# Using the natural capital index framework as a scalable aggregation methodology for regional biodiversity indicators

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*Abstract*

There is an increasing need for aggregated biodiversity indicators to inform policy decisions at all levels from local to global. Despite their similar policy goals, low-level (e.g. local, regional) and high-level (e.g. continental, global) indicator development is generally performed independently, and the resulting indicators are often incompatible both in their structure and data requirements. In this paper we focus on a particularly flexible aggregation framework originally developed for global assessments, the Natural Capital Index. We show that with the use of appropriate fine-scale data, the NCI framework can be applied in low-level policy contexts as well. To support this statement, we show that several established low-level indicators are essentially conforming to the NCI framework, and can be seen as existing low-level NCI implementations. The concept is illustrated with an implementation for Hungary, and the potential advantages and shortcomings of low level NCI implementations are discussed. NCI-based low level indicators can be implemented in any region, where a local indicator of ecological quality is systematically surveyed. Given the recent surge in monitoring activities worldwide, fuelled by global change and reporting obligations, fine-scale NCI implementations can become important additions to existing ecological state indicators useful in a wide range of local and regional policy contexts.

*Keywords:*

biodiversity index, habitat quality, naturalness, vegetation condition, ecosystem services

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Czúcz B., Molnár Zs, Horváth F, Nagy G G, Botta-Dukát Z, Török K (in press): Using the natural capital index framework as a scalable aggregation methodology for local and regional biodiversity indicators. *Journal for Nature Conservation*. doi:10.1016/j.jnc.2011.11.002

## Introduction

There is a long ongoing dissent between scientists and policy-makers regarding whether available information on biodiversity should or should not be aggregated into a limited number of comprehensive indices. Whereas scientists are mostly concerned with accuracy, reliability and replicability of their results, policy-makers are only confused by the overwhelming diversity of information available in the pure scientific results, and therefore need clear key messages (ten Brink, 2006). As it becomes increasingly obvious that the fundamental challenges of biodiversity conservation cannot be addressed effectively without extensive societal cooperation (MEA, 2005), the question of measuring and communicating the ecological state of our environment is getting more and more into the focus of applied ecological research.

One particularly useful way to provide key messages on the state of the ecosystems is in the form of concise figures, like economic and social indicators (e.g. GDP or unemployment rate), describing current state and tendencies of the studied systems. Useful indicators should be user-driven and policy-relevant, simplifying information in order to help communicate complex phenomena effectively. On the other hand, indicators also need to be factual and responsive to changes in time and/or space, representing a good compromise between scientific rigor and conceptual simplicity (CBD, 1997). Such indicators are often called 'biodiversity indicators', nevertheless, they are much more than simple metrics to measure the diversity of organisms, they should rather be regarded as general indicators monitoring the state of the ecosystems (ten Brink 2006).

The construction of reliable biodiversity indicators is one of the most actual and pressing tasks for nature conservation. Both the Convention on Biological Diversity (CBD) and the European Union finance projects to coordinate the identification and development of a core set of policy-relevant biodiversity indicators (CBD, 2003; EEA, 2007). In response to the ongoing policy efforts, several aggregated biodiversity indicators have been proposed in the last few years, generally synthesizing available information on broad spatial scales, mostly based on changes in the abundance of species (Loh et al., 2005), the spatial extent of ecosystems (e.g. Mayaux et al., 2005; EEA, 2006) or both (e.g. ten Brink, 2000; Scholes and Biggs, 2005; Butchart et al., 2006; Certain et al., 2011).

As most of the biodiversity metrics in focus of the international indicator development process rely on data collected at coarse resolution over broad spatial scales, they are useless for local or regional policy decisions. Several suggested ecosystem state indices are uninterpretable for sub-national scales (e.g. Butchart et al., 2005, 2006), while others, like the 'ecosystem coverage' indicator proposed by EEA (2007) lose resolution and reliability in a fine scale context. While detecting large-scale trends is an unquestionably important goal for biodiversity indicators, there are many cases when information at local/regional scales is needed (Bubb et al., 2005). As most of the land-use decisions are made at the local or regional levels, instruments of decision-support are also most needed at these low levels. Even institutionalized forms of such decision making processes, like environmental impact assessments (EIA) or strategic environmental assessments (SEA) in the European legislation, could be benefited of ecological state indicators of high spatial and thematic resolution. Nevertheless, there are no widely accepted flexible and easily adaptable methodologies to construct aggregated biodiversity indicators at the local or regional levels. In our opinion, this constitutes an important gap in applied research and science policy communication.

From a certain perspective all quantitative descriptors calculated from local environmental data can be considered as fine-scale indicators, including e.g. the abundance of a single species, or a single structural attribute of the local community. But in order to be congruent with the broad-scale indicators, in the following sections we only consider those fine-scale *biodiversity indicators*, which aim at giving an overall quantitative description of *general ecosystem state* at both site (local) and landscape (regional) scales. Such fine-scale indicators (e.g. Parkes et al., 2003; Ferrari et al., 2008) are generally composed of (1) a site level indicator of local ecosystem state (e.g. its 'naturalness' or 'vegetation condition') and (2) an aggregation framework, both of which have applicability outside the regions in which they were developed.

Aggregation frameworks play a crucial role in biodiversity indicators for both low-level and high-level policy contexts. In this paper we show that the natural capital index (NCI), a particularly flexible indicator framework originating from the international biodiversity policy arena (ten Brink, 2000), is essentially a scale-free aggregation framework, which can be used to develop meaningful biodiversity indicators based on a broad range of ecological data at all scales. We begin with introducing the NCI framework and its existing broad-scale implementations, as well as its potential application as a fine-scale aggregation scheme for low-level policy contexts. We point out that several already established regional indicators (e.g. the 'index of landscape conservation' (ILC): Pizzolotto and Brandmayr, 1996; the 'hemeroby index': Steinhardt et al., 1999; or the 'habitat hectares': Parkes et al., 2003) are essentially conforming to the NCI framework, and can be seen as existing low-level NCI implementations. Next, we provide further illustration to our statements with a new national NCI implementation for Hungary, based on the results of a recent national vegetation survey. Finally we discuss the advantages and shortcomings of the NCI aggregation framework in fine-scale indicator development, as well as its potential contribution to the national and international indicator sets.

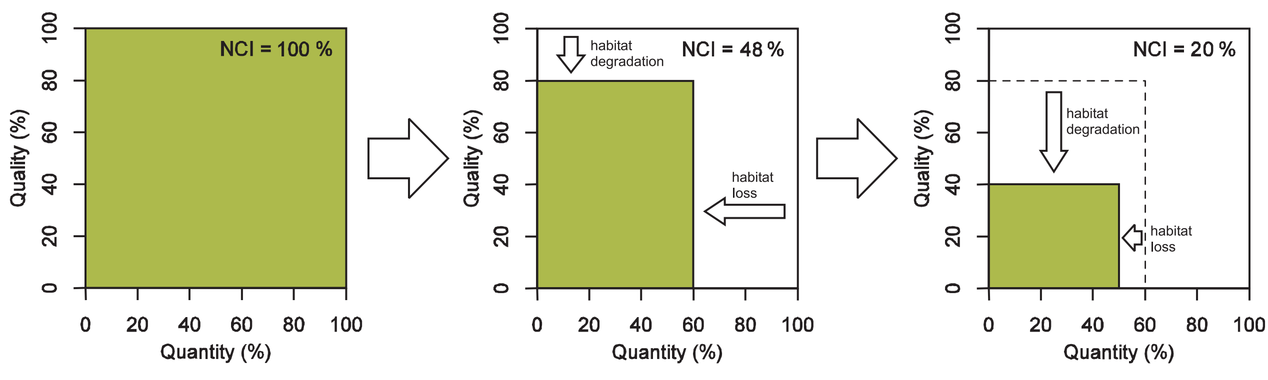


Figure 1: Natural capital is defined as the product of remaining ecosystem size (quantity) and its quality. For example, if the remaining ecosystem size is 50 %, and its quality is 40 %, then 20 % of the natural capital remains.

## The natural capital index framework

*The original concept and existing high-level implementations*

The natural capital index (NCI) is one of the first high-level aggregated biodiversity indicators proposed for widespread international application (CBD, 1997; ten Brink, 2000; ten Brink et al., 2002). The construct of NCI relies on a simple and straightforward conceptual model:

$$NCI = ecosystem\ quality \times ecosystem\ quantity = q \cdot a$$

where both quality and quantity are expressed relative to an 'optimal' or 'intact' baseline. Accordingly, both are scaled between 0 and 1 (dimensionless), where for quality 1 means the intact state, and the quantity of 1 means that the natural ecosystems still occupy the entire study area. Practically, if there is more than one distinct ecosystem (biome, habitat type, patch, etc. depending on the study goals and the exact formulation of the indicator) in the target area with separate estimations for quantity and quality, then NCI turns into:

$$NCI = \sum_{i=1}^{n} q_i a_i$$

In this case the areas ($a_i$: ecosystem quantity) are always relative to the total study area, so that

$$\sum_{i=1}^{n} a_i + a_{anthropogenic} = 1$$

where $a_{anthropogenic}$ means the relative quantity of all anthropogenic areas within the study area which are excluded from the study (or, equivalently, which are considered to have 0 quality).

The concept of NCI is based on the assumption that biodiversity loss can be modeled as a process driven by two main components: habitat loss due to conversion of natural areas into agricultural fields or urban areas, and degradation of the remaining habitat patches, caused by overexploitation, pollution, fragmentation, invasive species, etc. Thus, NCI summarizes the extent to which a landscape has preserved its original (baseline) natural capital (Figure 1; ten Brink, 2007). Combining quality and quantity into one indicator, NCI relies on a hypothetical equivalence between smaller intact, and larger, but degraded patches in terms of ecological value.

NCI as explained above is more an aggregation framework than a standalone indicator. It provides a powerful and flexible methodology for aggregation, but it still leaves many 'degrees of freedom' for implementation details (the selection of 'ecosystem types', the measurement of quality, etc.) depending on the policy objectives and the availability of data sources. Existing NCI implementations focus at the high-level policy contexts, as can be seen from the fact, that NCI has only been applied in global (UNEP, 1997, 2002) and national (ten Brink et al., 2002; CBD, 2005) assessments yet. The most problematic question of high-level NCI implementations is the estimation of habitat quality for the identified habitat types, based on data available for the entire study area. In this respect two solutions can be found (ten Brink, 2000):

(1) based on species abundance data for several characteristic species of the different habitat types, ecosystem quality can be estimated (abundance-based NCI – ten Brink et al., 2002; CBD, 2005),

(2) in the absence of suitable observations, environmental and socio-economic data describing the different pressures (acidification, nitrogen deposition, climate change, GDP, etc.) affecting ecosystem quality are supposed to suffice as surrogates (pressure-based NCI – UNEP, 1997, 2002).

The second solution was usually resorted to only in case of global ecosystem assessments (UNEP, 1997, 2002), and has the advantage, that starting out from global pressure scenarios, future projections on the state of biodiversity can be made. However, due to the complex interactions and the inertia of the modeled ecosystems, meaningful calibration of pressure-based NCI values is almost impossible, which seriously questions its applicability in diagnostic applications. Consequently, according to the inventors of the NCI concept (ten Brink, 2000), habitat quality should be approximated with species abundance data whenever such data are available, which is generally the case for country level NCI applications (e.g. for the Netherlands, the Philippines, Ukraine or Ecuador: ten Brink et al., 2002; CBD, 2005). In this case, the calculation of abundance trends is usually based on national monitoring data for native birds and butterflies, or other easily observable vagile animals, known to be sensitive to anthropogenic effects. The same concept has been extended to a larger set of indicators (including several ecological characteristics in addition to species abundance data) in the case of the Norwegian Nature Index (NI; Certain et al., 2011; Nybø et al., 2011).

*Aggregating fine scale data for low-level policy contexts*

In this paper we point out a third alternative for including quality in NCI implementations, which relies on local (site-scale) measures for ecosystem state. There are several alternative definitions of such measures primarily based on local structure, composition, and type-specific key processes (Noss, 1990), some examples are ‘vegetation condition’ (Gibbons et al., 2006, 2008), naturalness (e.g. Machado, 2004) or hemeroby (e.g. Sukopp et al., 1990). Ecosystem health (e.g. Costanza et al., 1992) and ecological integrity (e.g. Woodley et al., 1993) also indicate very similar concepts. Such local indicators are usually estimated (1) on a simple ordinal scale in the field by comparing observations to a standardized list of criteria (e.g. Machado, 2004; Molnár et al., 2007), or (2) composed as a weighted aggregation of several field-observed nominal or ordinal scale indicators (e.g. Bartha, 2004; Gibbons and Freudenberger, 2006; Standovár et al., 2006), or (3) based on field-calibrated modeling and/or remote sensing data (e.g. Li and Kräuchi 2004; Cohen et al. 2005; Gibbons et al. 2008). For the sake of simplicity, in the following we will refer to such site-level ecosystem state indicators as local ‘naturalness’ indicators.

Most authors, who have developed local naturalness indicators, do not provide methodologies for spatial aggregation. This does not question the usefulness of their results, since naturalness indicators themselves are already instantaneously applicable in several low level policy questions. Nevertheless, some authors go one step further, and also propose methodologies for the spatial aggregation of local quality ratings into landscape level state indicators. Having performed a thorough literature survey, we found several solutions for this, including three of presumably independent origin: the ‘index of landscape conservation status’ (ILC; Pizzolotto and Brandmayr, 1996), the ‘hemeroby index’ (M; Steinhardt et al., 1999) and the ‘habitat hectares’ approach (Parkes et al., 2003). The recently proposed ‘index of vegetation naturalness’ (IVN; Ferrari et al., 2008) is essentially a reformulation of ILC, whereas Gibbons et al. (2009) apply a formula very similar to the habitat hectares approach for the ‘regional value’ of an area containing several vegetation fragments. Remarkably, even though starting out from fairly different backgrounds, the found solutions all end up with methodologies that are computationally equivalent to the NCI concept. This relationship is relatively straightforward in the case of the habitat hectares approach and the hemeroby index, but it also holds in the case of ILC (and IVN) as it is shown in Appendix A.

Ricotta et al. (2003) provide further justification for the ILC approach, showing that ILC based landscape evaluation naturally conforms to the partial Lorenz ordering arising from the order of the naturalness categories – which can be seen as an important sanity criterion for any potential aggregation schemes. It can be shown that this partial Lorenz ordering is also preserved by all well defined NCI's (Appendix B).

Accordingly, the NCI aggregation algorithm seems to be a really universal solution, providing a coherent and flexible way for aggregating spatially explicit quality and quantity data at a broad variety of scales, thus establishing a link between indicators for high-level and low level policy contexts. In the following, we illustrate the data requirements, design decisions and potential applicability of a fine-scale NCI framework with a case study for Hungary.

## A multiscale NCI implementation: the 'vegetation-based natural capital index' of Hungary

### *Study area and data source*

The Hungarian national NCI implementation (called 'vegetation-based natural capital index' or vbNCI) is based on standardized vegetation survey data with a habitat quality attribute. The study area is the country of Hungary (93 000 km$^2$) in Central Europe. The primary data source is the MÉTA database, a grid-based, multi-attributed vegetation map, based on the results of a broad-scale vegetation mapping exercise between 2003 and 2006 covering the entire country (Molnár et al., 2007; Horváth et al., 2008). Altogether 86 different types of natural and semi-natural habitats were distinguished, with a detailed Habitat Guide to assist the participants and standardize the process (Bölöni et al., 2007). The spatial units of the mapping were grid cells with a size of 35 ha constituting a regular hexagonal grid covering the entire country. Within the grid cells, a list of the habitats was given, with the area (relative to the grid cell), and the estimated naturalness as attributes. There were several other attributes (connectivity, land use, invasion, etc) describing the state and pressures apparent at the habitat or grid cell level. Most attributes were collected at nominal or ordinal scales to speed up the mapping process and ease standardization. The identification of attribute levels was supported by a detailed protocol called Mapping Guide, and in the case of naturalness standardized lists of habitat-specific criteria were also given in the Habitat Guide (Bölöni et al., 2007). Additionally, in order to enhance the coherence of individual perceptions, all mappers participated in a series of obligatory field training before they could begin working (Molnár et al., 2007).

### *Assessing habitat quality*

During the MÉTA survey naturalness was assessed on a five grade ordinal scale, based on a multitude of characteristics (e.g. presence/abundance of certain species or structural elements) and a detailed protocol for each habitat type (Bölöni et al., 2007; Molnár et al., 2008). To transform the ordinal naturalness values to an absolute scale habitat quality ($q$), we identified two simple weighting schemes: a linear 'equal steps' approach ($q_{lin}$), and a (near-)exponential approach ($q_{exp}$ – see Table 1 and Czúcz et al., 2008). The proposed habitat quality weights are interpreted as quality relative to the baseline of an imaginary ‘ideal state’ for the habitat type, which equals to the presumable pristine state in the case of most habitats, or to a low-intensity traditional land use in the case of semi natural ones, such as hay meadows. The most important difference between the two scalings is the weights assigned to medium quality habitats, which affects the interpretation of quality changes in a degradation or regeneration process. $q_{exp}$ assigns ten times larger weight to a 5 to 4 naturalness decrease than to a 3 to 2 change, whereas $q_{lin}$ considers these changes to be of equal importance. Accordingly, $q_{lin}$ is considered to be a proxy for functional diversity, whereas $q_{exp}$ estimates suitability for rare and endangered species, with the first being much more resilient to degradation due to the high level of functional redundancy generally observable in ecological systems (Fonseca and Ganade, 2001 – a more detailed description and evaluation of the applied weighting schemes can be found in Czúcz et al., 2008). Similar solutions can be found for other well-known biodiversity indicators as well: e.g. in the case of the Red List Index (Butchart et al., 2004, 2005) there are also two different weighting schemes suggested (an ‘equal steps’ with linearly changing weights, and an ‘extinction risk’ using nearly exponential weights). According to the authors’ evaluation, both approaches are based on meaningful theoretical advisement, and the decision of which one to use should be based on preliminary considerations of the objectives of the analysis – exactly as in the case of our two scalings.

Table 1: The two weighting schemes used for transforming the field-estimated ordinal “naturalness” levels (1-5) onto the absolute scale [0,1]: $q_{lin}$: linear (‘equal steps’) weights, $q_{exp}$: exponential weights

| Naturalness | *5* | *4* | *3* | *2* | *1* |
|---|---|---|---|---|---|
| $q_{lin}$ | *0.9* | *0.7* | *0.5* | *0.3* | *0.1* |
| $q_{exp}$ | *0.9* | *0.3* | *0.1* | *0.03* | *0.01* |

*Calculating index values*

Vegetation-based NCI can be calculated for the whole mapped area or any sub-region on any scale with the following formula:

$$vbNCI_{lin/\exp} = \frac{1}{A_r} \sum_{i \in S_r} q_{i,lin/\exp} A_i \text{ , where}$$

$A_r$: area of the examined region (in arbitrary units, e.g. km$^2$),

$S_r$: (the set of) all the individual habitat patches within the examined region,

$A_i$: the estimated area of a habitat patch (in the same units as $A_r$)

$q_{i\ lin/exp}$: the estimated quality of the habitat patch, according to one of the weighting schemes (either $q_{lin}$ or $q_{exp}$; see Table 1).

We calculated vbNCI for both equal steps and exponential weighting schemes for the entire country, as well as the main geographical regions of Hungary (Marosi and Somogyi, 1990;

see Figure 3 and Czúcz et al., 2008 for some examples). The location of the individual habitat patches could only be resolved to the level of the mapping units (grid cells), which may result in limited precision near region boundaries. Accordingly, the total area of the regions has been approximated by the cumulative area of grid cells the centers of which fall within the region. Grid cells with missing data were simply omitted from the analysis (approximately 9.2 % of the data was still missing at the time of the analysis – see Horváth et al., 2008).

The NCI framework provides opportunities for both the overall characterization of larger areas and the flexible spatial and thematic disaggregation of the results. Policy relevant analysis can be obtained by reasonable combinations of spatial and thematic (habitat type specific) detail. As a tool for informative thematic disaggregation, we used the compact and informative diagram type proposed by ten Brink et al., (2002; see also MNP, 2008), presenting the contribution of the major habitat classes to overall habitat quantity and quality in an intuitive way. For this reason the original list of 86 habitat types was grouped into 10 major habitat classes.

*Some diagnostic results*

Figure 2 shows the detailed results for our calculations. It can be seen that only 17 % of the area of the country is still covered by natural or semi-natural vegetation; the remaining 83 % is mostly occupied by arable fields, forestry plantations or settlements. This results in a $vbNCI_{lin}$ of 9.9 % (indicating an overall 90 % loss in terms of functional diversity and the general supporting ecosystem services depending thereon), and a $vbNCI_{exp}$ of 3.2 % (unveiling even greater losses in terms of rare species and the key ecosystem service of biodiversity conservation). Figure 3 illustrates how vegetation-based NCI can provide a diagnostic spatial overview of larger regions. More in depth discussion about the spatial and thematic distribution of vbNCI within Hungary is given in Czúcz et al. (2008).

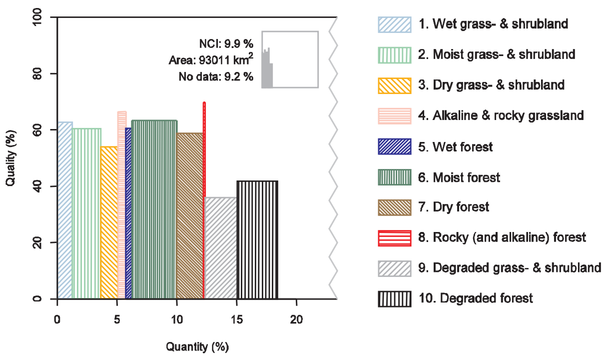


Figure 2: The natural capital index ($vbNCI_{lin}$) of Hungary, shown in a disaggregated structure identifying contributions of 10 main habitat groups. To add perspicuity to the NCI components, the scaling of the axes is not identical, to provide a visual overview of the magnitudes, a pictogram with identically scaled axes is shown in the upper right corner.

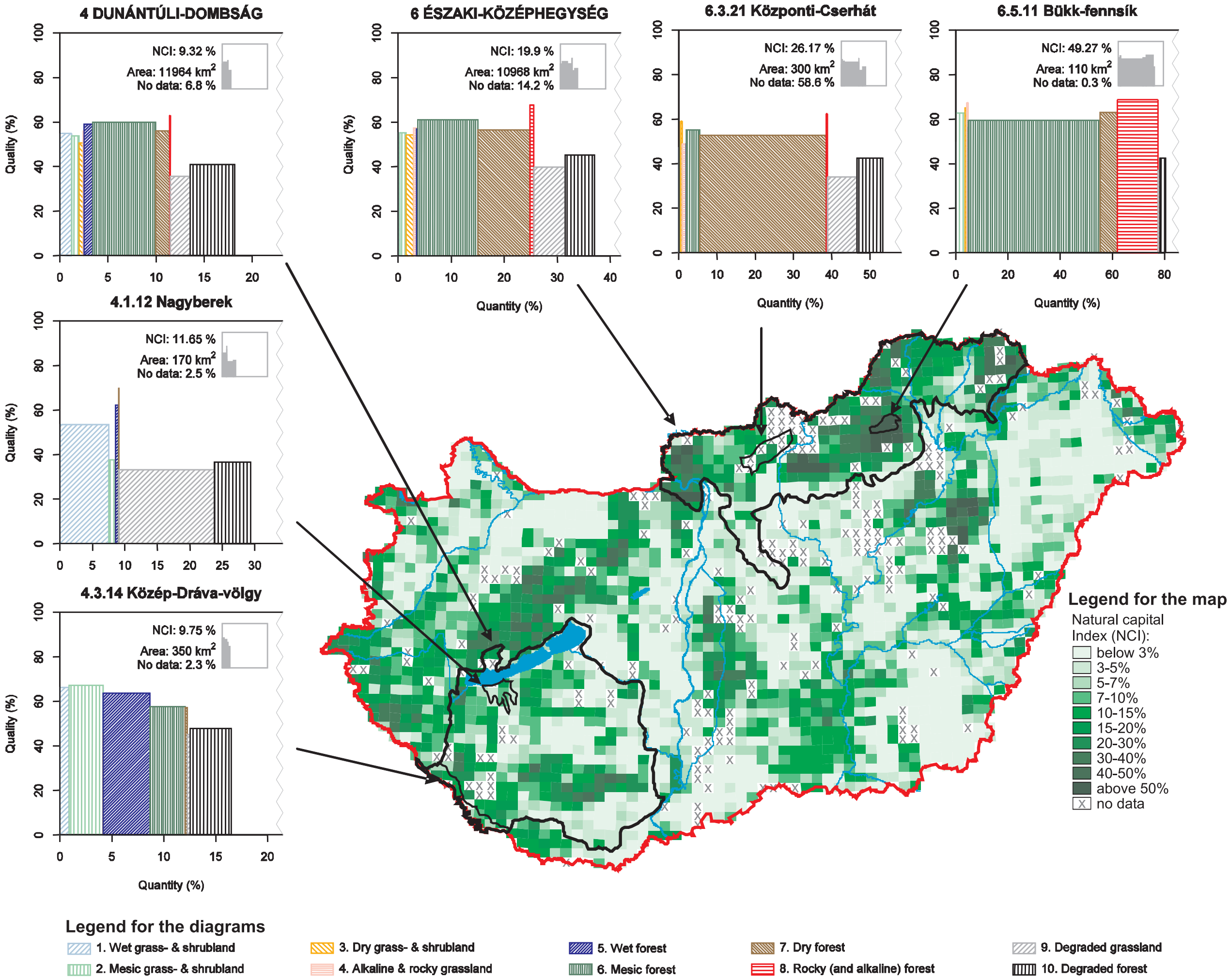


Figure 3: The vbNCI map of Hungary illustrating the flexibility of the NCI framework with respect to spatial and thematic aggregation (overall $vbNCI_{lin}$ values indicated with graduated color, whereas for a few selected geographic macro- and micro-regions a more detailed analysis is given, highlighting the share of major habitat categories in the regional vbNCI)

## Discussion

Being based on a lucid and easy to understand framework applicable on several spatial scales, fine-scale NCI implementations are potentially useful for a wide range of local and regional policy development and evaluation purposes, including environmental impact assessments (EIA) and strategic environmental assessments (SEI). Moreover, being particularly adapted to local scales, such indicators can become powerful tools in environmental communication and education as well. If simple, standardized protocols for naturalness assessment are available, it is feasible to perform local surveys even for non-experts, which means an opportunity to involve local schools, NGOs, etc into local NCI assessments. In Hungary a study is currently underway, in which primary school students are given tasks of evaluating small portions of their local environment with a radically simplified NCI-based methodology. This task is supposed to improve the sense of place and develop a certain 'ecological eye' in the pupils, revealing the real *state* of their very environment in a similar manner as Ecological Footprint describes the *pressures* caused by their everyday activities. Thus, in environmental campaigns pressure and state indicators could be considered in parallel, which can be considered beneficial for awareness raising and local environmental communication.

In the following sections we give a thorough discussion of the potential issues concerning the practical applicability of NCI implementations for policy applications, identifying major research needs wherever possible. We start by discussing what kind of information NCI values are able to provide, and what they are not, paying particular attention to issues arising at different spatial scales. Next we analyze typical data availability situations, and evaluate their relevance to the applicability of NCI-based indicators. And finally, we conclude by discussing the potential role of fine-scale NCI implementations in an international policy context – based on which NCI may become a real link joining distant policy levels from local to international and perhaps eventually global.

*Interpretation at different scales*

As we have seen most fine-scale biodiversity indicators rely on a site-level indicator of local ecosystem state, reflecting certain ecological characteristics of the site and neglecting others. This site-level naturalness indicator is central to all NCI implementations, since the aggregation framework does nothing more than to calculate an area-weighted regional average of this local indicator. Accordingly, NCI cannot capture any information which is not present in the corresponding local naturalness indicator. The implications are twofold:

- By applying different habitat quality evaluations (e.g. using different naturalness indicators, or just different weighting schemes for the same indicator), different NCI values for the same area can be calculated. This might be seen as a shortcoming (for policy makers generally prefer all information condensed into a single number), but taking into aspect the multidimensional nature of society–biosphere relationship (e.g. reflected in the concept of ecosystem services – Daily, 2000; MEA, 2005) this can also be considered to be an advantage. Linking ecosystem services to biodiversity indicators (e.g. by developing service-specific local naturalness indicators) is an important field of further research for the development of all aggregated biodiversity indicators. Nevertheless, NCI, which is an inherently linear aggregation scheme, should only be for those services, the supply of which can be described with extensive variables measured on an approximately linear scale.
- Being based on a site-scale metric, fine-scale NCI implementations can never allow for structures and processes only observable at broader spatial scales. However, several aspects of biological organization can only be observed above site level (Noss, 1990). Particularly, fine-scale NCI implementations cannot account for landscape pattern (spatial structure within habitats, on the other hand, is considered along with habitat quality).

*Cross-sectional and longitudinal comparisons*

Aggregated biodiversity indicators should ideally be available for longer time periods and wide geographical areas to offer insight into both spatial and temporal variability of ecological state. Unfortunately, such data are hardly ever available – either the spatial or the temporal coverage of real-life data generally exhibits smaller or larger deficiencies. However, both ‘cross-sectional snapshots’ and ‘longitudinal monitoring’ of ecosystem state can still be meaningful for practical policy applications.

- Cross-sectional surveys with fine-scale NCI are relatively easily implementable for areas where ‘maps’ (detailed extensive spatial surveys) for a local naturalness indicator are already available (e.g. Victoria / Australia: Newell et al., 2006; Fort Riley / Kansas / USA: Freeman and Delisle, 2004; several islands from the Galapagos and the Canary archipelagos: Machado, 2004; natural parks of the Northern Appeninnes: Ferrari et al., 2008, etc.). Such cross-sectional surveys can be able to provide a flexible framework for policy-relevant spatial comparisons on a broad range of different scales. The case study presented in this paper is an example of this approach.

- On the other hand, existing monitoring networks with wide spatial covering (e.g. Hungarian National Biodiversity Monitoring System: Kovács-Láng et al., 2000; Swiss Biodiversity Monitoring System: Hintermann et al., 2002) may provide an opportunity to implement NCI-based indicators as longitudinal indicators, which are able to track down temporal changes in the different ecosystems / habitat types surveyed. As for Hungary, the similarities in the assessment methods used for the MÉTA mapping and for the field surveys of National Biodiversity Monitoring System in Hungary may yield an opportunity for the combination of spatial and temporal NCI data series. In other cases, existing monitoring activities might need to be improved to be appropriate for NCI calculations, as in the case of the European long term ecological research (LTER) networks (Cocciufa et al., 2006). Reporting obligations of the European countries under the Art 17 of the Habitats Directive (92/43/EEC) may also serve as valuable data sets for national NCI-like aggregations (e.g. Guth and Kučera, 2005).

*The role of NCI in high-level policy contexts*

The NCI framework was originally proposed for high-level policy contexts. Nevertheless, existing high-level implementations have been found too aggregated and difficult to interpret (Klok 2007; Cocciuffa et al., 2007). In our opinion this is related to a conceptual problem of the abundance-based implementations, where species abundance data are considered to yield a consistent proxy for average quality of the different habitat types. However, species abundance data (especially data on easily observable large bodied animals with large area requirements) already incorporate impacts of both quality and quantity of habitat, which results in some ambiguity concerning the separability of quality and quantity impacts, fundamental to the NCI concept. Abundance values of such animals can, in fact, be considered themselves as aggregated ecological state indicators – indicators of the ‘goodness’ of the landscape from the perspective of the species in question. Thus, abundance data from a large number of species can be regarded as a sufficient basis for the compilation of aggregate biodiversity indices, and consequently, there is no need for further complicating the issues with the inclusion of habitat area. This recognition might eventually have lead to the development of the habitat quality part of abundance-based NCI into an autonomous species trend indicator (Mean Species Abundance: de Heer et al., 2005; ten Brink, 2007).

Nevertheless, the problems with abundance-based NCI implementations illuminate some important aspects of potential fine-scale NCI implementations, which make them more desirable for high-level policies:

- Local naturalness indicators often focus on keystone species (most typically plants, which are sessile and thus easily observable, and have generally important community functions constituting the basis of the food chain), as well as other meaningful surrogates (e.g. the presence of dead wood in forests), which indicate good quality habitat for several important groups of species relevant for that habitat type (Keith and Gorrod 2006). Consequently, such local naturalness indicators can be considered as a proxy for habitat quality for a broad range of habitat-specific and/or low mobility organisms – a category largely underrepresented by the most popular high-level biodiversity indicators.

- Apart from questions of underrepresentation, the inclusion of each new independent and reliable data source by itself, increases the robustness of policy evaluation based on the resulting set of indicators. Local naturalness indicators constitute a data source, which are currently excluded from high-level international ecosystem evaluations.

On the other hand, the inclusion of fine-scale NCI indicators (and the underlying local naturalness metrics) into high-level policy settings is limited by some serious problems:

- Local naturalness metrics are generally available only for a limited area. What is worse, the validity of the assessment protocols is also limited to a certain region (for which they have been developed and tested). The application of aggregated NCI over large areas (e.g. as a major international indicator) would demand a reasonably homogeneous underlying layer of fine-scale NCI data. This is hard to be achieved across biome boundaries, with an increasing number of habitat types and diverging criteria for habitat quality evaluation. Widespread international application of intercompatible fine-scale NCI metrics, would demand huge efforts of developing standardized rapid assessment protocols for a broad range of habitats and a long gradient of environmental conditions (not to mention performing the actual assessments using the developed metric). On a global scale this seems to be an impossible mission, nevertheless in some parts of the world (e.g. the European Union) international monitoring networks and reporting obligations may set the scene for such developments (Cocciufa et al., 2006; Guth and Kučera, 2005).

- In the case of broad scale comparisons the aggregation of relative values may not be meaningful if the reference values (absolute values pertaining to the baseline conditions) are significantly different. Accordingly, even if meaningful local naturalness indicators could be found for the entire globe, comparisons between geographically distant regions and highly contrasting ecosystems (e.g. the Sahara and the Amazonas basin) should be interpreted with care.

**Acknowledgements**

This work was supported by the grant NKFP6/013/2005. The contribution of Gergő G. Nagy was supported by the grant TAMOP-4.2.1/B-09/1/KMR-2010-0005. Thanks to Eszter Illyés, János Bölöni and two anonymous reviewers for their insightful suggestions to earlier version of this manuscript, to Tim Hoelzle for improving English and to all people who contributed to the success of the MÉTA program.

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**Appendix A**

Proof that ILC also uses the natural aggregation scheme.

For a region covered by a land use map with *m* categories which can be combined into *n* (< *m*) ordered degradation classes (degrees of naturalness) the index of landscape conservation state (ILC) proposed by Pizzolotto and Brandmayr (1996) is defined as

$$ILC = 1 - \frac{A}{A_{\max}}$$

In this definition

$$A = \sum_{i=1}^{n-1} c_i$$

is an 'artificiality' value, defined as the sum of the cumulative relative areas ($c_i$) of the degradation classes, and

$$A_{\max} = n - 1$$

is the potential maximum of artificiality obtainable in the case of a landscape consisting entirely of the most degraded category. To form these cumulative relative areas the classes are ordered according to their degradation from the most artificial ($i$ = 1) to the most natural ($i$ = $n$), and then

$$c_i = \sum_{j=1}^{i} a_j \ ,$$

where $a_j$ is the relative area of each degradation class (its area divided by the total area of the study area). Accordingly ILC can be expressed as

$$ILC = 1 - \frac{1}{n-1}\sum_{i=1}^{n-1} c_i = 1 - \frac{1}{n-1}\sum_{i=1}^{n-1}\sum_{j=1}^{i} a_j \ \text{, which since } \sum_{i=1}^{n} a_i = 1 \text{ further equals to}$$

$$ILC = \sum_{i=1}^{n} a_i - \frac{1}{n-1}\sum_{i=1}^{n-1}(n-i)a_i = \sum_{i=1}^{n-1} a_i - \sum_{i=1}^{n-1}\frac{n-i}{n-1}a_i + a_n =$$

$$= \sum_{i=1}^{n-1}\left(a_i - \frac{n-i}{n-1}a_i\right) + a_n = \sum_{i=1}^{n-1}\left(1 - \frac{n-i}{n-1}\right)a_i + a_n = \sum_{i=1}^{n-1}\frac{i-1}{n-1}a_i + \frac{n-1}{n-1}a_n = \sum_{i=1}^{n}\frac{i-1}{n-1}a_i \ .$$

Consequently, the aggregation framework applied is equivalent to the natural aggregation of the landscape patches calculated with simple 'equal steps' quality weights of $q_i = (i-1)/(n-1)$.
■

## Appendix B

Proof that the natural aggregation preserves the inherent natural ordering of landscapes.

Let us consider two landscapes (*A* and *B*), which consist of several patches characterized with some sort of naturalness values measured on the same ordinal scale. As values measured at an ordinal scale cannot be averaged directly, it is not possible to decide which landscape is "more natural". However, in certain cases(e.g. when both A and B consist of one single patch, but of different naturalness values) one of the patches can clearly be seen as being of superior naturalness. This potential relationship between landscapes can be generalized using a partial Lorenz ordering (a.k.a. Lorenz majorization – e.g. Mosler, 2001). If for A and B the cumulative relative areas (starting from the most natural category) are all larger for A than for B, then landscape A is said to majorize (dominate) landscape B in terms of naturalness.

Ricotta et al. (2003) provides a proof that the Index of Landscape Conservation (ILC, Pizzolotto and Brandmayr 1996) conserves this majorization intrinsic to the naturalness scale used. We will show here, that the aggregation scheme of the NCI formula using any well-defined quality weights will also conserve this majorization.

Let NCI be a natural capital index defined for a specific region, based on locally measured $q_i$ quality values assigned to patches of size $a_i$. Let **T** indicate the domain of NCI, a set containing all landscapes (e.g. subregions of a geographic region) for which NCI can be evaluated. Accordingly, the NCI of a landscape $T$ ($\in$**T**) ca be computed as

$$NCI(T) = \sum_{i=1}^{n_T} q_{Ti}\, a_{Ti} \,,$$

where $n_T$ is the number of patches in the landscape. We can define a partial Lorenz ordering ($\preceq$) on **T** with the following definition:

$$A \preceq B \;\Leftrightarrow\; \forall q' \in [0,1] \mid F(A,q') \le F(B,q'),$$

where A and B are any two landscapes of **T**, and $F$: **T** → **R** is defined as

$$F(T,q') = \sum_{i=1}^{n_T} a_i \, I(q_i \ge q'),$$

with, and $I$(*condition*) being the indicator function (1 if *condition* is met and 0 otherwise). Thus, $F(T, q_i)$ gives the proportion (relative area) of a landscape $T$ that has a naturalness equal to or better than $q'$.

As Ricotta et al. (2003) pointed out, the preservation of this intrinsic partial Lorenz ordering ($\preceq$) of the landscapes is an important sanity criterion that any spatial aggregation scheme should preserve. We prove here that this ordering is preserved by the natural aggregation scheme, thus:

$$A \preceq B \;\Rightarrow\; NCI(A) \le NCI(B)$$

To show this, we first prove an even stronger lemma: that the formula of natural aggregation (NCI) equals to the Lebesgue integral of the function underlying the intrinsic Lorenz majorization for landscapes suggested by Ricotta et al. (2003). In other words, we show that

$$\forall T \in \mathbf{T}: \quad NCI(T) = \int_{[0,1]} F(T,q')\, dq'.$$

To prove the lemma let's number the patches of $T$ patches in an order of increasing $q_i$, so that $q_i \le q_{i+1}$, and set $q_0 = 0$ and $q_{n+1} = 1$ (this can be done without any loss of generality). Thus, the integral of $F$, which is a piecewise constant function, can be broken down to small constant intervals:

$$\int_{[0,1]} F(T,q')\, dq' = \int_{[0,1]} \sum_{i=1}^{n} a_i \, I(q_i \ge q')\, dq' = \sum_{j=0}^{n+1} \int_{[q_j, q_{j+1}]} \sum_{i=1}^{n} a_i \, I(q_i \ge q')\, dq' =$$

$$= \sum_{j=0}^{n+1} \sum_{i=1}^{n} a_i \, I(q_i \ge q_j)(q_{j+1} - q_j).$$

Switching the order of summations this further boils down to the natural aggregation formula:

$$= \sum_{i=1}^{n} \sum_{j=0}^{n+1} a_i \, I(q_i \ge q_j)(q_{j+1} - q_j) = \sum_{i=1}^{n} a_i \sum_{j=0}^{n+1} I(q_i \ge q_j)(q_{j+1} - q_j) =$$

$$= \sum_{i=1}^{n} a_i \sum_{j=0}^{i-1} (q_{j+1} - q_j) = \sum_{i=1}^{n} a_i \, (q_i - q_0) = \sum_{i=1}^{n} a_i \, q_i.$$

Now, if there is an intrinsic order between landscapes A and B, i.e.

$$\forall q' \in [0,1] \mid F(A,q') \le F(B,q'),$$

then

$$NCI(A) = \int_{[0,1]} F(A,q')\, dq' \leq \int_{[0,1]} F(B,q')\, dq' = NCI(B)$$

according to the basic properties of the integral. ■