The relevance of habitat quality for biodiversity and ecosystem service policies

Czúcz Bálint[1], Arany Ildikó[1], Kertész Miklós[1], Horváth Ferenc[1], Báldi András[1], Zlinszky András[2], Aszalós Réka[1]

1 Institute of Ecology and Botany, Centre for Ecological Research, Hungarian Academy of Sciences, Klebelsberg u. 3, 8237 Tihany, Hungary

2 Balaton Limnological Institute, Centre for Ecological Research, Hungarian Academy of Sciences, Alkotmány u. 2-4, 2163 Vácrátót, Hungary

Email: balint czucz@okologia.mta.hu

What is habitat quality?

Habitat quality is an inherently abstract concept which tries to summarize the "goodness" of an ecosystem in terms of its deviation from an ideal reference state. This property is generally considered to be related to the long-term functionality and self-organizing capacity of ecosystems, including their capacity to supply ecosystem services. There are several alternative definitions for similar concepts, most of which are based on the structure, composition, and type-specific key processes (Noss 1990) of the local ecosystems. Notable examples include 'vegetation condition' (Gibbons et al. 2006, 2008), naturalness (e.g. Machado 2004), hemeroby (e.g. Sukopp et al. 1990), ecosystem health (e.g. Costanza et al. 1992) and ecological integrity (e.g. Woodley et al. 1993). To measure this property of ecosystems, spatial information is needed. Most generally, habitat quality can be estimated (1) in the field by comparing observations to a standardized list of criteria (e.g. Machado 2004; Molnár et al. 2007), (2) based on field-calibrated modeling and/or remote sensing data (e.g. Li and Kräuchi 2004; Cohen et al. 2005; Gibbons et al. 2008), or (3) constructed as an aggregated index based on several field-observed and/or remotely sensed components (e.g. Bartha 2004; Gibbons and Freudenberger 2006; Standovár et al. 2006).

Habitat quality in global and EU biodiversity policies

Habitat quality is one particular way to provide a key message on the state of the ecosystems, which focusses on the general tendencies with no particular emphasis on any predefined groups of species. Such indicators, describing current state and tendencies of the studied ecosystems, are often called 'biodiversity indicators', as although they might not correspond directly to any of the generally interpreted components of biodiversity, but they can be used as surrogates to reflect general relationships and tendencies (ten Brink 2006). Habitat quality metrics can enter at several points into mainstream policy discussions.

Probably the most important policy initiative at the global level is the recently founded Intergovernmental Platform for Biodiversity and Ecosystem Services (IPBES). One of the primary goals of IPBES is to deliver global, regional and thematic assessments on the status and trends of biodiversity and ecosystem services. Species occurrence data exist only for a tiny proportion of the 3 million species known to date, many of which decline partly due to insufficient data and knowledge to counteract negative trends (Pimm et al. 2014). Habitat quality, however, can be an effective proxy for the occurrence and abundance of many species and for the integrity of ecosystems, due to the technical possibility to gain detailed data from large geographic areas, many of which largely unexplored locally. The key role of habitat quality in IPBES is also underlined by the fact that one of the thematic assessments of the platform will be focusing on land degradation and restoration.

Information about habitat quality also plays a key role in several policies in the EU policy arena:

- Measuring the status of ecosystems: The Habitats Directive (Article 17) and the Water Framework Directive require a periodic reporting on the ecological status of major terrestrial ecosystems and water bodies, which is assumed to rely on a detailed evaluation of a vast number of species at a high number of locations. Nevertheless, these data are difficult to be used for compiling habitat quality evaluations, which is caused by deficiencies of the reporting process (e.g. just one data sheet for each Natura 2000 species and habitat type for each country and biogeographical region)
- Ecosystem service indicators: According to Action 5 of EU's Biodiversity Strategy to 2020, each member state is required to map and assess the ecosystems and their services within their territory. As the most accepted conceptual model for ecosystem service delivery, the cascade model (Haines-Young and Potschin 2010, de Groot et al 2010) contains the state of ecosystems as a fundamental element, theoretically all ecosystem service maps and assessments should also embrace habitat quality maps.
- Restoration prioritization support: The 2020 Biodiversity Strategy (Target 2) prescribes that each MS should restore 15% of the degraded ecosystems within its territory. A similar 15% goal is also declared at a global level by the Convention on Biological Diversity (Aichi Target 15). According to an EU-funded study (Lammerant et al 2013) this process should be coordinated by classifying all natural and man-made ecosystems into to a 4-grade ordinal scale based on their level of degradation. Measuring and monitoring habitat quality for many ecosystems can have a great practical importance for this kind of use.
- No net loss of ecosystems and their services: Action 7 of the EU Biodiversity Strategy proposes "an initiative to ensure there is no net loss of ecosystems and their services (e.g. through compensation or offsetting schemes)". Under this target, the Commission seeks innovative new mechanisms, providing systematic tools for compensation for damages to biodiversity in the wider countryside outside Natura 2000 sites. This has to be applied in the context of the mitigation hierarchy (with an order of priority favoring avoidance and reduction of adverse impacts to the use of offsets or compensation), for which measurement of habitat quality is a key indicator from planning to implementation and monitoring of offsetting measures (Rayment 2013).

Considerations for reliable habitat quality metrics

As discussed above, a reliable and informative habitat quality metric should reflect deviation from an ideal state, which should be interpretable in a consistent way for all typical pathways of degradation. This is not easy, not even conceptually. For example, a forest can be too even aged, infected by invasive alien species, can lack a shrub layer, old trees, dead trees, gaps, etc. Furthermore, it is not always easy to identify a single unambiguous reference state, particularly for strongly transformed ecosystems. How can we define an ideal reference state for a cropland or a city? Should it be the last pristine vegetation, which used to be there at the same location prior to the transformations? Or some sort of ideal cropland, or ideal city? Even in the case of well-known semi-natural ecosystems, as European oak forests, an appropriate reference state can be difficult to determine simply because of the constant human presence since the ice age.

Being inherently local and spatially explicit, habitat quality metrics can never allow for structures and processes observable at spatial scales broader than their resolution. Thus habitat quality cannot account for landscape pattern and /or diversity. Furthermore, local habitat quality cannot generally capture outstanding natural values, like the presence of a specific rare species, or unique compositional, structural or historical features.

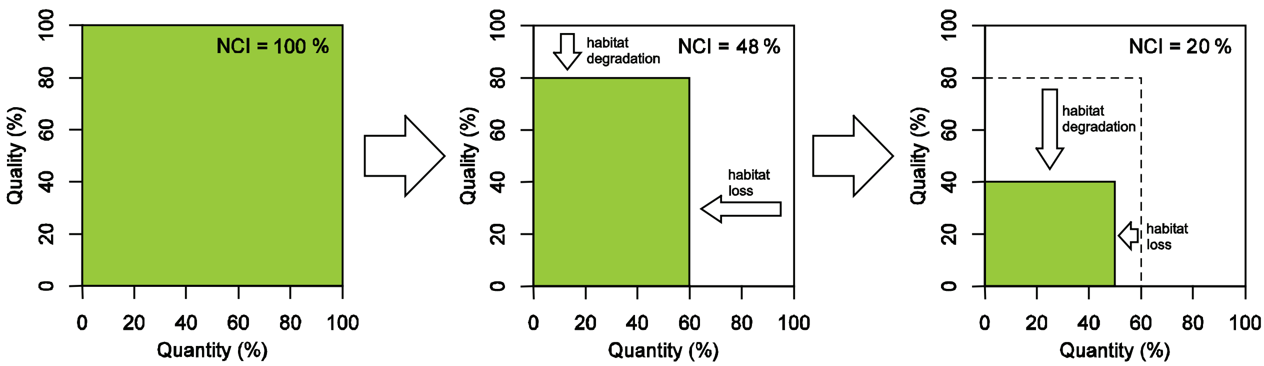

Figure 1: Natural capital is defined as the product of remaining ecosystem size (quantity) and its quality. For example, if the remaining ecosystem size is 50 %, and its quality is 40 %, then 20 % of the natural capital remains (from Czúcz et al. 2012).

Aggregating habitat quality for landscapes – the case of the Hungarian Natural Capital Index

Local habitat quality values can be very useful by themselves, nevertheless, there are several policy contexts, where an overview of larger areas is necessary. To this end habitat quality values can be aggregated in a standardized way so that they could be used effectively in evaluating and comparing ecological state in larger and smaller areas. The simplest and most straightforward aggregation scheme is the Natural Capital Index (NCI) formula (ten Brink 2000, Czúcz et al. 2008, 2012; Figure 1), which is the area weighted mean naturalness of the landscape:

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

for a landscape consisting of n homogeneous patches of size a_i and quality q_i. If both size and quality are scaled between 0 and 1 (relative to the entire landscape and the pristine reference state), then an NCI value of 1 will mean a landscape in its original, pristine or undegraded state. 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 (Figure 1).

It is apparent from the definition and the methods of calculations that NCI is flexible enough to give evaluations of landscapes at various scales. An important and advantageous property of this metric is that it can be used for quick and superficial comparisons, as well as extensive and detailed evaluations. NCI values for larger areas can namely be disaggregated in various ways into the sum of different components:

- Thematic disaggregation: the contribution of specific ecosystem types to the overall NCI value of a larger region can be easily estimated in a straightforward way. To visualize the contributions of specific ecosystem types to an overall NCI value, habitat-profile diagrams can be constructed (Figure 2).
- Spatial disaggregation: the NCI value of a larger region corresponds by definition to the area-weighted average of NCI values of its sub-regions, no matter how the sub-regions are delineated. This rule can help to identify the specific contributions of any area of interest to the NCI of the larger region.

The evaluation of the contributions of different subregions and ecosystem-types can bring new perspectives for policy applications. Flexible disaggregation makes it possible that it is not only the factual numerical values, but also the underlying causes and patterns that can be surveyed in a decision-making process. Consequently, this standardized metric can be used successfully in local and regional policy-relevant decision-making and in environmental communication.

This index is also especially suited for remote sensing: accurate data on spatial extent of each habitat and the naturalness of each study unit (e.g. pixel) can easily be integrated into this model for quantitative evaluation of natural capital at regional or local scale. In addition, the quantification of habitat quality in terms of deviation from a reference state (a “perfect” ecosystem) can also be relatively well followed up with remote sensing and GIS as long as the reference state of the ecosystem also exists and has been covered by the survey. Differences in spectral properties, spatial structure, patch diversity can all be calculated between the reference and the units of the study area.

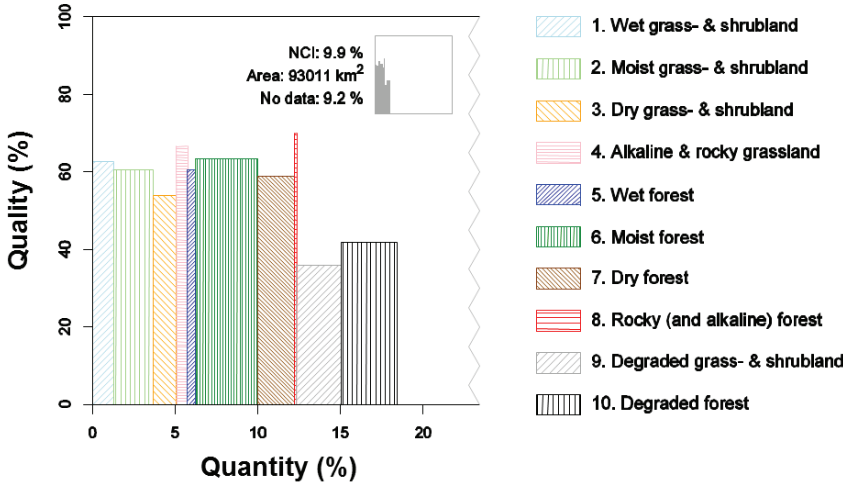

Figure 2: The Natural Capital Index of Hungary, shown in a disaggregated structure identifying contributions of 10 main habitat groups. To add perspicuity to the NCI components, the scaling of the axes is not identical, to provide a visual overview of the magnitudes, a pictogram with identically scaled axes is shown in the upper right corner (from Czúcz et al 2012).

Conclusions

While habitat quality is an abstract concept that can be defined in many different ways, there is a clear demand for reliable and transferable definitions that support quantitative analysis. International biodiversity policy, spearheaded by the IPBES targets, and EU commitments such as the 2020 Biodiversity Strategy all require spatially explicit assessments of habitat quality.

We propose the Natural Capital Index, which is based on deviation from a hypothetical reference state in terms of both area and quality of habitats. While some problems remain, this index is compatible with remote sensing and GIS analysis, and could pave the way towards more reliable habitat assessments in EU and global policy

References

Bartha, D (2004). Die Naturnähe der Wälder – Bewertung auf Bestandesebene. *Allgemeine Forst und Jagdzeitung, 175(1-2)*, 8–13.

ten Brink, B. (2000). *Biodiversity indicators for the OECD Environmental Outlook and strategy – A feasibility study.* RIVM Report 402001014. Globio Report Series No 25.

ten Brink, B. (2006). *Indicators as communication tools: an evolution towards composite indicators.* ALTER-Net research report WPR2-2006-D3b.

ten Brink, B. (2007). *The Natural Capital Index framework (NCI).* Contribution to Beyond GDP Virtual Indicator Expo, Brussels, 19-20. November 2007.

Cohen, M. J., Lane, C. R., Reiss, K. C., Surdick, J. A., Bardi, E. & Brown, M. T. (2005). Vegetation based classification trees for rapid assessment of isolated wetland condition. *Ecological Indicators, 5,* 189–206.

Costanza, R., Norton B. G. & Haskell, B. D. (1992). *Ecosystem health.* Island Press, Washington DC.

Czúcz B., MolnárZ, Horváth F, Nagy G G, Botta-Dukát Z, Török K (2012): Using the natural capital index framework as a scalable aggregation methodology for local and regional biodiversity indicators. *Journal for Nature Conservation,* 20: 144-152.

Czúcz, B., Molnár, Z., Horváth, F. & Botta-Dukát, Z. (2008). The Natural Capital Index of Hungary. *Acta Botanica Hungarica, 50(Suppl),* 161–177.

Gibbons, P. & Freudenberger, D. (2006). An overview of methods used to assess vegetation condition at the scale of the site. Ecological Management and Restoration, 7, S10-S17.2007. Expert Panel-based Assessment of Forest Landscapes for Land Use Planning. *Mountain Research and Development, 27*, 220–223.

Gibbons, P., Briggs, S., Ayers, D. A., Doyle, S., Seddon, J., McElhinny, C., Jones, N., Sims, R. & Doody, J. S. (2008). Rapidly quantifying reference conditions in modified landscapes. *Biological Conservation, 141,* 2483–2493

Gibbons, P., Zerger, A., Jones, S. & Ryan, P. (2006). Mapping vegetation condition in the context of biodiversity conservation. *Ecological Management and Restoration, 7(Suppl),* 1–2.

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.

Haines-Young R & Potschin M (2010). The links between biodiversity, ecosystem services and human well-being. In: Raffaelli, D.G. and Frid, C.L.J., eds., *Ecosystem ecology: a new synthesis*. Cambridge University Press. 110-139

Lammerant J, Peters R, Snethlage M, Delbaere B, Dickie I, Whiteley G. (2013) Implementation of 2020 EU Biodiversity Strategy: Priorities for the restoration of ecosystems and their services in the EU. Report to the European Commission. ARCADIS (in cooperation with ECNC and Eftec)

Li, M. H. & Kräuchi, N. (2004). *Using a combined index of native and non-native plant diversity for estimating ecosystem and environmental change over time and space.* Bridging Scales and Epistemologies Conference, Millennium Ecosystem Assessment, Alexandria, Egypt (March 17–20, 2004).

Machado, A. (2004). An index of naturalness. *Journal for Nature Conservation, 12,* 95–110.

Molnár, Z., Bartha, S., Seregélyes, T., Illyés, E., Botta-Dukát, Z., Tímár, G., Horváth, F., Révész, A., Kun, A., Bölöni, J., Biró, M., Bodonczi, L., Deák, J. Á., Fogarasi, P., Horváth, A., Isépy, I., Karas, L., Kecskés, F., Molnár, C., Ortmann-Ajkai, A. & Rév, S. (2007). A GRID-Based, Satellite-Image Supported, Multi-Attributed Vegetation Mapping Method (MÉTA). *Folia Geobotanica, 42*, 225–247.

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

Pimm, S. L., Jenkins, C. N., Abell, R., Brooks, T. M., Gittleman, J. L., Joppa, L. N., … Sexton, J. O. (2014). The biodiversity of species and their rates of extinction, distribution, and protection. *Science*, 344(6187), 1246752.

Rayment, M. (2013). Exploring potential demand for and supply of habitat banking in the EU and appropriate design elements for a habitat banking scheme. Executive Summary. http://ec.europa.eu/environment/enveco/taxation/pdf/Habitat_banking_Report.pdf

Standovár, T., Ódor, P., Aszalós, R. & Gálhidy, L. (2006). Sensitivity of ground layer vegetation diversity descriptors in indicating forest naturalness. *Community Ecology*, *7*, 199–209.

Sukopp, H., Hejný, S. & Kovarik, I. (Eds.) (1990). *Urban ecology – Plants and plant communities in urban environments.* SPA Academic Publications, The Hague, The Netherlands.

Woodley, S., Francis, G. & Kay, J. (1993). Ecological integrity and the management of ecosystems. Boca Raton, FL, USA: St Lucie Press.