus complexes, the stabilization of soil particles by mucous substances, and the decomposition and mineralization of organic material (e.g., litter), as well as the facilitation of its availability to plants, are activities performed by various interacting groups of invertebrates and micro-organisms (Römbke & Breure 2005). Provisioning and cultural services were less represented, although 9 % of soil indicators were connected with provision of habitat. For instance, indicators such as the so-called 'ecosystem engineers' (mainly earthworms, termites and ants), through their burrowing and casting activities, modulate the availability of resources to other soil inhabitants, e.g., micro-organisms and plants (Jones et al., 1994; Jouquet et al., 2006). These relations of soil organisms to processes such as nutrient cycling, decomposition, soil formation and plant production, are well documented in the literature (e.g., Lavelle et al., 1997; 2006; Helling & Larink, 1998; Brown et al., 2004; Cole et al., 2006; Jiménez & Lal, 2006; Ortiz-Ceballos et al., 2007). Moreover, despite the fact that soil life support functions could best be based on the direct measurement of processes (indicators such as N mineralization, litter mass loss, aggregate stability, bulk density, water holding capacity), it had been more practical to use species composition, aggregated in functional groups (effect life-traits, such as biomass of decomposers, proportion of different trophic groups), as an indicator for the capability of processes (Heemsbergen et al., 2004; Beck et al., 2005). This is particularly true for biodiversity assessments due to the common soil feature of ecological redundancy which impedes the quantification of the relationship between ecosystem functioning and biodiversity. In fact, the maintenance of a process is possible while species composition has changed or degraded. Hence, biodiversity protection cannot be guaranteed by the simple measuring of process values. Functional dissimilarity (e.g., Heemsbergen et al., 2004) based on species composition or life-traits seems to be the best indicator to assess and monitor soil life support functions and the sustainable use of soil biodiversity.

3.5 Wetland ecosystems

Wetlands are some of the most threatened ecosystems of the world, partly because of the historical significant loss of wetlands linked to human hydrological alteration of the landscape (i.e. draining and filling of wetlands). However, threats to wetlands have become especially severe, because their importance as highly diverse systems of functional importance for humanity has only very recently been acknowledged widely by non-scientists.

Indicator types

Wetland assessment is presently driven by the management of environmental protection, recreation, aesthetics and production of renewable resources (Mitsch & Gosselink, 2000). Depending on the management focus, different sets of indicators have been identified belonging to very different ecological conceptual approaches, ranging from species indicators used to identify wetland habitats (Tiner, 2006) to the quantification of biological and physico-chemical processes (see a review in Gutknecht et al., 2006). The highly diverse and heterogeneous nature of wetland systems is reflected in the diversity of research approaches used for their environmental assessment. A total of 45 references have been examined, four of which were based on hybrid assessment systems combining abiotic-biotic indicators. The remaining references corresponded to biotic (72 %), abiotic (26 %) and landscape (2 %) indicators.

Due to their inherent nature as ecotones between terrestrial and aquatic systems, wetlands need to be delimitated within the landscape. Geographic information systems are advanced tools for wetlands delimitation, usually based on the coupling between remote sensing interpretation and field data. Physical indicators, like the NIR/blue ratio can discriminate among moisture-driven land use/land cover variations, wetland types and the extent of moisture stress (Dupigny-Giroux, 2007). Such indicators can also be used to develop indices of human disturbance (the level of human induced impacts on the biological, chemical, and physical processes of surrounding lands or waters) e.g., the landscape development intensity (LDI) index at the watershed scale. Based on land uses and land cover, the LDI can be applied using available GIS land use/land cover data, aerial photographs, or field surveys (Brown & Vivas, 2005). Other indicators like total area of wetland (linking response to water levels) are used to assess the effects of different regulation plans under current and future (climate change) water-supply scenarios (Hudon et al., 2006).

Plant-lists enumerate indicator species that grow in wetlands and that could be used to identify wetlands according to defined classification systems (Cowardin et al., 1979; Tiner, 2006). Plant-lists are part of inventories (listing observable or measurable physical, chemical and biological features), methods gathering quantitative information to classify wetland types (Innis et al., 2000). Wetland classification based upon hydrological and geomorphic settings allows their functional assessment

using a combination of abiotic and biotic indicators (Adamus et al., 1991; Brinson, 1993; Brinson et al., 1995; Brinson, 1996). The classification suggested by Brinson (1993) is used to evaluate wetland functions:

- assessing the physical, chemical and biological functions of wetlands,
- being useful for comparing the level of functional integrity of wetlands within the same functional class or
- evaluating the impact of proposed human activities on wetlands and mitigation alternatives.

Indicator types: Biotic and abiotic indicators

Linking abiotic and biotic indicators, process-based and biochemical or gene-based assays are becoming more and more important as we seek a mechanistic understanding of the response of wetland ecosystems to current and future anthropogenic perturbations (Gutknecht et al., 2006). Thirteen percent of the abiotic indicators are used to evaluate functional processes, either by direct measurement of chemicals or model equations. Wetland processes being mediated by the biota (i.e. nitrification, denitrification and methanogenesis) and/or by particular abiotic environmental features are characterised at local scale wetlands. This variation in the chemical-nutrient compartments reflects the response of nutrient cycling to fluctuating hydrology and nutrients. The use of chemical indicators in wetland assessment is based on the assumption that wetlands perform specific hydrological and water quality functions (Fisher & Acreman, 2004). Some of the chemical measurements correspond to classical indicators of wetland eutrophication such as nitrate, phosphorus and dissolved oxygen (Kennedy & Murphy, 2004; Stratford et al., 2004a), while others relate to bio-accumulation processes of importance under pollution by heavy metals (Evers et al., 2007). In general rates of retention or removal of contaminants and nutrients are measured to evaluate wetland functioning as either sinks or sources in relation to nutrient loadings to water bodies, restoration programmes and creation of treatment wetlands (Knox et al., 2006; Jansson et al., 1998; Fisher & Acreman, 2004).

The biochemical indication, addresses biotic biomass evaluations over disturbance gradients, either nutrient enrichment or diverse pollution sources from base trophic levels, including bacteria (Cordova-Kreylos et al., 2006) and algae (McNair, 2003), to higher predatory levels, i.e. the European eel (Teles et al., 2007). The biochemical indicator methods also approach functional integrative measures of food web structure and its response to hydrology and nutrient enrichment (Sierszen et al., 2006).

Within the biotic indicators, 55 % correspond to the species and community level of a very diverse array of flora and faunal groups, from micro-organisms (algae and bacteria) to vertebrates. All of these may be used in general as biodiversity indicators for conservation evaluation, supporting wildlife enhancement in wetlands (Stefanescu et al., 2005; Calhoun et al., 2002; Paillisson et al., 2002; Garcia,

2007), and including a wide variety of floral and faunal groups, for example, bacteria, algae, herbaceous plants, trees, invertebrates, amphibians, fish, birds etc.

Forty-eight percent have been developed with the aim of indicating the effects of human activities on ecological functions, health, or integrity of wetland systems, for instance:

- Index of Biological Integrity (IBI, U.S. EPA, 1998),
- European Water Framework Directive Ecological Status (Boix et al., 2005),
- Wetland Biological Condition (Lougheed et al., 2007) and
- Ecological Quality according to Seilheimer & Chow-Fraser (2006).

Biological indicators that aim at the classification of water quality in general, include:

- single community structural metrics (e.g., richness, abundance, etc.)
- ecological diversity indices including added effects of weighted tolerance values of invertebrate species (Boix et al., 2005),
- multimetrics (U.S. EPA, 1998), and
- predictive models, such as RIVPACS (Davis et al., 2006).

Indicator types: Traits

Species traits in wetlands (8 % of SCIE references) are used for the setting of a vast array of management objectives, and also to assess their success, from wetland identification goals by plant species whose indicator status reflects its frequency of occurrence in wetlands (Tiner, 2006), to ecological life history traits reflecting the adaptation of species to the environment. Life history attributes of butterflies can also be related to habitat types (Stefanescu et al., 2005), whilst a water bird community showed trait responses to the management regime of a wet grassland system, small changes in the water regime and hunting disturbance (Paillisson et al., 2002). Species traits are also increasingly being considered in building up multimetric indices covering a range of structural and functional community metrics, for example the Index of biological integrity (U.S. EPA, 1998).

Spatial scales

Eighty-three percent of the indicators are used at scales ranging from local to regional, emphasising the need for small scale indication due to wetland heterogeneity and diversity. The greater sub- and global scales of indication corresponding to landscape indicators are of significance in evaluating large scale management strategies (e.g., Jansson et al., 1998) and wetland evaluation in relation with other less valuable ecosystems (Costanza et al., 1997).

Relation to management

Wetland classification schemes highlight the great diversity of wetland types (Finlayson & van der Valk, 1995). Moreover, the high diversity of habitats in wetlands is mainly regulated by habitat

heterogeneity and the organisms themselves (i.e. plant structure within the water column). Nonetheless there are not many habitat indicators for wetlands, even if some approaches (e.g., evaluation of wetland condition, U.S. EPA, 2002) deal with the evaluation of habitat and hydrology alteration.

Historical wetland alteration and destruction, mainly caused by land conversion to agricultural use, has reduced the wetlands that existed in the world since 1900 by up to 50 % according to some estimates (OECD, 1996). Present management of wetland habitats has been focused towards enhancement of wildlife habitats to increase biodiversity. Hudon et al. (2006) provide an example of linking wetland response to water levels and assessing the effects of different regulation plans under current and future (climate change) water-supply scenarios for the total area of wetland. Restoration or creation of wetland habitats as "treatment wetlands" to improve water quality in surface waters makes use of indicators in assessing success (Stratford et al., 2004b; Acreman et al., in press). Due to human modification of natural hydrology, many restoration programmes are based on the management of water level, and the success of these programmes is being evaluated with biodiversity biological indicators.

Thirty-one percent of the indicators are used for the habitat management of wetlands, addressing conservation objectives from the EU Habitats Directive and NATURA 2000 network, while ca. 38 % are indicators for the Water Framework Directive and nearly 13 % are indicators considered in National Water Quality Monitoring schemes.

Indicators of biodiversity and ecosystem services

Wetland biodiversity covers aspects of diversity of species, genes and ecosystems, and the diversity and functions of the whole ecosystem of which they are part. Accordingly, most indicators of the wetland database are directly or indirectly addressing biodiversity. The strong focus on wetland biodiversity for conservation reflects the high representation in the database of biodiversity indicators. Meanwhile, recent comparative studies on wetlands across world wide regions point at the lack of proper assessment on wetlands biodiversity, thus limiting wetland comparisons, and the need for increase research and conservation of biodiversity in wetlands (Gopal & Junk, 2001). Wetlands are appreciated for their provisioning of wetland habitats and species. Indeed a high percentage of the indicators (65 %) are related with provisioning services such as genetic resources or biodiversity, and other products obtained from wetlands such as freshwater (14 %), habitat (13 %) and food (6 %). A key issue in wetlands is the relationship between the conservation of genetic resources, species and habitats, and the ecosystem provision of basic needs of life, which promotes the need of integrated management strategies and human sustainable use of wetland resources. Wetlands are some of the most productive ecosystems in the world. Wetlands support high productivity of plants but not always high plant diversity, while animal communities are more diverse (Zedler & Kercher, 2005). It is the presence of water, high plant productivity and other habitat qualities that attracts high numbers of

animals and animal species, many of which depend entirely on wetlands (Zedler & Kercher, 2005). Still there is a need for further research on hypotheses relating biodiversity to system stability, productivity and species redundancy in wetlands. If we aim for effective management, conservation and restoration of wetlands, an important driving question is "how could species loss and changes in community composition affect ecosystem functioning, resilience after disturbance, and services to humans?"

The wetland indicators are highly related to important ecosystem services such as self-purification (44 %), water retention (20 %) (i.e. flood mitigation, storm abatement and aquifer discharge), waste detoxification and decomposition (15 %) and disease regulation (12 %), the latter two of human health concern. At the regional and global scales wetland functions maintain water and air quality, and influence global cycles of nitrogen, sulphur, methane and carbon dioxide (Mitsch & Gosselink, 2000). The indicators emphasise the importance of ecosystem functions in relation to the support of nutrient cycling (30 %), photosynthesis and primary production (38 %), decomposition (16 %) and water cycling (15 %), all of which are important regulation processes on wetland productivity.

Some evaluation techniques, such as the classification of Brinson (1993) provide functional evaluation related to ecosystem values. Following Costanza et al. (1997) wetlands are valued in second place after estuaries as being most valuable for ecosystem services at the world scale.

3.6 River ecosystems

History and indicator types

Altogether, 167 references were used for this study, about half of which describe the development or application of single indices, either biotic or abiotic. The application of biotic indices for monitoring rivers dates back more than 100 years, when the first saprobic system was developed in Germany (Kolkwitz & Marsson, 1902; 1908). The underlying principle of biotic indices has not fundamentally changed since then. Saprobic values are being allocated to species – mainly micro-organisms and benthic macroinvertebrates – which are known to be sensitive (low value) or tolerant (high value) to pollution with organic substances. The average saprobic value weighted by the species' abundance is the Saprobic Index. Saprobic indices are still widely used in Europe (overview in Rolauffs at al., 2004) and have several times been adapted to changing demands in water quality monitoring (Friedrich, 1990; Rolauffs et al., 2004), most recently to fulfil the requirements of the European Water Framework Directive (Friedrich & Herbst, 2004). While the first approaches targeted steep pollution gradients, Saprobic Indices are more recently addressing subtle differences in organic load and differentiate between river types.

Another large group of indicators (18 % of all single indices) address the assessment of eutrophication with benthic diatoms (see also Kelly, 1998). Diatom indices adapted to regional conditions have recently been developed for Eastern Canada (Lavoie et al., 2006), the USA (Fore & Grafe, 2002; Wang et al., 2005), South Africa (Bate et al., 2004), China (Tang et al., 2006), Brazil (Salomoni et al., 2006), Nepal and India (Juttner et al., 2003), Italy (Torrisi & Dell'Uomo, 2006) and Australia (Dela-Cruz et al., 2006). Macrophytes are less frequently used to indicate eutrophication intensity, however, some systems have been developed recently, particularly in Europe (Meilinger et al., 2005; Soszkiewicz et al., 2006). Trophic indices follow the same principal as saprobic indices: Sensitivity/tolerance values for species are subsumed and result in a final index, typically the weighted average of the tolerance values.

While saprobic and trophic indices address the intensity of a single stressor, multimetric indices consider functional aspects of the riverine community (23 % of the indices). Karr (1981) developed an Index of Biotic Integrity (IBI) for fishes in North America using 12 component indices (metrics). The IBI includes, for instance, the proportion of feeding types, spawning habits or migration types, all of which are community-based measures. The use of numerous functional metrics, each reflecting a different aspect of the community, provides a more general result, which may respond to the effects of multiple stressors. These multimetric indices form the basis of the US rapid field assessment and river monitoring (Barbour et al., 1999; Karr & Chu, 1999, 2000) and are also widely used for river assessment and monitoring according to the European Water Framework Directive, covering macroinvertebrates (Hering et al., 2004; Furse et al., 2006), fish (FAME consortium, 2004; Pont et al., 2006) and macrophytes (Schaumburg et al., 2004).

An approach comparable to multimetric indices aggregates (abiotic) environmental parameters, such as land use, habitat or chemical parameters (24 % of indicators regarded). Raven et al. (1997) developed the River Habitat Survey (RHS). It is based on an extensive field survey of hydrological and morphological parameters recorded at ten spot checks along a section of 500 m river length, such as the bed substrata, flow patterns, bank and riparian vegetation and modification (weirs, dams, or bank enforcement). Two indices are computed based on field records: the Habitat Quality Assessment and the Habitat Modification Score. The system is well-established in the U.K. and part of the environmental framework of RIVPACS (see below). Principally, it is applicable in the major European ecoregions: lowlands, mountains, Alps and Mediterranean (Buffagni & Kemp, 2002; Furse et al., 2006; Szoszkiewicz et al., 2006). Similar field survey-based methods have been developed for Germany (LAWA, 2000; national standard for Germany) and Portugal (Oliveira & Cortes, 2005). The German "Strukturgütekartierung" considers 26 parameters classified into six groups: channel course, longitudinal and transverse profile, bed substrata, bank character and surroundings. The parameters are recorded along 100 m channel segments and compared to a reference condition. Flotemersch et al. (2006) summarised several multimetric habitat indices developed in the USA, for instance, the non-

wadeable stream habitat index for Michigan (NWHI, Merritt et al., 2005; Wilhelm et al., 2005) or the qualitative habitat evaluation index for Ohio (QHEI, Rankin, 1989).

A recent and relatively small group of habitat indicators measures and validates river restoration success (3 % of SCIE references). This group is gaining increasing attention, particularly as a consequence of recent changes in European water policies. For instance, Rohde et al. (2004) introduced several landscape metrics to assess river restoration, while Woolsey et al. (2007) presented 49 indicators to measure river restoration success in Switzerland at regional and local scales. Proposed indicators address: project acceptance, stakeholder participation, recreational aspects, landscape aspects, longitudinal connectivity, hydro-geomorphologic and hydraulic parameters, measures of bed load and organic material and river bed conditions.

Six percent of indicators comprise predictive models, using environmental predictor variables to model community composition under the given environmental conditions. The RIVPACS system (Armitage et al., 1987; Wright et al., 1984, 1989) considers more than 730 reference sites in the U.K. to predict the macroinvertebrate community of any test sites. The observed community is compared to the modelled expected reference; the deviation is equivalent to the degree of impairment. This approach has been modified for Australia (AusRivAS; Smith et al, 1999), while Pont et al. (2006) use a predictive system for calculating the expected fish community according to the site's locality and abiotic characteristics (altitude, temperature, physico-chemical conditions, hydromorphology). By comparison with reference conditions, the model predicts ten single related metrics, which are finally combined into a multimetric index, the 'European Fish Index' (FAME consortium, 2004). Reynoldson et al. (1995) developed predictive models that relate site habitat attributes to an expected community, the BEnthic Assessment of SedimenT (BEAST). At the pan-European scale, Brack et al. (2005) recently developed models for the impact of environmental pollutants (MODELKEY). MODELKEY built on existing studies linking chemical pollution in European river basins to measurable ecotoxic effects and aimed at forecasting the risks of key pollutants on both freshwater and marine ecosystems at the river basin scale (including the marine environment). Self-organising maps and artificial neural networks are used for predicting the communities of diatoms (Park et al., 2006), benthic invertebrates (Park et al., 2004) and fish (Guegan et al., 1998). Predictive models have also been developed for environmental variables. The Physical Habitat Simulation System (PHABSIM, e.g., Maddock, 1999) is based on field measurements of channel shape, water depth, velocity and substrates. PHABSIM provides simulations of the quality and quantity of existing habitat *vs.* flow relationships.

Indicator types: Traits

Townsend & Hildrew (1994) introduced another group of indicators referring to functional community aspects: the species traits. Species traits are biological attributes, such as life cycle, feeding behaviour, reproduction, or body size, which are supposed to be controlled by habitat conditions and reflect

environmental stress on the habitat. Traits were initially utilised to test the habitat template theory (Townsend & Hildrew, 1994; Townsend at al., 1997). An overview of benthic macroinvertebrate species traits has been given by Usseglio-Polatera (2000), while Dolédec (1999), Bady (2005) and Statzner (2001, 2005) addressed their potential for river biomonitoring. Pont et al. (2006) apply fish traits for river assessment at a European scale. In multimetric indices, species traits already constitute part of the component metrics (e.g., Lorenz et al., 2004). Altogether, 12 % of SCIE references refer to this indicator group. An increasing application of traits in river assessment and monitoring is expected in the future, since species and community traits potentially provide valuable insight into the functional composition of riverine communities and the functional aspects of biodiversity.

Spatial scales

River assessment usually aims at indicating the water quality and – more recently as a consequence of the Water Framework Directive (WFD) – the overall ecological quality. The assessment is usually based on field sampling at the local scale (water body-related). Upscaling has been possible, with some limitation, to the national level, at which water and river quality maps are regularly compiled and published. In theory, river bioassessment should be possible at the regional or even sub-global scale in the near future, since the WFD requires comparable assessment at a pan-European scale. However, to achieve this goal, additional effort will be necessary to make national indicators and indication systems comparable throughout Europe, which is still aimed at the so-called intercalibration exercises.

The river habitat indices presented above also depend on a number of parameters recorded in the field and, thus, inherently represent local- to regional-scale methods, too. At larger spatial scales (landscape, sub-global, global) the increasing availability of powerful GIS tools recently stimulated the use of remote sensing for river quality assessment, however, they still constitute only 5 % of the total indicators. Tiner (2004) described a method for monitoring the general condition of natural habitat in watersheds based on remotely-sensed data. Likewise, Jones et al. (2007) used large-scale indicators derived from the 1990/2000 CORINE dataset on land cover (EEA, 2006) and digital elevation models, for a cross-European landscape analysis, for instance, '% land cover change near rivers' or 'change in cropland on > 3 % slopes'. Revenga et al. (2000) introduced a global river fragmentation index and applied it to 227 rivers worldwide. Vorosmarty et al. (2005) presented indicators of emerging water stress based on geospatial measures (metrics).

Indicators of biodiversity and ecosystem services

Nearly a fifth of the SCIE references (19 %) directly or indirectly account for biodiversity. These indicators mainly refer to species richness, either of the river community (e.g., Angermeier & Winston, 1997; Ferreira et al., 2005; Jacobsen et al., 1997) or the riparian area (e.g., Fierke & Kaufmann, 2006). Although the use of species richness as a proxy for biodiversity has often been

criticised (e.g., Angermeier & Karr, 1994; Sanjit & Bhatt, 2005), more generic approaches are still in their infancy. Biodiversity has many components: genetic diversity, species number and abundance, community composition, range of functional traits and related processes, and spatial distribution (Diaz et al., 2006). Generic concepts for rivers aim at the quantification of ecosystem processes and functional diversity (biological integrity *sensu* Angermeier & Karr, 1994) and the analysis of species traits (Charvet et al., 1998; 2000; Dolédec et al., 1999; Gayraud et al., 2003; Bady et al., 2005; Statzner at al., 2005; Bêche et al., 2006), which are frequently analysed by fuzzy correspondence analysis (Chevenet et al., 1994). Unlike Bady et al. (2005), who presented an 'Index of Functional Diversity' for biomonitoring large rivers at a European scale, most authors focus on species traits as indicators of environmental stress. 89 % of all references are applied for quality assessment and trend monitoring, particularly if the multimetrics incorporate various diversity measures. The 2002 inventory of biodiversity indicators in Europe (EEA, 2003) listed 655 existing biodiversity indicators, only 25 of which directly refer to rivers (water) and only one accounts for (fish family) biodiversity.

"Biodiversity has well-established or putative effects on a number of ecosystem services mediated by ecosystem processes" (Diaz et al., 2006). Following the service classification of the Millennium Ecosystem Assessment (MA, 2005a), rivers provide food (fish, crayfish, water) and regulate ecosystem processes (nutrient transport, decomposition, self-purification), they provide cultural and aesthetic services (religious, spiritual and recreational values, tourism), and support nutrient and water cycling. The specific role, however, that biodiversity plays for service provision of rivers has not yet been clearly and adequately defined. The provision of fish, for instance, is often linked to a couple of (economically important) target species, which, moreover, may be alien to a river system (e.g., Rainbow trout *Oncorhynchus mykiss* in European rivers). The service 'nutrient and sediment buffer' depends on specific characteristics of the riparian vegetation (see Naiman et al., 1993 for a general overview). The buffer width and density, the composition of the riparian plant community, the growth forms and their zonation control the service rate. Dosskey (2001) quantified the sediment retention capacity of riparian buffers and found that 5–20 m wide grass strips retain 40–100 % sediments. According to Correll (2005) a 30 m wide mixed riparian buffer removes 92–100 % nitrate from the upper (root-penetrated) groundwater layer. Nitrate uptake was mainly controlled by riparian trees, as their roots grow deep enough to reach the groundwater layer. Therefore, functional attributes can be linked to ecological processes, whereas the combination of attributes determines the service (rate). Thus, we need to account for the multiple aspects of biodiversity *sensu* Noss (1990) in order to cover biodiversity at an appropriate level.

To our knowledge, very few studies, besides the MA (2005a), focus on river ecosystem service indicators (none of the SCIE references analysed for rivers). The MA indicators do not sufficiently cover the regional and smaller spatial scales, which are often subject to regular regional and national assessment and monitoring programmes.

3.7 Landscapes

Theoretically, landscapes exist at several spatial and temporal scales, depending on the process or organism being studied (Turner, 2005). Relationships between landscape patterns and species traits, populations, and entire biotic communities have been described at a range of scales, ranging from beetles in very small landscapes (Wiens et al., 1997) to birds, mammals, and reptiles at regional and continental scales (Robinson et al., 1995; O'Connor et al., 1996; Atauri et al., 2001; Donovan & Flather, 2002). At basin and regional scales, landscapes usually consist of a mixture of ecosystems such as wetlands, streams, forests, woodlands, agriculture, and urban settings (Wickham & Norton, 1994; Wascher, 2005). It is the interactions of these ecosystems in space and time that often determine the outcome of species and communities (Hovel & Lipcius, 2001; Murphy & Lovell-Doust, 2004).

History

The importance of spatial variability and landscape arrangement in ecology had its roots in the mid 1800s (Schreiber, 1990). By the late 1930s, aerial photography started to revolutionise landscape analysis by providing unprecedented views of the environment. Geographers and ecologists started to ask basic questions about the causes and consequences of landscape configuration on ecology, and in the late 1930s, Carl Troll, a German geographer and ecologist, coined the term landscape ecology (Schreiber, 1990; Turner, 2005). Since then, landscape ecology has been defined in many ways (Risser et al., 1984, Forman & Godron, 1986; Urban et al., 1987; Turner, 1989; Schreiber, 1990; Pickett & Cadenasso, 1995; Manel et al., 2003), but all definitions emphasise reciprocal relationships between spatial heterogeneity and ecological processes and functions (Turner, 2005). Landscape indicators have their foundation in these relationships (Turner, 1989; O'Neill et al., 1988; Jones et al., 1996; Gustafson, 1998; Lathrop et al., 2007). With the advent of new imagery from aircraft and earth observing satellites (e.g., Landsat), and advances in computing, landscape ecology and associated landscape indicators have exploded since the early 1990s (Haines-Young & Chopping, 1996; Gustafson, 1998; Turner, 2005).

Many different biophysical classifications have been derived for landscapes. Some of them classify different combinations of spatial heterogeneity of primary land-surface attributes such as land cover (Wickham & Norton, 1994), some classify spatial variability in biophysical attributes and patterns (Detenbeck et al., 2000; Jensen et al., 2001; Jongman et al., 2006), whereas others attempt to capture spatial variation and patterns of environmental, cultural features and history (Mucher et al., 2003; Wascher, 2005). In some cases landscapes are classified based on their ability to capture fundamental disturbance processes such as fire (Rollins et al., 2004). Significant increases in the number and availability of digital spatial databases at multiple scales have facilitated development of many of these classification systems.

Indicator types and spatial scales

Simple measures of landscape composition and pattern originated from island biogeography theory (MacArthur & Wilson, 1967), before digital database were available. Patch sizes and distances have been used to estimate emigration and immigration processes, as well as extinction probabilities of species across a range of spatial scales (MacArthur & Wilson, 1967; Brown, 1971; Martin, 1980; Jones et al., 1985). Generation and use of landscape metrics in environmental and ecological research and assessments has exploded since the late 1980s, but especially since the late 1990s (Gustafsen, 1998). Proliferation of landscape metrics has resulted from increased availability of digital databases of land cover, vegetation distribution, soil characteristics, topographic characteristics, and other biophysical data via websites and data portals, and because of advances in computer processing speed, the amount of data that can be stored and processed, and software development (e.g., image classification software, geographic information systems or GIS, and statistical and modelling packages; Jones et al., 2005). Development of the field of landscape ecology in the mid-to-late 1980s, and interest in relationships between pattern and process, also fuelled development of landscape metrics and indicators (Risser et al., 1984; Forman & Godron, 1986; Urban et al., 1987; Turner, 1989; Turner, 2005).

Landscape metrics include a wide variety of measures of spatial composition and pattern of habitats, land-cover and land-use, ecosystems, and other land-surface features over a given area of interest (O'Neill et al., 1988; Haines-Young & Chopping, 1996; Gustafson, 1998; Lathrop et al., 2007). These include measures of composition (e.g., the amount or proportion of habitat in an area), shape complexity (e.g., fractal dimension), patch size, connectivity (e.g., fragmentation, contagion, and percolation measures), distance measures (e.g., between patches), and attribute diversity and/or complexity. Areas of analysis can include very local scale analysis of habitats (Wiens et al., 1997), within specific parts of a landscape (e.g., riparian ecosystems, Baker et al., 2006; Jones et al., 2007; Rheinhardt et al., 2007), within and across catchments (Jones et al., 1997; Jennings & Jarnagin, 2002; Walker et al., 2002), and across entire regions (Jones et al., 1997), continents (Wade et al., 2003; Jones et al., 2007), and the globe (Riitters et al., 2000). Landscape metrics also have been used in near-shore and coastal water habitats (Hovel and Lipcius, 2001).

A landscape metric becomes an indicator when qualitative and quantitative relationships are established (Jones et al. 1996), although the terms "landscape metrics," landscape indicators,", and "landscape indices" have often been used interchangeably. In this way the metric becomes an indicator or surrogate of important biophysical processes, ecological states, or pressures. Mander et al. (2005) make a similar distinction, describing metrics and indicators as either structural or functional. Structural indicators include the wide range of metrics generated from programs like FRAGSTATS (McGarigal & Marks, 1995), whereas functional indicators are more directly linked to processes. In many cases, landscape metrics and indicators have been used in models (empirical and process

models) to evaluate the response of specific environmental themes to changes in landscape composition and pattern.

Landscape metrics, indicators, and models have been used to evaluate individual species distributions, as well as patterns of species richness and diversity. Some studies have documented significant quantitative relationships between species distribution and home ranges and landscape composition and pattern. Kie et al. (2002) predicted mule deer distribution and home ranges. They found mule deer home ranges to be inversely correlated to edge density, mean shape, and fractal dimension, and positively correlated with contagion, across all scales of investigation. Significant correlations with mean patch size, patch richness, and an index of mean edge contrast were scale dependent. Verboom et al. (1991) found correlations between extinction rates of European nuthatch metapopulations and patch size and quality, and correlations between colonization rates and the density of surrounding patches occupied by nuthatches. Forest edges appear to be important for migration in some species, such as chickadees (Descrochers & Fortin, 2000), but also may have negative effects on migratory bird populations by increased predation and nest parasitism by cowbirds (Robinson et al., 1995). Jones et al. (2000) found a significant negative correlation between interior forest bird abundance and distribution and forest edge across a large geographic region of the U.S.

Several studies have attempted to evaluate the relative roles of habitat fragmentation and basic habitat distribution and quality in accounting for species distribution, community structure, and extinction probabilities. Results of these studies are mixed. Although some studies suggest a potential role of fragmentation in determining species distributions and community structure (McCollin et al., 1993; Kattan et al., 1994; Robinson et al., 1995; Jones et al., 2000, Donovan & Flather, 2002), studies that incorporate experimental designs to tease apart the influences of habitat fragmentation from habitat size and quality are rare. Habitat fragmentation has been implicated in changes in trophic levels in fragmented landscapes, including reduced food webs between fungi and insects (Komonen et al., 2000) and herbivory by insects (Valladares et al., 2006), but these changes may reflect influences of edges and habitat patch size rather than fragmentation per se. Other studies have shown a more important role of habitat quality and size, and cautioned interpretations of the impacts of fragmentation (Trzcinski et al., 1999; Hoval & Lipcius, 2001; Donovan & Flathers, 2002; Brotons et al., 2004). Lack of concordance among studies may reflect the species and scales of investigations. Individual species traits (e.g, dispersal capacity, demographic characteristics, food preferences, home ranges) may dramatically affect how species and communities respond to structure and functional attributes of landscapes (With & Crist, 1995; Hovel & Lipcius, 2001; Donovan & Flather, 2002; Brotons et al., 2004; Horskins et al., 2006).

Connectivity between patches, which may not be captured by some fragmentation metrics, has been shown to be an important determinant of species distributions and population dynamics in black beers

(Dixon et al., 2006), plant species diversity (Damschen et al., 2006), and in plant/mammal succession in old field habitats (Schweiger et al., 2000). In some cases, corridor analysis derived from maps and presence of species may be insufficient to reconstruct relations among populations connected by corridors (Horskins et al., 2006). Phylogenetic analysis (Froufe et al., 2003) and the evolving field of landscape genetics (Mech and Hallett, 2001; Michels et al., 2001; Manel et al., 2003) provide an independent validation of the influence of landscape connectivity on populations and the role of species traits. Moreover, new ways of mapping landscape corridors, such as morphological image analysis (Vogt et al., 2007) and determining functional aspects of landscapes mosaics, such as graph theory (Urban & Keitt, 2001), may improve our understanding of functional landscape connectivity.

Landscape context and patch-matrix interactions also determine population and community composition. Some species exist in isolated habitat patches and depend on immigration from more contiguous habitats in the surrounding landscape (Robinson et al., 1995; Donavan & Flather, 2002), and the surrounding matrix can dramatically affect the functional isolation of patches (Ricketts, 2001). Additionally, shapes of patches can influence the degree to which species use the surrounding matrix (Tubelis et al., 2004).

Multiple biophysical data layers have been used to predict species distributions and habitat suitability across local to continental scales. Scott et al. (2003) used multiple biophysical data layers, including land cover distribution, soils, elevation, and temperature, to model species habitat suitability across broad geographic areas. O'Connor et al. (1996) modelled bird distributions across the U.S. using a number of biophysical data layers. Temperature explained continental scale patterns of bird diversity, whereas land cover and landscape metrics became more important at regional scales, although the relative importance of different metrics varied within and among regions. Thuiller et al. (2004) reported similar results for modelling distributions of plants, amphibians, reptiles, and mammals across Europe, although their study did not permit an analysis of scaled relationships among biophysical variables. Relatively new statistical modelling approaches, such as the Genetic Algorithm for Rule-set Predictions (GARP) and Maximum Entropy, have been used to model species distributions from multiple biophysical data bases (Stockwell & Peters, 1999; Phillips et al., 2006, respectively).

Landscape indicators and models have also been used to predict biological and ecological conditions of streams (Roth et al, 1996; Snyder et al., 2003; Strayler et al., 2003; Baker et al., 2006; Donohue et al., 2006; Malony & Feminella, 2007) and estuaries (Hale et al., 2004). Landscape characteristics, including land cover at the catchment and riparian scales, are common in many of these models. An important aspect of interpreting indicators and models is to establish reference site conditions (O'Connell et al., 2000; Rheinhardt et al., 2007).

New remote sensing data and analytical approaches offer significant potential to improve landscape assessments of habitat and species distributions. These include application of LIDAR to map vertical structure and composition of habitat (Anderson et al., 2006; Hinsley et al., 2006; Goetz et al., 2007) and IKONOS imagery to map agricultural land use (Vina et al., 2003). Moreover, sensors that provide more frequent temporal coverage offer great potential to monitor habitat and vegetation dynamics and phenology over broad scales (Schwartz et al., 2002).

Landscape indicators and models have been developed to evaluate potential drivers and pressures potentially affecting individual species, communities, and habitats. These include evaluation of impervious surface (Slonecker et al., 2001), road density and distribution (Watts et al., 2007), urbanisation (Theobald & Romme, in press), marginal land use (Jones et al., 2007), climate change (Iverson and Prasad, 1998; Westerling et al., 2006), light population (Longcore & Rich, 2004), economic drivers (Wamelink et al., 2003), and non-indigenous invasive species (Allen et al., 2006).

Additionally, landscape indicators, models, and remote sensing have been used to evaluate a number of other environmental themes and ecological services, including but not limited to assessments of forest fragmentation and urbanization (Riitters et al., 2000 and Galleo et al., 2004, respectively), forest transpiration and photosynthesis (Anselmi et al., 2004), terrestrial ecosystem productivity in response to climate variability (Lupo et al., 2007), landscape change and consequences of change to ecological resources (Vogelmann, 1995; Jones et al., 2001), fire and disturbance frequency (Keane et al., 2002; Rollins et al., 2004), water quality (Behrendt, 1996; Wickham et al., 2000), water quality risks (Wickham et al., 2002), soil loss (Van Rompaey & Govers, 2002), ecological forecasting (Reynolds et al., 2000; Vallete-Silver & Scavia, 2003; Running et al., 2004), environmental justice (Mennis, 2005; Mennis & Jordan, 2005), pathogen and disease exposure in human populations (Jackson et al., 2006), risk of flooding to set insurance costs (Sanders et al., 2006), evaluation of risks to natural hazards (Wood et al., 2002), and alternative futures analysis involving a range of environmental, ecological, economic, and cultural conditions (Theobald & Hubbs, 2002; Baker et al., 2004). As such, landscape assessments offer the potential to evaluate and link all aspects of the Driver Pressure State Impact Response framework (DPSIR; Jones et al., 2007).

Although landscape metrics and indicators have been used to predict species distributions, habitat quality, ecosystem productivity, and water quality, several authors caution their use and interpretation. Li & Wu (2004) conclude that many studies employ landscape metrics to describe spatial pattern as an end in itself, without explicitly relating pattern to processes. Landscape metrics are often applied without consideration for scale-dependency of the processes and patterns they are attempting to capture (Wiens, 2001; Li & Wu, 2004; Mander et al., 2005). Additionally, many landscape metrics are correlated and may represent as few as six orthogonal axes of composition and pattern (Riitters et al., 1995). There have been mixed results in quantifying relationships between landscape metrics and

specific environmental and ecological themes or issues (Gustafson, 1998; Li & Wu, 2004; Turner, 2005). Differences in results emanate from a variety of sources, including but not limited to mismatches in data and scales of application (temporal and spatial), lack of in-situ data on specific theme of interest (e.g., species' demography and traits across space and time), lack of historical data to address legacies, and lack of spatial data that capture important biophysical features and processes (Gustafsen, 1998; Li & Wu, 2004; Jones et al., 2007). Finally, horizontal interactions among patches and biophysical conditions are largely unknown, and it is these relationships that determine flows and fluxes of energy, water, biota, and materials.

Indicators of biodiversity and ecosystem services

Because many different biological, ecological, economic, and cultural characteristics, patterns, and processes can be represented by landscape metrics, indicators, and models, landscape assessments offer significant potential to capture a range of ecosystem services, and important interactions between people and landscapes. They also offer significant potential to scale up biological traits and processes across landscapes to understand their cumulative effects on ecosystem services at broader scales. For example, the blue-green veining concept is based on revegetation and connection of field borders to increase natural predators of crop pests, thereby reducing the need for pesticides across an entire landscape. The result is lowered exposure of people and wildlife to hazardous chemical and improved water quality.

Landscape indicators and models have been used to assess multiple ecosystem services, and their relationship to economic drivers, under different alternative landscape futures (Baker et al., 2004; Naidoo & Ricketts, 2006). In these cases, ecosystem services were related to spatially explicit habitat and biophysical conditions, which in turn were linked to demands for resources. Those demands are influenced by the quality and distribution of the service, cultural values, and local, regional, and global economic drivers (Naidoo & Ricketts, 2006). Land use practices in agricultural communities have been linked to ecosystem services (Baudry et al., 2003; Dale & Polasky, 2007).

Landscape analyses offer the potential to link ecosystem services through space and time. As such, it may be possible to conduct a full cost-accounting of how optimization of one ecosystem service (e.g., flood abatement through dam construction), might affect other services. From a species and community traits standpoint, these linkages should enhance valuation of these traits at a landscape scale.

As mentioned earlier, one of the key needs in landscape assessments is to develop a better understanding of horizontal relationships among landscape components. Most alternative futures studies lack horizontal influences of different landscape elements (including position), although some studies have quantified horizontal interaction among landscape pattern and elements indicators (Urban

& Keitt, 2001; Voinov et al., 2004; Ludwig et al., 2005; Peters et al., 2006). An understanding of horizontal influences in landscapes will help mangers prioritize where to change, improve, or protect land-surface conditions.

4. Comparison of Ecosystems

4.1 Indicator types

Except for the landscape-level, biotic indicators are most frequently used in all ecosystem types. This is likely owed to the fact that the ecosystem reviews partly referred to the role of biodiversity indicators, which is bioindication *per se*. Yet, there is also a wide field of development and application of bioindicators apart from the assessment of biodiversity. This is reflected by the analysis of the SCIE references reviewed for this study (Figure 2). Except for soils, species and community measures are frequently developed and widely applied in all ecosystems. Thus, the literature review revealed the effort that has been spent on the development of bioindicators during the past ten years. A reason might be the widely reported advantage of bioindicators, as they enable indication over space and time. Bioidincators are potentially applicable from the farm scale (e.g., single plant species or soil micro-organisms) to the global scale (e.g., the Farmland Bird Index, proportion of rare/threatened species). On the other hand, micro-organisms integrate over a relatively short time-span, while birds are intermediate and large mammals potentially indicate habitat and ecosystem quality over decades.

Bio-indicators span from single keystone or umbrella species to proportional measures at the community level. Functional measures, i.e. process-related characteristics, such as productivity, were especially abundant in forests, soils and rivers. Part of these functional measures belong to traits and were also frequently, but less abundantly, reported in SCIE references. Interestingly, there is a considerable evidence of traits as indicators of soil quality and biodiversity; nearly 22 % of soil indicators accounted for this biotic subgroup.

For this study, abiotic indicators were grouped into physico-chemical measures and landscape attributes, such as area and fragmentation. Although physical and chemical indicators are being frequently applied in all ecosystems, they seem to play a rather subordinate role compared to bioindicators. However, it has to be considered that this study relies on the SCIE, i.e. abiotic indicators do not seem to be subject to frequent development and improvement, which are most frequently reported in publications. Once set up and standardised, physical and chemical measures have been proven to be reliable and manageable indicators of environmental stress. Therefore, they are still being frequently and widely applied in agro-ecosystems, rivers and wetlands, irrespective of the scientific effort that has been dedicated to their further refinement during the past ten years.

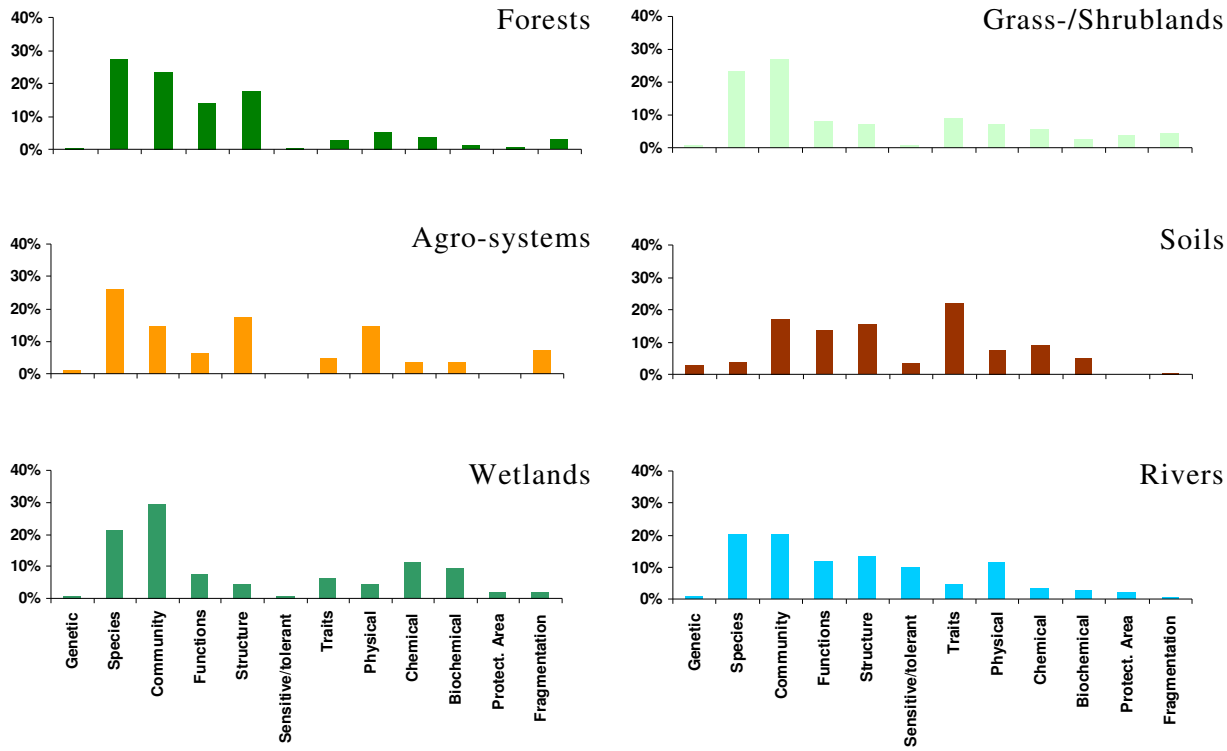

Figure 2: Classification of indicators into indicator types per ecosystem.

A considerable lack of indicator development is evident for genetic indicators. Except for very few examples reported for agro-systems and soils, this component of diversity has rarely been dealt with in the scientific literature. This is despite the assumption that quite a large portion of biodiversity is dependent on the degree of genetic variability. Whether there will be increased effort on this component of biodiversity depends on the success of communicating the value of genetic diversity to decision makers.

4.2 Indicators of biodiversity and ecosystem services

The number of indicators referring to biodiversity ranges between 16 % (rivers) and 32 % (agro-systems; see Figure 4). Their importance becomes similarly obvious by a general SCIE search for literature related to the indication of biodiversity (Figure 3): Between 100 and 1,000 hits were reported back by the SCIE for individual ecosystems, however, irrespective of whether they were really dealing with the development and application of novel indicators. According to our database, the latter was only considered in 15–59 references per ecosystem (mean: 35). Thus, biodiversity seems to be an important issue for the scientific community and has gained much interest during recent years. But

compared to the overall role of biodiversity, the development of new indicators seems to have gained considerably less effort.

Very few references directly address the indication of ecosystem services. The general SCIE search, moreover, revealed large differences between ecosystems: while only 13 references were reported back for grasslands, soil services were addressed by more than 250 references. With more than 100 references each, river and forest ecosystem research also dealt with ecosystem services (and their indication). But do the indicators and indication systems really refer to ecosystem services? The database contains 425 (out of 534 = 80 %) references that somehow could be linked to the indication of ecosystem services. Most indicators – indirectly or directly – refer to regulating and supporting services, such as erosion control, soil formation, nutrient cycling, self-purification of water, water retention and decomposition of organic material. Very few could be linked to provisioning services, such as food and fuel supply (< 6%). Direct links to ecosystem services and the Millennium Ecosystem Assessment were very sparse. This finding reflects that the issue of ecosystem services is comparatively recent (MA, 2003, 2005a).

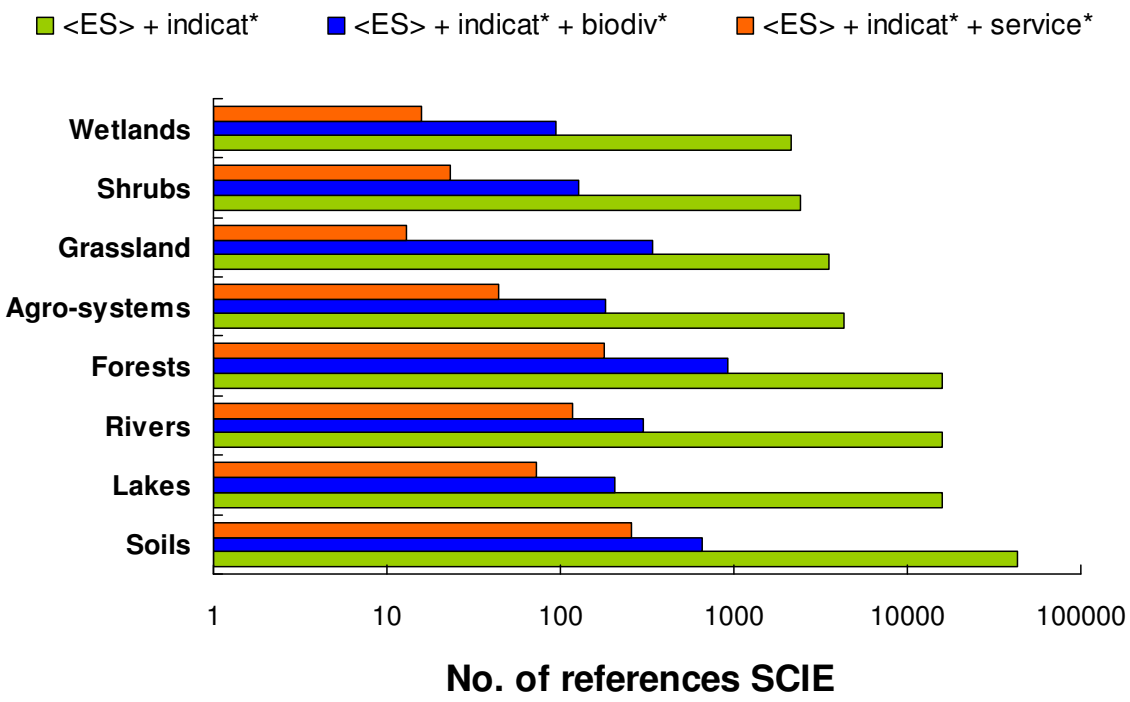

Figure 3: Number of references in the SCIE related to three keyword combinations (ES = respective ecosystem; time span for literature research: 1997–May 2007).

4.3 Purpose of indication

If ordered according to the purpose of indication, the SCIE references revealed some differences between ecosystems (Figure 4). A clear dominance of biodiversity indicators is evident for agro-ecosystems (32 % of references), but they also play a considerable role in all other ecosystems. Habitat quality assessment and trend monitoring are major purposes in all ecosystems, except in soils, where both categories account for less than 5 %, each. Habitat quality assessment clearly dominates indicator development in river ecosystems, which is likely owed to the increasing effort due the implementation of the European Water Framework Directice (WFD, for policies see also below). A similar pattern is evident for forest, grass-/shrubland and wetland ecosystems. Indicator development aims here to provide new measures for habitat and ecological quality assessment and monitoring, and biodiversity assessment. These ecosystems are also subject to some effort on the indication of ecosystem functioning (> 10 % of SCIE references, each), a field that does not seem to be sufficiently covered in soil and river ecosystems.

A lack of development is evident for predictive indicators (models) and those suited to indicate economic values. In the case of predictive models this reflects traditional assessment and monitoring systems which rely on on-site data to assess actual status and its change over time instead of modelling the changes. However, predictive models seem to be gaining more interest recently, which may be related to the increasing effort to indicate the ecological effects of Global Climate Change (GCC). Also subject to predictive modelling, GCC presumably stimulates the development of models to indicate and assess its impact, which is likely to be better approached by models as opposed to on-site sampling. Yet, frequent on-site sampling shall remain necessary to develop suitable, and calibrate reliable, models over time.

Altogether, only 11 out of 534 of the SCIE references (< 2 %) referred to monetary indicators or presented results of the application of monetary inventories. It has been shown in the previous chapter that economic indication has been rarely tackled by both SCIE references and additional references reviewed for this study. Facing the challenge of SEBI 2010 and the Millennium Ecosystem Assessment, ecosystem (service) valuation has become an integral component of ecosystem indication towards halting the loss of, and sustaining, biodiversity at the level required to maintain their service provision. Gren et al. (1995), for instance, calculated the value of the entire Danube floodplain, mainly with respect to its regulative function, i.e. nitrogen and phosphorous retention: the value was at least 650 million Euros per year. Comparable studies on ecosystem (service) valuation are still fairly sparse, but urgently needed to monitor ecosystem service values and communicate them to decision makers.

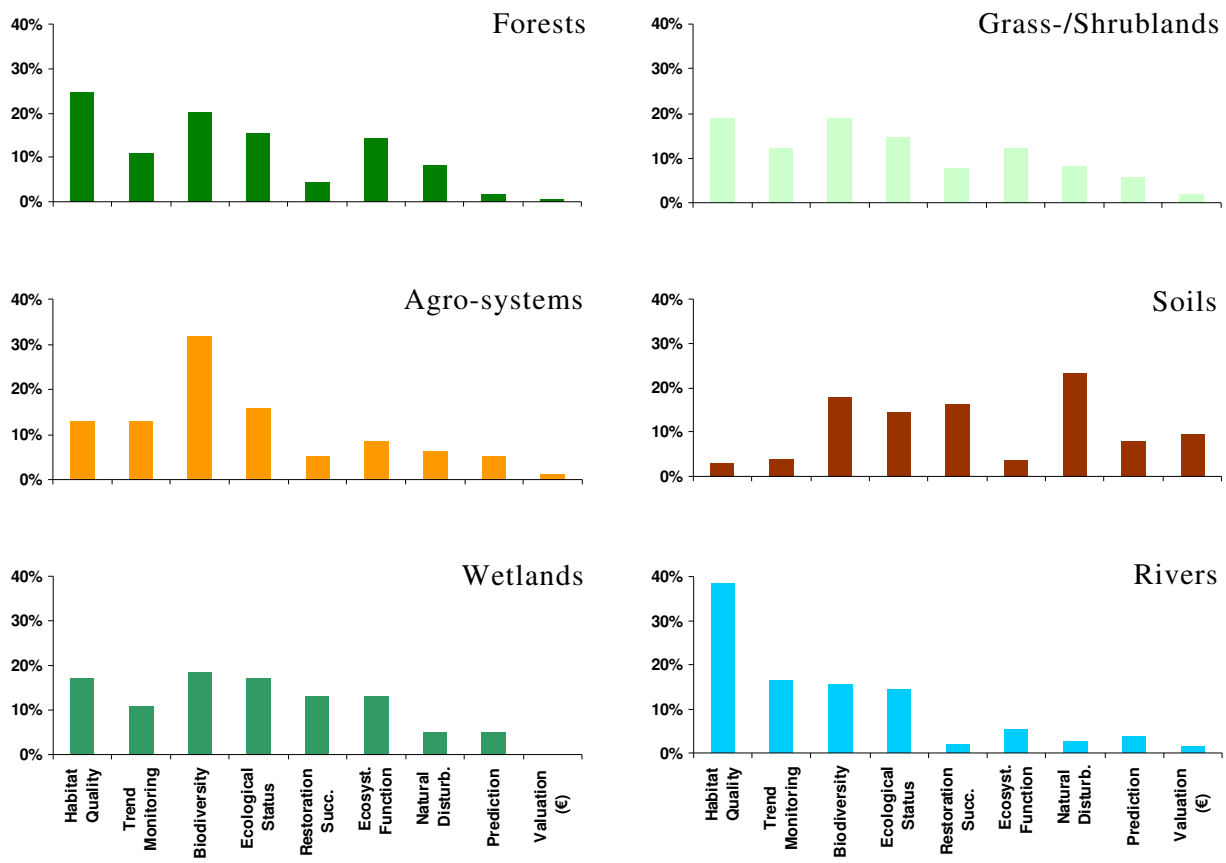

Figure 4: Allocation of indicators according to the major purpose of indicator development.

4.4 Spatial scales

The analysis of the indicator database revealed a clear dominance of indicator development at the local and regional scale in all ecosystems, except for soils (Figure 5). Indicators of the latter are strongly – and inherently – related to the smallest scales, i.e. to the farm and local scale. This is owed to the spatial limitation of the typical soil micro- and mesofauna. It seems to be evident, therefore, that indicator development during the past ten years was mainly focussed on applications within national limits. The slight shift of indicators of river ecosystems to larger spatial scales is due to the WFD, which stimulated the development of bioindicators applicable at slightly larger scales compared to the other ecosystems (Figure 5). From Figure 2, this seems to particularly apply to bioindication.

Indicators at the sub-global and global level have been rarely published. It is presumably not owed to a lack of availability of tools to develop indicators at large spatial scales, since suitable GIS and remote sensing applications have been developed long before. As has been shown in Chapter 3.7 on landscape indicators, much effort has been spent on the development of indicators at the landscape scale. However, it may be the case that bioindication *per se* is poorly applicable at very large spatial scales, at least in terms of genetic and functional components.

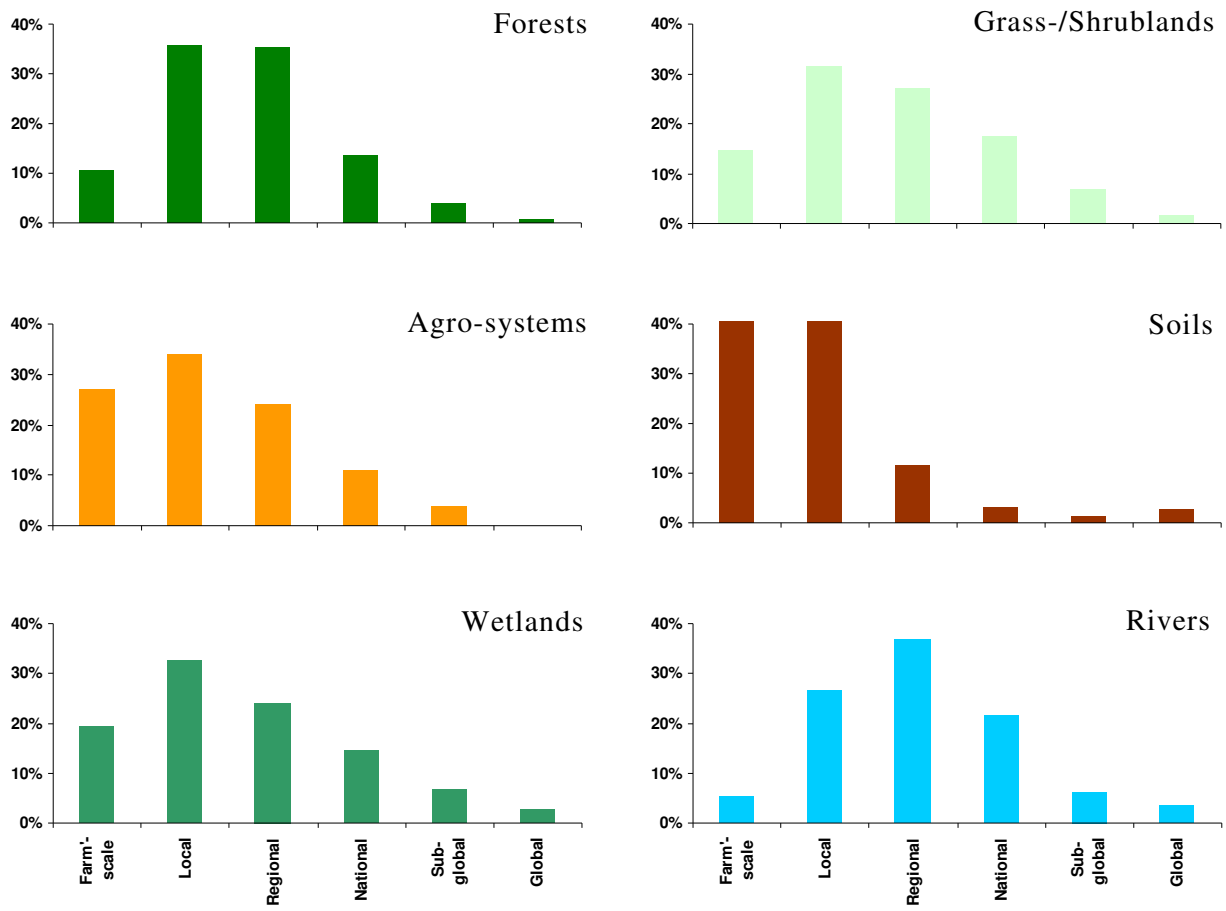

Figure 5: Allocation of indicators to main spatial scales of application. Multiple scales were possible for a single indicator.

4.5 Policies

Policies can be powerful drivers for indicator development (Figure 6). Many references were directly or indirectly linked to the Convention on Biological Diversity (CBD), which is, of course, also owed to their linkage to the indication of biodiversity. Particularly, indicator development and application in forest, grass-/shrubland and agro-ecosystem seems to be mainly driven by the CBD (73 %, 35 % and 41 % of indicators, respectively).

Being mandatory for 27 countries, the European Water Framework Directive (WFD) has clearly stimulated indicator development and indication of river and wetland ecosystems. Issued in 2000, the WFD required a focus on bioindication in line with a series of clearly defined criteria. Existing indicators in many European countries did not fulfil the new demands and, consequently, a huge effort on the development of novel indicators and indication systems was spent. A similar finding is evident for soils, where ecosystem indication has been strongly driven by the Soil Thematic Strategy (STS, see

also Chapter 3.4). Similar to the WFD, the STS clearly sets the framework for future *bio*indication in soil ecosystems and, hence, considerably stimulated the development of new indicators and indication systems.

Nature conservation policies, such as the EU Habitats Directive, Birds Directive and the NATURA 2000 network, stimulated indicator development in grasslands and shrublands, wetlands and agro-ecosystems. Between 40 and 50 % of references were linked to these policies in the three ecosystems.

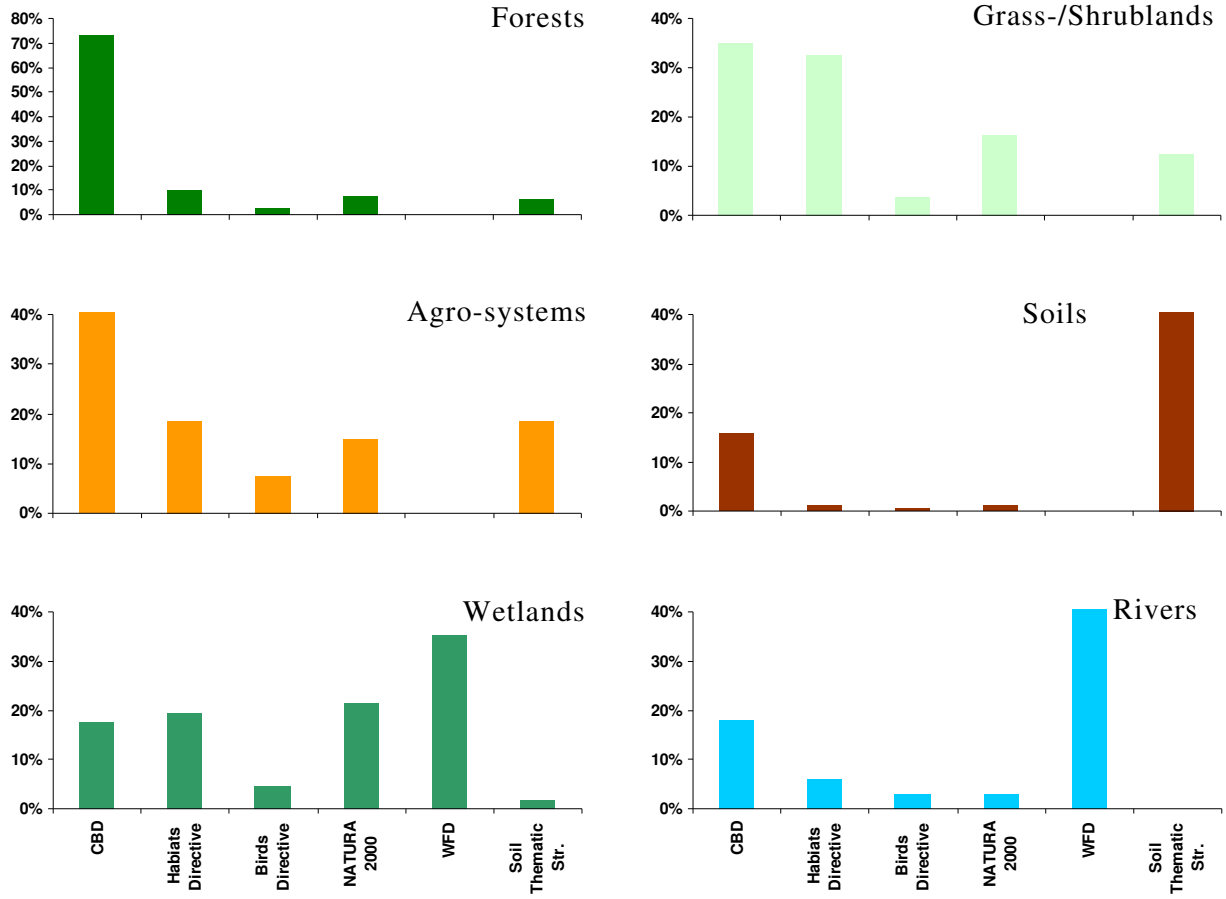

Figure 6: Allocation of indicators to selected international policies. Multiple assignments were possible for a single indicator.

4.6 Standardisation and validation?

This last question is crucial for ecosystem indication, as both standardisation and validation mark those indicators reliable and applicable over large (international) spatial scales. Standardisation may cover not only indicator calculation and application but also the methods and techniques to gather the data for calculation. Our data reveal, however, that the majority of references did not refer to

standardisation (Figure 7). Only between 20 and 40 % of SCIE references referred also to standardisation or clearly provided information on the status of standardisation of an indicator. Least effort was found for forests and rivers, where only 15 and 5 % of indicators, respectively, were subject to standardisation.

The generally low level of standardisation may be supported by the large number of references stimulated "simply" by science, i.e. indicators have been developed and applied solely for scientific purposes without any obvious relation to policies or target field of application. More than half the references (52 %) referred to this group.

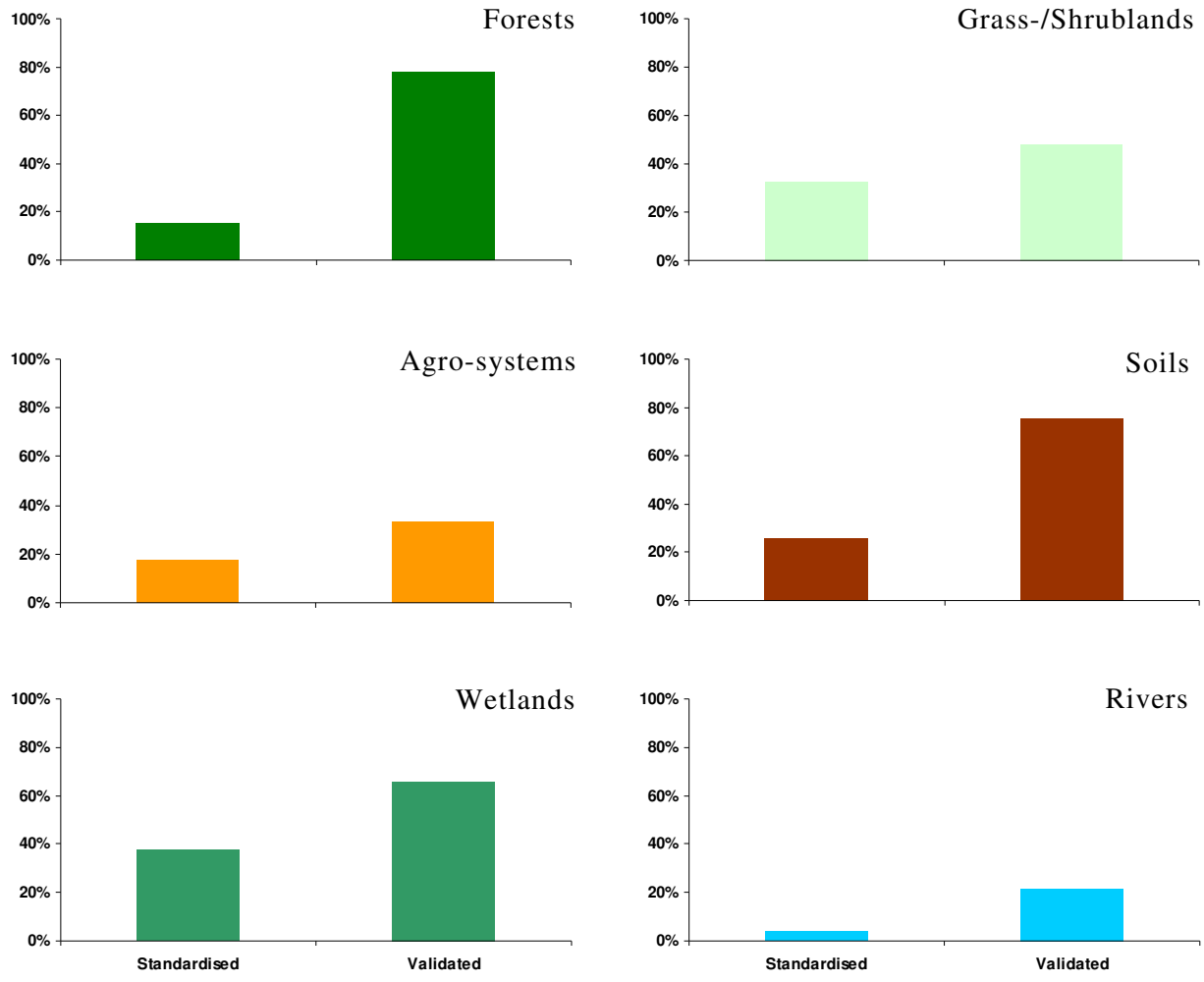

Figure 7: Proportion of indicators subject to standardisation and validation.

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Annex 1: Parameters used to classify indicators in the database.

Group	**Characteristic**
Numerical type of measurement	Direct measure
	Proportional measure
	Index
	Multimetric Index
Purpose of indication	Biodiversity evaluation
	Water quality assessment
	Ecological status assessment
	Restoration (success) assessment
	Ecosystem function (assessment)
	Response to natural environmental stress (e.g., fire, climate)
	Prediction
	(Monetary) valuation
Indicator type *biotic*	Genes
	Species
	Community (acc. to Noss, 1990)
	Functions (acc. to Noss, 1990)
	Structure (acc. to Noss, 1990)
	Relation sensitive/tolerant taxa
	Traits
abiotic	Physical
	Chemical
	Biochemical
'landscape'	(Protected) Area
	Fragmentation
Driving force behind indication	Policy
	National initiatives
	Science
	Global Climate Change
Policy most relevant for indication	Convention on Biological Diversity (CBD)
	EU Habitats Directive
	EU Birds Directive
	NATURA 2000 Network
	EU Water Framework Directive (WFD)
	Soil Thematic Strategy
	National Nature Protection Policy
	National Water Quality Monitoring

Group	**Characteristic**
	WSSD
	OECD
Spatial scale of indication	"Farm" scale
	Local
	Regional
	National
	Sub-global
	Global
Relation to ecosystem service (MA, 2005a) *provisioning*	Food
	Fibre (fuel)
	Genetic resources
	Power supply
	Irrigation
	Ornamental resources
	Fresh water
	Habitat
	Biodiversity
regulating	Water retention
	Self-purification
	Micro-climate
	Air quality
	Erosion
	Wastes
	Diseases
	Pests
	Pollination
	Natural hazards
	Wild fire
supporting	Water cycling
	Nutrient cycling
	Decomposition
	Soil formation
	Photosynthesis
	Primary production
Indicator standardised	Yes/No
Indicator validated	Yes/No

Annex 2: References of indicators compiled in the database.

No.	Full reference:
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