DEPARTMENT OF ECONOMIC AND SOCIAL AFFAIRS
STATISTICS DIVISION
UNITED NATIONS

System of Environmental Economic Accounting

SEEA EEA Revision

Expert Consultation

Working group 2: Ecosystem condition

Review copy of:
Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables

Consultation date: 13 March 2019

Recommended citation:
Czúcz, B., Keith, H., Jackson, B., Maes, J., Driver, A., Nicholson, E., Bland, L. (2019) Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables. Paper submitted to the SEEA EEA Technical Committee as input to the revision of the technical recommendations in support of the System on Environmental-Economic Accounting. Version of 13 March 2019. 23 pp.

Research area #2: Ecosystem condition

Discussion paper 2.3: Proposed typology of condition variables for ecosystem accounting and criteria for selection of condition variables

Lead author: Bálint Czúcz

Contributing authors: Heather Keith, Bethanna Jackson, Joachim Maes, Amanda Driver, Emily Nicholson, Lucie Bland

Contents

The SEEA EEA revision process

Ecosystem condition is defined in the SEEA EEA as the overall quality of an ecosystem asset in terms of its characteristics (United Nations, 2012).

How do we measure and report on the condition of ecosystems in an ecosystem accounting framework? Addressing this question means establishing a common definition of ecosystem condition, selecting suitable indicators of condition, evaluating the actual condition of an ecosystem against a reference level, and providing an overall, comparable condition score for reporting or accounting. It also requires a further understanding of the relationship between the ecosystem condition, biodiversity and the delivery of ecosystem services as well as knowledge about the pressures (or in a broader sense the drivers of change) that continue to impact ecosystems.

The SEEA EEA Technical Recommendations (United Nations 2017) do not yet provide definitive advice on how to address these several challenges when reporting ecosystem condition in condition accounts. These challenges have been addressed in a Revision Issues Note for the Ecosystem Accounting Revision 2020 (United Nations, 2018) which recommends providing further guidance on ecosystem condition.

This paper is part of a series of discussion papers on ecosystem condition. It aims to provide **a conceptual basis for the selection of variables and indicators that can be used to describe the most relevant ecosystem characteristics with a view to assess ecosystem condition.** Two other papers are part of this series: a paper on the purpose and structure of ecosystem condition accounts (Discussion Paper 2.1) and a paper reviewing existing approaches for ecosystem condition accounting (Discussion Paper 2.2).

These discussion papers have been developed by a working group established as part of the revision process. The working group on ecosystem condition is one of five working groups for the four research areas (RAs) identified in the Revision Issues Note: RA1 focuses on spatial units, RA2 on ecosystem condition, RA3 on ecosystem services and RA4 on valuation.

1. Introduction

The SEEA EEA defines ecosystem condition as the overall quality of an ecosystem asset in terms of its characteristics (United Nations et al., 2012). The origins and purposes and of the ecosystem condition concept, its role in the SEEA EEA framework, and the basic structure of ecosystem condition accounts is discussed in detail in DP 2.1. This paper builds on the fundamentals laid down in DP 2.1, and adds further details to several key elements necessary for any practical implementations:

- the *characteristics* of the ecosystems (which ones are relevant for describing ecosystem condition); and
- the concrete quantitative metrics (*variables*, *indicators*, and *indices*) which can be used to describe these characteristics

In the context of this discussion paper, variables, indicators, and indices are all considered as quantitative measures (numbers) that characterize the studied system (i.e. specific ecosystem units) from a certain perspective. The most general term is *variable*: any quantitative measure reflecting a phenomenon of interest can be seen as a variable. *Indicators* are variables with a normative interpretation associated (e.g. a reference level, or any other way of distinguishing "good" from "bad"), with a view to informing policy and decisions. An *index* is a (thematically) aggregated indicator, which represents relatively broad aspects of the studied system in a single number.

Ecosystems have many quantifiable characteristics but not everything can be measured. A first step in exploring the inherent multidimensionality of ecosystems is to understand how many relevant characteristics of ecosystem condition can be determined and what should be the criteria to decide on the relevance of characteristics. Then for the most relevant characteristics indicators can be selected. The main objective of this discussion paper is to provide (1) a detailed and consistent set of *criteria* for selecting relevant ecosystem characteristics and condition indicators, and (2) a proposal for an indicator *typology*.

2. Criteria for ecosystem condition indicators

The identification an adequate set of indicators can be a complex and challenging process. Indicators need to be scientifically credible, responsive to user needs (salience), and be perceived as such by their end users (legitimacy, Cash et al., 2003). From a policy perspective, the success of an indicator resides in its utility for policy actors, influence on policy processes, and impact on policy outcomes (Bauler, 2012). There are many papers in the scientific literature that aim at supporting and standardizing the process of identifying and selecting indicators with criteria and recommendations (e.g. Dale and Beyeler, 2001; Niemeijer & de Groot, 2008; Giannetti, 2009; Kandziora et al., 2013).

There are several criteria that potential indicators must meet in order to characterize the condition of ecosystems in a way that complies with accounting principles, is policy-relevant, and is also meaningful from a biophysical perspective. This chapter gives a comprehensive overview on all potential selection criteria that need to be observed in order to create an adequate set of ecosystem condition indicators in a SEEA EEA context (Table 1). Following Niemeijer & de Groot (2008) we distinguish two types of selection criteria: *individual criteria*, which can be used to appraise the relevance or usefulness of each ecosystem characteristic or indicator proposed; and *ensemble criteria*, which need to be applied to the whole set of candidate indicators (e.g. to ensure that there are no gaps or double counting).

SEEA EEA 2012 (e.g. §4.60, 4.66) clearly distinguishes (1) ecosystem characteristics (i.e. major groups of system properties or components based on ecological understanding), and (2) the indicators which are used to quantify them. Characteristics can be broad and abstract (for example, SEEA EEA lists water, timber, carbon and biodiversity as characteristics relevant for "basic resource accounts"), whereas indicators should be concrete and specific as much as possible. Characteristics and indicators can be seen as two hierarchical levels of structuring and organizing condition information. The selection procedure should address both stages. Adapting the recommendations of Niemeijer and de Groot

(2008), the general procedure for the selection of ecosystem condition indicators should include the following three steps:

- Defining the scope and the purpose of the study (ecosystem, accounting goals...),
- Identifying key characteristics (ecosystem components and processes),
- Selecting the best indicators for the selected characteristics.

Table 1. Selection criteria for ecosystem characteristics, variables and indicators

Criterion	Short description
Individual criteria (meaningful for the individual variables and their derived indicators)	
Relevance	ecosystem characteristics (and their indicators) should be relevant in terms of the fundamental purpose (intrinsic or instrumental) that can be linked to ecosystem condition accounts (see DP 2.1)
State orientation	ecosystem characteristics and indicators should describe the state of the studied (ecological or socio-ecological) system
Framework conformity	ecosystem characteristics and indicators should be differentiated from other components of the SEEA EEA framework
Spatial consistency	ecosystem condition indicators should be linked to a specific location (mapped) or spatially referenced
Temporal consistency	ecosystem condition indicators should be linked to a specific time period and be sensitive to change
Feasibility	ecosystem condition indicators should (potentially) be covered by data sources over large areas
Quantitativeness	ecosystem condition indicators should be measured at a well-defined quantitative scale that allows comparisons in space and time
Reliability	primary (measured) data should be preferred over derived data which, in turn, should be preferred over modelled data.
Normativity	ecosystem condition indicators should preferably have an inherent 'normative' interpretation
Simplicity	ecosystem condition indicators should be as simple as possible
Ensemble criteria (which can only be interpreted for the whole set of indicators)	
Parsimony (or complementarity)	the final set of ecosystem condition indicators should cover as much information on the studied ecosystem as possible with as few indicators as possible
Data gaps	ecosystem characteristics which seem to be relevant, but which are not covered adequately by available data sources should be highlighted as data gaps

2.1. Criteria for individual indicators

The criterion of **relevance** implies that the selected condition indicators should address those characteristics of the ecosystems that are most relevant from the perspective of the *fundamental purpose* underlying the condition accounts. With an instrumental perspective (see also DP 2.1) *those characteristics should be selected, which exert the most influence on the capacity of the ecosystems for*

providing multiple ES.[1] On the other hand, from an intrinsic perspective a good scientific understanding on what constitutes ecological integrity can also be used as a starting point to determine which characteristics need to be considered relevant.[2]

The relatively abstract criterion of '**state orientation**' requires that ecosystem condition indicators should describe the state of the studied ecosystem as much as possible.[3] Most ecosystem characteristics don't have a single 'default' formulation (definition/quantification approach)[4]. If there is a choice between two alternative ways (indicators) for quantifying a relevant system characteristic, the one which conforms more to the idea of 'state' is the one which should be preferred.

The criterion of **framework conformity** is of key importance for the integrity of the whole assessment. According to this criterion, each aspect of the studied system should only be described under a single component of the SEEA EEA conceptual framework. Characteristics that can be better considered under *ecosystem extent* (e.g. forested area or deforestation) or *ecosystem service* (e.g. carbon sequestration) should be handled there and only there. If such characteristics are re-considered as condition indicators, that might lead to double counting and user confusion, and thus can discredit the whole assessment.

The criterion of **spatial consistency** means that the indicators need to be linked to a specific location (mapped). More specifically, all candidate variables have to be interpretable over *any area* that is (1) larger than a predefined minimum area ('*spatial grain*'), and (2) is covered by one of the ecosystem types for which the variable makes sense ('*thematic domain*'). The grain and the thematic domain of each variable should be included in their definition. Ideally, the whole accounting should have a

[1] The criterion of relevance could be established by well-designed systematic reviews exploring the relationships between ecosystem characteristics and ecosystem services (e.g. Verhagen et al., 2016; Czúcz et al., 2017; Schwartz et al., 2017; Smith et al., 2017). There is a high amount of primary research studies that test the relationship between a particular ES and a particular ecosystem characteristic in a specific context (ecosystem type), but general syntheses are still largely missing.

[2] As long as a widely applicable and accepted general synthesis of ES~EC relationships is not available (see previous footnote), a broadly agreed typology of ecosystem condition indicators reflecting long standing ecological knowledge can be used as a surrogate. To this end the typology has to be considered an indicator template (i.e. as a shortlist of characteristics which are considered relevant, and thus have to be included into the condition account, see also Chapter 3).

[3] The DPSIR framework distinguishes pressures (influences, inputs) and responses (consequences, outputs) from state descriptors. In system science, state is considered to be a set of variables that describe enough about the system to determine its future behaviour in the absence of any external forces (Palm et al., 2005). Furthermore, a system analysis perspective, can also help to justify/interpret the criteria of relevance & parsimony, as the set of 'state' indicators of a system are expected to "describe enough about the system to determine its future behaviour" (Palm et al., 2005). As in the context of ES accounting / assessments the key output ("future behaviour") of the studied system is the portfolio of ES generated, it is important that the 'state variables' describing the system would capture everything that can influence this portfolio.

[4] Most of the (eco)system characteristics can be measured/quantified in several potential ways. There can be many potential reasons for this: e.g. the characteristic can be abstract & ambiguous (e.g. biodiversity, use intensity), or difficult to measure (e.g. NPP, grazing intensity) or just too highly fluctuating (water availability, vegetation cover) which needs to be 'averaged' somehow (and there are multiple options...) The arising options can lead to different quantitative measures (=indicators) which just indicate the same ecosystem characteristic.

harmonized spatial grain and the definition of the variables should respect this grain. For example, a large grassland is handled as an ecosystem type, whereas a small ('sub-grain') grassland is considered as a part of the embedding ecosystem type (e.g. cropland) with the 'density' of such embedded fragments considered as a condition attribute of the embedding ecosystem type).

Temporal consistency implies that indicators are sensitive to change and linked to a specific time period (*temporal grain*), which should be regularly in SNA / SEEA EEA, e.g. every year or every 5 years. Biophysical considerations also suggest that the grain should cover at least one full annual cycle which is the key periodical cycle for the studied system. Sensitivity to change should also be considered (with respect to this temporal grain), so that condition would be reasonably variable across a few time steps (i.e. *quasi-constant* or *extremely variable* candidate variables should be excluded or reformulated). This means that for data streams with relatively fine temporal resolution (e.g. remote sensing data) the precise definition of the condition indicator should involve some sort of 'temporal aggregation', e.g. in the form of appropriate statistical aggregation functions (central tendencies or extremities, e.g. mean annual values, annual maxima, etc.). Defining the timeframe and time resolution is important for selecting indicators: for instance, to estimate change over 50 years with data every 5 years, the most useful variable may be different than a variable which measures intra-seasonal variations of an ecosystem characteristic.

Feasibility means that indicators should be covered by (potentially) available data sources over large areas. This implies that those characteristics that are difficult to measure or are in any way unfeasible to be covered by data in the foreseeable future should be avoided.

According to the criterion of **quantitativeness** indicators should be measured at a well-defined quantitative scale, which allows for meaningful comparisons and change detection. Indicators should ideally be measured at a ratio/interval scale or at least at an ordinal scale (sensu Stevens, 1946). Attributes measured at a categorical/nominal scale should preferably not be used as condition variables, unless they can be reformulated to an ordinal/interval/ratio scale (e.g. by using scores/weights, or by being quantified as the 'share' of a relevant subtype over a larger area).

The concept of **reliability** is linked to the uncertainties concerning the indicators. Suitable indicators should rely on data that are measured in an objective and standardized way. Primary (measured) data should be preferred to modelled/derived data, which always rely on a number of assumptions, contain inherent errors, and are thus more prone to being criticized or disputed.[5] Modelled data can change even retrospectively if the modelling technique is updated. In the formulation of the indicators, subjective elements should be avoided as much as possible, and if unavoidable (e.g. scores used for weighting 'components' of a composite indicator) they should rely on a broad consensus of 'experts' in a clearly documented way. Data streams should ideally be resistant to malicious tampering/manipulation.

[5] This does not involve primary data cleaning/harmonization/transformation operations, such as spatial/temporal interpolation/aggregation, or quality control (noise reduction) techniques. Such preprocessed data can be more accurate than non-transformed ones.

Indicators should by definition have a '**normative**' interpretation, i.e. they should be able to distinguish what is 'good' or desirable from what is 'bad' or undesirable -- preferably with general consensus. This distinguishes indicators from simple variables, which don't have an agreed normative interpretation associated to them (Heink & Kowarik., 2010)[6]. Assigning agreed reference values to variables is a frequently used technique to make them indicators, and any deviation from the reference value is seen as undesirable. Ideally, desirable and undesirable should lie at the opposite ends of the scale of an indicator, so that there is a monotonous 'quasi-linear' relationship between the variable and the underlying human value judgement (i.e. an increase in the indicator value should always mean a better condition, and the same increase in the indicator value should always mean approximately the same degree of 'improvement' in the condition -- at all parts of the scale). The criterion of normativity also implies that indicators should be selected / constructed in a way that approximates this ideal situation as much as possible[7] (as this can allow easier policy interpretations, and more straightforward aggregation procedures).

Finally, good indicators should be as **simple** as possible (but not any simpler). If every other criterion is ensured, simpler indicators (allowing easier policy interpretation and more powerful messages) should be preferred over more complex/abstract metrics.

2.2. Ensemble criteria

The remaining two criteria do not focus so much on the characteristics of the individual indicators, but on the whole indicator set. **Parsimony** or complementarity means that there should be as few indicators as possible so that they would cover as much information as possible (without any unnecessary redundancies). To this end the selected indicators should be independent/non-correlated as much as possible, but the set should still represent all major 'aspects' of the studied system (so that they could be rightfully considered as 'key indicators' of system state). Many times there can be correlations between seemingly unrelated variables describing different system aspects (and listed under different ECI headings, see Chapter 3), which are generated by the internal mechanisms of the studied system. Correlations can also be introduced by technical artefacts (e.g. by inconsistent reuse/re-labelling of data streams). In case of 'correlation conflicts', the most appropriate (relevant, simple, reliable, framework conform...) candidate should be chosen following the criteria for individual indicators. All kinds of correlations in the final indicator set imply a high risk of confusion and (particularly the technical artefacts) can lead to a loss of credibility.

Finally, the set of indicators should be checked for comprehensivity, and all system characteristics which are known to be relevant, but are not covered adequately by available data sources should be

[6] In this sense variables are 'neutral', and 'neutrality' can also be defined as the lack/opposite of normativity. Neutrality (in this sense) can be a useful property of variables in some applications, but in the frame of condition accounts normative variables (i.e. indicators) seem to be more useful than neutral variables.

[7] Nevertheless, other criteria (e.g. simplicity) can be in conflict with this one, and even some of the most widely used indicators (e.g. human body temperature as a health indicator) may not have a single linear normative interpretation (i.e. an increase in the temperatures is bad above 36 °C, but it is good at lower parts of the scale: fever vs. hypothermia).

highlighted as **data gaps**. The identification of such information gaps can help in guiding future research activities.

2.3. Implications of the selection criteria for pressures, protection and land management

The conceptual criteria of state orientation and framework conformity are also of key importance for SEEA EEA, which can offer valuable guidance in open questions related to particular types of variables.

Pressures are often considered as an "indirect approach" for measuring ecosystem condition (e.g. Erhard et al., 2016, p.31). If there are little data available on state, then pressures can be considered a useful surrogate, as long as the relationship between the two is well understood and justified (Bland et al. 2018). This is clearly a compromise, as conflating pressures with state variables can compromise the credibility and salience of the resulting accounting tables. Nevertheless, this does not necessarily mean that accounting tables should be blind to the policy issues highlighted by the most relevant pressures. In the case of most pressures (*erosion*, *pollution*, *invasion…*) there is an underlying 'hidden' variable, that reflects the 'degradation' of the ecosystem with respect to that specific pressure. This underlying variable is an environmental 'stock' (e.g. the thickness of soil layer, the concentration(s) of pollutants, or the abundance of invasive species) that is gradually degraded (depleted, accumulated...) by the pressure. Typically, such stocks can meet all the criteria, so they can be more appropriate for condition accounting than their change or the connected flows (degradation / depletion rates, fluxes, flows, or other indicators of flow intensity). Using these 'degradable stocks' as condition indicators comes with multiple further advantages: they can be used to formulate very clear and pertinent policy messages on ecosystem degradation (as a change in these environmental stocks); and the degree of policy attention highlights those 'degradable stocks' that are perceived as the most valuable or most endangered.

Focussing at 'degradable stocks' in the condition accounts will allow significant progress in reporting on changes in the quality of ecosystem assets that is not available currently when change in area of an ecosystem asset is reported under ecosystem extent. An example of the benefit of quantifying and reporting condition as well as extent is the carbon accounting under the UNFCCC, where change in carbon stocks are reported if land use change occurs, that is change in ecosystem extent, but are not reported if degradation of stocks occur within a land use type, that is a change in ecosystem condition (IPCC, 2014). Treating degradable stocks in a condition account is particularly relevant when ecosystem extent is often measured by remote sensing, which will detect a stock loss due to change in ecosystem type, e.g. clearing vegetation, but may not detect a stock loss due to degradation e.g. loss of understorey or weed invasion.

A further important type of pressures worth considering is *overexploitation*, which can frequently, but not necessarily, also be linked to degradable stocks.[8] Most of the ecosystem types have a specific target ecosystem service (typically a provisioning service), and 'traditional' ecosystem **management**[9] aims at

[8] The underlying degradable stock is quite straightforward for e.g. forest management (timber stocks), and marine fishing (fish stocks), but it can also be non-trivial, e.g. for agricultural intensification or overgrazing.

[9] By management we mean 'natural resource management' aimed at optimizing ES service provision. Conservation activities, which try to maximize biodiversity, are discussed in the next paragraph.

the maximization of that service (de Groot et al., 2010). The intensity of these management activities has shown to exert very strong influences the supply of a broad range of services, well beyond the original 'target ES' of the management activities (Santos-Martin et al., 2019). No wonder that some of the case studies reviewed in DP 2.2 also apply indicators of management intensity (case studies 20, 21). Nevertheless, the inclusion of management intensity indicators into condition accounts is still a controversial option for SEEA EEA.[10]

Some pressures should probably *not* be considered in the ecosystem condition accounts, even if underlying environmental stocks can be identified. This includes pressures (or drivers) with rather indirect influence on ecosystems (e.g. *climate change*, *human population changes*), which should probably been considered external to the studied ecosystems.[11] *Habitat loss* is a direct pressure with a clearly identifiable degradable stock (the area of the ecosystem/habitat type in question), which should probably be omitted from condition accounts for framework conformity reasons (it should be addressed as ecosystem extent rather than ecosystem condition).[12]

Similarly to pressures, **protection status** (e.g. the location, area, or representativeness of protected areas) is also frequently proposed as a proxy for condition if no other information is available (e.g. Maes et al., 2016; see further examples in DP2.2 case studies 10, 20, 21, 22). Protection could also be thought of as a rough proxy for reduced pressures, especially for reduced overexploitation (lower management intensities).[13] Nevertheless, indicators describing policy interventions performed in response to degradation processes don't make good condition indicators. There is no inherent relationship between protection status and other indicators of ecosystem condition, for example, an ecosystem could fall within a protected area and nevertheless be in poor condition / intensively modified (for example if it is the site of a dam or lodge, or if the management of the protected area is ineffective). In order to avoid

[10] If we consider the 'default' (main/typical) management of an ecosystem type as an integral part of the studied ecosystem), then it can also be seen as an 'internal' process of the studied (socio-ecological) system. It can be argued that such internal processes can be characterized with state variables (e.g. the intensity of the default management), which can then be added to the condition account (just like a natural disturbance regime). Some ecosystem types, e.g. urban and agricultural ecosystems, are even uninterpretable without human management. If not considered in the context of condition accounts, management can be seen as important type of ancillary data, highly relevant for ES capacity modelling (see Fig 2.2 in DP 2.1).

[11] Given also that climate is already very well covered with indicators and policy attention in other policy domains, it could be proposed that climate is not addressed directly by SEEA EEA condition indicators.

[12] For 'habitats' that are distinguished as ecosystem types this type of stock and its 'degradation' should be discussed under the 'ecosystem extent' accounts. On the other hand, if a habitat change is 'internal' to a specific ecosystem type (e.g. soil sealing in the case of urban ecosystems), then it can be added to the condition account (preferably with an indicator describing the underlying degradable stock; e.g. the share of impervious surfaces for soil sealing). This indicator will then be specific to the given ecosystem type.

[13] This means that, in principle and if data availability is not an issue, direct measures of pressures should be preferred to protected areas as indicators. So, a simple rule of thumb reflecting the "direct/proxy" nature of the different types of variables discussed in this chapter could sound like: "For any pressure (1) first try to use the underlying environmental stock as an indicator. If there is no data then (2) you can try using "pressure" data (loss rates, management intensity, etc.) as a proxy. And if that fails too, then you can try to use protected areas, as a loose proxy (of last resource)."

confusion and double counting, indicators describing policy response categories should be avoided. Including such indicators into the SEEA EEA ecosystem condition accounts would, among other issues, compromise their applicability in measuring the impact/efficiency of policy changes (e.g. the efficiency of a newly designated protected area).

3. A typology for ecosystem condition

A *typology* or *classification* is the operation of distributing objects into classes or groups that are less numerous than the original objects. This operation is very broadly and frequently used in science, as it can create an order among the "chaotic and muddled multiplicities" of life and thus can reduce the complexity of the problems (Parrochia, 2019). Classifications are therefore the essence of accounting systems. Classifications need to be ***exhaustive*** and ***mutually exclusive***: classes should not overlap, and their union should restore the divided concept. As each division (class) can be further subdivided, classifications can also be hierarchical.

An *ecosystem condition typology* is a hierarchical classification for ecosystem condition indicators. The main reasons why SEEA EEA should propose a typology for ecosystem condition is that it can create a meaningful order among ecosystem condition indicators. This order can have multiple advantages:

- it can help to establish a common language and a shared understanding;
- it can make different studies (assessments, countries, etc.) more comparable;
- it can be used as a structure for aggregation; and
- it can be used as a template for indicator selection.

As also emphasized by SEEA EEA, different ecosystem types have different relevant characteristics, which should be described by different indicators (see e.g. SEEA EEA 2012: §A.5). Nevertheless, in order to facilitate communication, as well as comparisons and aggregation across ecosystem types, an ecosystem condition typology should be ***universal*** at least at the top level (i.e. it is expected to be relevant for all major ecosystem types). On the other hand, the typology also needs to be able to host ecosystem-specific indicators at the lower levels.

3.1. Classification systems for ecosystem condition

Related to the concept of ecosystem condition, there are already several classifications in the scientific literature. Many of the ecological concepts discussed in DP 2.1 (e.g. ecosystem integrity, ecosystem health, naturalness…) also come with a typology, created either on a theoretical or a practical basis (e.g. for use with real life data / indicators). Such typologies include:

- the classification of essential biodiversity variables (EBV), as outlined by Pereira et al. (2013);
- the ecosystem integrity typology proposed by Müller et al. (2005);
- the BESAFE/OpenNESS typology the characteristics of 'natural capital' (Smith et al., 2017);
- the MAES typology for the mapping and assessment of ecosystem condition in the EU (Maes et al., 2018).

These proposals have a lot in common, but there are also differences (see Annex 1). The following section aims to deliver a common denominator of these classification systems by placing them through the filter of the abovementioned criteria.

3.2. Proposal for a general typology of ecosystem condition indicators

We propose a simple hierarchical typology of ecosystem condition indicators (ECI) for ecosystem accounting (Table 2). The structure of the proposed classes reflects a combination of long-standing ecological tradition (composition, structure and function, cf. Noss, 1990), theoretical considerations as discussed in the previous chapters, and practical considerations from DP2.2. Nevertheless, composition, structure, and particularly function are extremely broad concepts, interpreted in different ways by the different researcher communities. To avoid ambiguities, and to ensure the *mutual exclusivity* of the classes, we also propose a detailed interpretation for each class, with a detailed discussion on boundary cases[14] (what should be included and what should not) based on the criteria discussed in the previous chapter. The proposed ECI classes are also linked to EBVs and the case studies (from DP 2.2) in Table 2, and in more detail in Annexes 1 and 2.

Table 2. Proposal for a typology of ecosystem condition indicators (ECI) for ecosystem accounting

ECI class and subclasses	link to EBV classes	Link to case studies in DP2.2*
I. Species-based indicators (compositional characteristics) ● birds ● trees ● fish ● *...other relevant species groups*	B1-2, D1 (E3)	2, 5, 6, 8, 10-14, 16, 17, 19, 20-22
II. Vegetation and biomass (structural characteristics) ● tree cover (density / biomass) ● shrub cover ● litter ● pelagic (chlorophyll, phytoplankton etc) ● *...other relevant vegetation layers*	E1 (partly)	3, 4, 10, 12-14, 16
III. Ecosystem processes (functional characteristics) ● disturbance intensity (fire, flood…) ● *… other relevant ecosystem processes*	F4, (F1-3, D2)	2, 10, 12, 16, 20
IV. Physical and chemical state (abiotic characteristics) ● air ● soil ● water ● *...other relevant (abiotic) ecosystem compartments*	--	1, 2, 5, 8, 9, 11, 12, 14, 16, 20-22
V. Landscape pattern (landscape-level characteristics)	--	2-5, 7-10, 13, 16, 21

[14] This discussion is mainly done in footnotes, not to break the line of thoughts in the main text. Any suggestions for a better structure and further examples for overlaps and borderline cases are welcome!

• landscape diversity (overall) • connectivity/fragmentation (specific to an ecosystem type)		

* case study numbers as listed in Annex 1 of DP 2.2

The EIC class **species-based indicators**[15] (I) comprises a broad range of 'typical' biodiversity indicators, describing the composition of ecological communities from a biodiversity perspective. This includes the indicators based on the presence/abundance[16] of a species or species group, or the diversity of specific species groups[17] at a given location and time.

The class **vegetation and biomass** (II) contains indicators describing the local amount of living and dead plant matter (vegetation, biomass) in an ecosystem.[18] This class includes all metrics of vegetation density and cover, either related to the whole ecosystem, or just specific compartments (aboveground, belowground, litter...). For marine and freshwater ecosystems this class can include chlorophyll concentrations, phytoplankton availability, or plant biomass (e.g., seagrasses).

The class **ecosystem processes** (III) can host simple summary statistics (e.g. frequency, intensity) of the most relevant ecosystem functions[19], which meet the selection criteria discussed in the previous chapter, and which are not already covered by other indicators (e.g. natural disturbance regimes[20]).

[15] The term "species-based" does not intend to exclude important taxonomic groups with a high number of unknown species (e.g. bacteria, fungi). The diversity of such groups (e.g. estimated with metagenomic techniques) definitely belongs to this ECI class. There are many possible options to name this ECI class, including s that can options

[16] From a location-based perspective (required by spatial consistency) the distribution of a species also boils down-to local presence/absence. Furthermore, presence/absence is just abundance on a binary scale.

[17] This can include small groups of closely related species, large taxonomic groups (birds, butterflies), or non-taxonomic guilds (trees, macrozoobenthos...). However, indicators based on highly specialist functional groups, where even data collection was performed from a functional perspective (e.g. decomposers, N-fixers, etc.) should be considered under functional characteristics, or in other components of the SEEA EEA framework (e.g. as ecosystem service indicators).

[18] There is some overlap between species-based and vegetation indicators for foundation-species-based ecosystems such as mangrove, or where species groups and vegetation compartments coincide (trees on savanna, lichens on mountain rocks). Such cases should probably best be registered in this class (II).

[19] Ecosystem functions is a hugely diverse umbrella concept, which is used in highly different ways by the various research communities. Many of the characteristics that can be seen as "ecosystem functions" can also be seen as community characteristics (I), plant biomass (II), abiotic state descriptors (IV), or even as ecosystem service indicators (ES accounts). It might be a good practice to avoid including functional characteristics into this class whenever they can find a better home in another class.

[20] From the four EBV classes (F1: net primary productivity, F2: secondary productivity, F3: nutrient retention, and F4: disturbance regime) listed under the heading ecosystem function by Pereira et al., 2013, it is only F4 that cannot be connected to any other parts of the typology (as F1 is broadly covered by / redundant with II vegetation, F2: overlaps with I species-based indicators, and F3 can be closely related to the ES water quality regulation -- so even if these processes are considered relevant in a specific context, their indicators should probably be used in other parts of the framework). Furthermore, the criterion of reliability and feasibility can also be limiting factors for ecosystem function metrics which are difficult to measure, or which come from complex modelling studies.

The class **physical and chemical state** (IV) can host various 'degradable environmental stocks' (e.g. soil organic carbon, tropospheric ozone, water table level, impervious surfaces, as discussed in Chapter 2.3) which directly change (deplete, accumulate) during a degradation process (e.g. erosion, pollution, desiccation, or soil sealing).[21] This class is able to accommodate several important pressures, in a way that is compatible with accounting (the pressures are related to the changes in the indicators).

The **landscape-level pattern** (V) of ecosystem mosaics can also be considered as a relevant class of characteristics, which can describe the integrity of landscapes at broader spatial scales, and also exerts influence on a broad range of ecosystem services (Verhagen et al., 2016). This class typically comprises simple indicators of landscape diversity and connectivity/fragmentation.[22]

3.2 Further types of indicators

The ECI typology, as introduced in the previous chapter, does not cover all policy-relevant environmental indicators. Both the EBVs and the case studies reviewed in DP 2.2 contain some variables/indicators that cannot be hosted in the five classes discussed above. In general, these indicators violate some of the selection criteria to some degree, so this does not necessarily mean that the ECI classification would not be *exhaustive*. In the following we discuss the most important types of these indicators in the context of the selection criteria. We hope that this way we can provide meaningful material for the SEEA EEA consultation process, which should decide about the inclusion or exclusion of these further types in the list of ECI classes.

- **Pre-aggregated indicators** (case studies 3, 6, 7, 9, 11, 12, 14-16, 18, 22, 23): Data collected and processed for various policies are often available in a highly aggregated format. This aggregation may combine data from several ECI classes, but other data types that are irrelevant or problematic for ecosystem condition accounting may also be involved. Driven by the necessity to reuse what's already available, such pre-aggregated indicators are often considered in practical ecosystem condition accounts. Such data might violate high numbers of criteria (e.g. *reliability*, *spatial & temporal consistency*, *state orientation*, and *framework conformity*). If these violations are minor, then the pre-aggregated index should be considered in the ECI class where it most logically belongs (typically I. species-based indicators). Pre-aggregated indices covering multiple ECI classes should be avoided, if possible. The ideal practice would be to add all

Also the case studies reviewed in DP 2.2 propose surprisingly few condition indicators that can be linked to EBVs F1-F3 (e.g. NPP was proposed as (the only) general indicator of terrestrial condition in case study 2).

[21] This offers another plausible way for dividing this class (IV) into subclasses: according to the main 'pressure types' (e.g. the 'HIPOC' typology of pressures, see e.g. Maes et al., 2018). Nevertheless, for accounting purposes, aggregation according to ecosystem compartments (the physical components of 'natural capital') seems to be more relevant, so the main proposal in Table 2 follows these physical compartments.

[22] Landscape diversity is typically an integrative (non ecosystem-type-specific) characteristic of the landscape, whereas connectivity/fragmentation metrics measure spatial configuration from the perspective of a given (group of) ecosystem type(s). Landscape diversity, and particularly connectivity can be interpreted very differently in terrestrial, freshwater, and marine biomes. For some ecosystem types, which themselves are "mosaics" of relevant subtypes (e.g. a cropland with nested seminatural vegetation fragments), the abundance or the spatial pattern of these subtypes can also be hosted under this ECI class.

relevant characteristics individually, and perform the appropriate aggregations within the condition account itself.[23]

- **Ecosystem extent** (EBV E2): The extent of the main ecosystem types should definitely go under the ecosystem extent component of the SEEA EEA framework (*framework conformity*).[24] The extent of a 'minor' ecosystem types can be registered in ECI class V, if necessary.
- **Accessibility** (case studies 11, 12, 14, 20-22): Distance from roads[25] or human population centres appears in a high number of condition accounts, yet they do not necessarily meet some of the selection criteria. Such 'accessibility' indicators can be seen as a factor behind ES demand, and may be considered as something external to the studied ecosystem (violating *framework conformity*, *state orientation*, or even *system scope*). Normativity can also be an issue: if an ecosystem becomes more accessible (e.g. there is a new highway), would we like to see this as a condition improvement or degradation?
- **Protected areas** (case studies 10, 20-22): Administrative land designations (including the status and degree of nature protection) do not reflect the state of an area, but rather a human response to degradation or perceived land value. Using such indicators in condition accounts violates state-orientation (see also Chapter 2.3).
- **Pressures** (case studies 1, 2, 5, 7): As discussed in Chapter 2.3, 'raw' pressure indicators (e.g. pollutant loads, habitat loss) should be avoided and preferably the underlying 'degradable stocks' (e.g. pollutant concentrations) should be used instead as condition indicator. If this is not possible, and pressures are still used as a proxy, then they should be assigned to the same ECI class that the underlying degradable stock would belong to (in most cases this will be IV).
- **Natural resource management** (case studies 20, 21): Ecosystem management (grazing, felling, fishing, agriculture...) is not necessarily considered as an internal part of the studied ecosystems (*system scope*, *state orientation*). If management is considered as an ecosystem characteristic then it should either be characterized with its intensity under ECI class III (as an 'anthropogenic' disturbance regime), or, if there is an underlying stock that is being extracted (timber, fish) this stock can be considered under class I, II, or IV (where it best fits).
- **Certificates, audits** (case studies 14, 22): Evaluation by companies or organizations (e.g. the 'blue flag' certificate for EU beaches, or the 'green flag' certificate for UK urban parks) cannot be seen as 'original' measurements (*reliability*), and such audits need to rely on primary (socio-) ecological data.[26] Certificates come with costs and they need to be applied for, thus the absence

[23] One of the main functions of the ECI typology is to provides a standardized aggregation scheme that can be meaningfully used across countries, continents and ecosystem types. Such "overarching" pre-aggregated indices might violate this function.

[24] Yet, almost all the ecosystem condition accounts that have been reviewed in DP2.2 include the extent of ecosystems in the condition table.

[25] The density of human populations or road networks (e.g. forest roads, or mountain trails, like in case study 20) can be considered either as an accessibility indicator, or as a pressure/management indicator (characterizing the 'infrastructure' for ES extraction), but they violate some of the criteria in both cases.

[26] Similarly to pre-aggregated indicators, in the case of certificates the ideal practice would also be to encode all relevant characteristics individually in the condition account, and create appropriate aggregated indices there (as a substitute of the audit process).

of a certificate does not mean that the location in question would not meet the necessary qualifications (*reliability*, *state orientation*). If such data are still considered as necessary, they should be classified under the ECI class where they best fit (probably IV. in most cases).

- **Species genetic diversity** (EBV A1-4): This type of data is not expected to be available on spatial and temporal scales that would make them relevant for SEEA EEA (*feasibility*). Existing data can be challenging to be linked to locations and/or timelines (*spatial & temporal consistency*). In case such indicators would be considered as relevant, they should definitely be classified under species-based indicators (I).
- **Species population structure** (EBV B3) **and traits** (EBV C2-6)[27]: Similarly to genetic diversity, data availability can be challenging (*feasibility*), especially for spatially explicit data (many of the possible trait variables are non-spatial: *spatial & temporal consistency*). *Normative* interpretations can also be challenging.
- **Species population phenology** (EBV C1): The phenological phase of species populations can be challenging to be integrated across *temporal* grain units, and lacks *normativity*. The timing (of the events) should not be seen as "state" by itself (a phase shift would not influence functioning) -- but if it is connected to something else (e.g. an earlier greening is connected to a longer veg.period) than the latter should probably be captured by a state indicator (e.g. more biomass, different species).
- **Stable environmental characteristics** (case study 8): Environmental variables that are virtually constant (e.g. climate,[28] local topography (slope, aspect), or geology) do not meet the criterion of *temporal consistency*, and in many cases they also lack *normativity*.

4. References

Bauler, T. (2012). An analytical framework to discuss the usability of (environmental) indicators for policy. Ecological Indicators, 17, 38–45.

Bland, L. M., Watermeyer, K. E., Keith, D. A., Nicholson, E., Regan, T. J., & Shannon, L. J. (2018). Assessing risks to marine ecosystems with indicators, ecosystem models and experts. Biological Conservation, 227, 19-28.

Cash, D. W., Clark, W. C., Alcock, F., Dickson, N. M., Eckley, N., Guston, D. H., … Mitchell, R. B. (2003). Knowledge systems for sustainable development. Proceedings of the National Academy of Sciences of the United States of America, 100(14), 8086–8091.

Czúcz, B., Götzl, M., Schwaiger, E., Sonderegger, G., & Schwarzl, B. (2017). Working paper on functional relationships between ecosystem characteristics and services in support of condition assessments.

[27] These EBVs have been extensively reviewed by Kissling et al. (2018).

[28] Climatic variables also used to be constant, but due to climate change this might change in the future. Nevertheless, the ecological impacts of climate change can already be covered by the classes listed above, while climate itself is rather external to the ecosystems (scope, state-orientation). In addition climate variables are well covered by different international conventions and data exchange processes (IPCC, WMO), so it might not be a good idea to dilute ecosystem condition accounts with climate variables.

ETC/BD Working Paper. 175A https://bd.eionet.europa.eu/Reports/ETCBD TechnicalWorkingpapers/Functional_relationships_ecosystem_condition_assessments

Dale, V. H., & Beyeler, S. C. (2001). Challenges in the development and use of ecological indicators. Ecological Indicators, 1(1), 3–10.

Erhard, M., Teller, A., Maes, J., Meiner, A., Berry, P., Smith, A., ..., Christiansen, T. (2016). Mapping and Assessment of Ecosystems and their Services. Mapping and Assessing the condition of Europe's ecosystems: Progress and challenges. 3rd Report. Publications office of the European Union, Luxembourg. http://catalogue.biodiversity.europa.eu/uploads/document/file/1328/ 3rdMAESReport_Condition.pdf

Giannetti, B. F., Bonilla, S. H., Silva, C. C., & Almeida, C. M. V. B. (2009). The reliability of experts' opinions in constructing a composite environmental index: The case of ESI 2005. Journal of Environmental Management, 90(8), 2448–2459.

de Groot, R. S., Alkemade, R., Braat, L., Hein, L., & Willemen, L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. Ecological Complexity, 7(3), 260–272.

Heink, U., & Kowarik, I. (2010). What are indicators? On the definition of indicators in ecology and environmental planning. Ecological Indicators, 10(3), 584–593.

IPCC (2014). 2013 Revised Supplementary Methods and Good Practice Guidance Arising from the Kyoto Protocol. Hiraishi, T., Krug, T., Tanabe, K., Srivastava, N., Baasansuren, J., Fukuda, M. and Troxler, T.G. (eds), IPCC, Switzerland. http://www.ipcc-nggip.iges.or.jp/public/kpsg/index.html

Kandziora, M., Burkhard, B., & Müller, F. (2013). Interactions of ecosystem properties, ecosystem integrity and ecosystem service indicators — A theoretical matrix exercise. Ecological Indicators, 28, 54–78.

Kissling, W. D., Ahumada, J. A., Bowser, A., Fernandez, M., Fernández, N., García, E. A., … Hardisty, A. R. (2018). Building essential biodiversity variables (EBVs) of species distribution and abundance at a global scale. Biological Reviews, 93(1), 600–625.

Maes, J., Teller, A., Erhard, M., Grizzetti, B., Barredo, J.I., Paracchini, M.L., … Werner, B. (2018). Mapping and Assessment of Ecosystems and their Services: An analytical framework for ecosystem condition. Publications office of the European Union, Luxembourg. https://biodiversity.europa.eu/maes

Müller, F. (2005). Indicating ecosystem and landscape organisation. Ecological Indicators, 5(4), 280–294.

Niemeijer, D., & de Groot, R. S. (2008). A conceptual framework for selecting environmental indicator sets. Ecological Indicators, 8(1), 14–25.

Noss, R. F. (1990). Indicators for Monitoring Biodiversity: A Hierarchical Approach. Conservation Biology, 4(4), 355–364.

Palm, W. J. (2005). System dynamics. McGraw-Hill Higher Education.

Parrochia, D (2019): Classification. In: The Internet Encyclopedia of Philosophy (ISSN 2161-0002). https://www.iep.utm.edu/

Pereira, H. M., Ferrier, S., Walters, M., Geller, G. N., Jongman, R. H. G., Scholes, R. J., … Wegmann, M. (2013). Essential Biodiversity Variables. Science, 339(6117), 277–278.

Santos-Martín, F., Zorrilla-Miras, P., Palomo, I., Montes, C., Benayas, J., & Maes, J. (2019). Protecting nature is necessary but not sufficient for conserving ecosystem services: A comprehensive assessment along a gradient of land-use intensity in Spain. Ecosystem Services, 35, 43–51.

Schwarz, N., Moretti, M., Bugalho, M. N., Davies, Z. G., Haase, D., Hack, J., … Knapp, S. (2017). Understanding biodiversity-ecosystem service relationships in urban areas: A comprehensive literature review. Ecosystem Services, 27, 161–171.

Smith, A. C., Harrison, P. A., Pérez Soba, M., Archaux, F., Blicharska, M., Egoh, B. N., … Wyllie de Echeverria, V. (2017). How natural capital delivers ecosystem services: A typology derived from a systematic review. Ecosystem Services, 26, 111–126.

Stevens, S. S. (1946). On the Theory of Scales of Measurement. Science 103: 677–680.

United Nations, European Union, Food and Agriculture Organization of the United Nations, Organisation for Economic Co-operation and Development, World Bank (2012). System of Environmental-Economic Accounting 2012 — Experimental Ecosystem Accounting. United Nations, New York. https://seea.un.org/sites/seea.un.org/files/seea_eea_final_en_1.pdf

United Nations (2017). Technical Recommendations in Support of the System of Environmental-Economic Accounting 2012 – Experimental Ecosystem Accounting. United Nations, New York. https://seea.un.org/sites/seea.un.org/files/technical_recommendations_in_support_of_the_seea_eea_final_white_cover.pdf

United Nations (2018) SEEA Experimental Ecosystem Accounting Revision 2020. Revision Issues Note. https://seea.un.org/sites/seea.un.org/files/documents/EEA/eea_2020_revision_issue_final_jul2018.pdf

Verhagen, W., Van Teeffelen, A. J. A., Baggio Compagnucci, A., Poggio, L., Gimona, A., & Verburg, P. H. (2016). Effects of landscape configuration on mapping ecosystem service capacity: a review of evidence and a case study in Scotland. Landscape Ecology, 31(7), 1457–1479.

Annex 1: Crosswalks linking ECI classes to other relevant indicator typologies

Table A1.1: Crosswalk to ECI classes (overlaps can be partial, strong matches are highlighted in bold)

ECI classes	**EBV (Pereira et al., 2013)**	**OpenNESS (Smith et al., 2017)**	**Ecosystem integrity (Müller, 2005)**	**EU ecosystem condition assessment (Maes et al., 2018)**
Species-based indicators	**B1 Species distribution** **B2 Population abundance** **D1 Taxonomic diversity** E3 Ecosystem composition by functional type	**C Presence of a particular species, functional group or trait** **D Biological and physical diversity**	**Biotic structures**	**Structural ecosystem attributes based on species diversity** Structural ecosystem attributes monitored under the EU nature directives Structural soil attributes
Vegetation and biomass	**E1 Habitat structure**	**B Provision of supporting habitat** D Biological and physical diversity	Energy balance	**Structural ecosystem attributes (general)**
Ecosystem processes	D2 Species interactions F1 Net primary productivity F2 Secondary productivity F3 Nutrient retention **F4 Disturbance regime**	**A Amount of vegetation** E Abiotic factors	Energy balance Water balance Matter balance	**Functional ecosystem attributes (general)**
Ecosystem physical and chemical state		**E Abiotic factors**	Water balance Matter balance	Pollution and nutrient enrichment, Structural soil attributes, Habitat conversion and degradation
Landscape pattern		D Biological and physical diversity	Abiotic structures	Structural ecosystem attributes (general)

Table A1.2: Crosswalk to the further types of indicators discussed in this paper

ECI classes	**EBV (Pereira et al., 2013)**	**OpenNESS (Smith et al., 2017)**	**Ecosystem integrity (Müller, 2005)**	**EU ecosystem condition assessment (Maes et al., 2018)**
Pre-aggregated indicators				Structural ecosystem attributes based on species diversity, Structural ecosystem attributes monitored under the EU nature directives
Ecosystem extent (E2)	E2 Ecosystem extent and fragmentation	B Provision of supporting habitat		Habitat conversion and degradation (Land conversion), Structural ecosystem attributes monitored under the EU nature directives
Accessibility				
Protected areas				
Pressures				Habitat conversion and degradation (Land conversion) Climate change Pollution and nutrient enrichment Over-exploitation Over-harvesting Introduction of invasive alien species Other pressures
Natural resource management (EBV F1-3)				Over-exploitation Over-harvesting
Certificates, audits				
Species genetic diversity	A1 Allelic diversity A2 Co-ancestry A3 Population genetic differentiation A4 Breed and variety diversity	D Biological and physical diversity	Biotic structures	

Species population structure and traits	B3 Population structure by age/size class C2 Morphology* C3 Reproduction* C4 Physiology* C5 Movement*	C Presence of a particular species, functional group or trait	Biotic structures	Structural ecosystem attributes (general)
Species population phenology	C1 Phenology*			Functional ecosystem attributes (general)
Stable environmental characteristics		E Abiotic factors		Climate Structural soil attributes

* EBVs for species traits listed after Kissling et al. (2018)

Annex 2: The condition indicators reviewed in discussion paper 2.2 grouped according to the ECI classes.

Table A2.1: Condition indicators from the case studies grouped according to broad ecosystem types and ECI classes. gen: any terrestrial ecosystem, urb: urban, cro: croplands, for: forests, shr: shrublands (incl. heathland), gra: grasslands, wet: wetlands, frw: freshwater (rivers, lakes, reservoirs); coa: coastal ecosystems (mangroves, estuaries, lagoons, transitional waters, beaches, sea cliffs), mar: marine (open sea) ecosystems. The numbers listed in each cell correspond to the ID number of each case study used in DP 2.2.

ECI classes and subclasses	**Broad ecosystem types**									
	gen	**urb**	**cro**	**for**	**shr**	**gra**	**wet**	**frw**	**coa**	**mar**
I. Species-based indicators (compositional characteristics)										
corals										2
(macro)invertebrates						12,20 21		16	5	
fish							12	12	5	2
birds	17		12	12, 13	12		10,11 12	11	12	
synthetic species and habitats-based indices (red-list indices, conservation status of species or habitats)	6, 8?, 12,17 19	14?						10	5, 22	
II. Vegetation and biomass (structural characteristics)										
vegetation cover (e.g. LAI, urban green cover...)	3,16	14?		10						
biomass / carbon / timber stock	12			10, 13						
litter	3									
forest age (age classes)				4,10						
III. Ecosystem processes (functional characteristics)										
flood risk	16			10						
fire risk	20				20	20				
NPP, biomass growth, carbon uptake	2, 16			12						
IV. Ecosystem physical and chemical state (abiotic characteristics)										
air	8	14?			20	20,21				
water quality (e.g. pollutant concentrations, dissolved oxygen, Chlorophyll-a, turbidity)	8						20	9,11,1 2,16	1,5, 22	2
water quantity (e.g. hydrological flow, reservoir stock, groundwater table...)	8					21		9		
soil (e.g. nitrogen content, heavy metal content, soil carbon stock)	8,12,2 2			10,13	20	20	11,20			
V. Landscape pattern (landscape-level characteristics)										
connectivity/fragmentation (barrier density, patch size, shape...)	3,7,8			13?		21	??			
the presence/abundance of specific habitat (sub)types (e.g. riparian habitats, seagrass fields, forest types)				4,10				9,16	5	2

Table A2.2: Condition indicators which cannot be unambiguously linked to an ECI class, grouped according to broad ecosystem types. gen: any terrestrial ecosystem, urb: urban, cro: croplands, for: forests, shr: shrublands (incl. heathland), gra: grasslands, wet: wetlands, frw: freshwater (rivers, lakes, reservoirs); coa: coastal ecosystems (mangroves, estuaries, lagoons, transitional waters, beaches, sea cliffs), mar: marine (open sea) ecosystems. The numbers listed in each cell correspond to the ID number of each case study used in DP 2.2.

ECI classes and subclasses	**Broad ecosystem types**									
	gen	**urb**	**cro**	**for**	**shr**	**gra**	**wet**	**frw**	**coa**	**mar**
Pre-aggregated indices (e.g. ecosystem integrity, naturalness)	3, 6, 7, 15, 18	14					23	9,11,12,16	22	
Accessibility (distance to ecosystems from population centres, length of trails)	12	14			20	20,21	20	11	22	
Protected areas (or other similar administrative designations -- e.g. Natura2000 (EU), SSSI (UK)...)				10	20	20,21	20		22	
Raw pressures (e.g. pollutant loads, habitat loss)	7								1,5	2
Management intensity (e.g. grazing)					20	20,21				
Certificates (e.g. blue flag (EU beaches), green flag (UK urban parks))		14							22	
Abiotic / climatic characteristics (e.g. annual rainfall, annual number of growing days)	8									