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COMMENTARY

Separating risks from values in setting priorities for plant community conservation

David A. Keith

Keith, D.A. (david.keith@unsw.edu.au): Centre for Ecosystem Science, University of NSW, Sydney, NSW 2052, Australia
and NSW Office of Environment and Heritage, Hurstville, NSW, Australia

Abstract

Berg and colleagues, in this issue, describe a framework for assessing risks to biodiversity and setting conservation priorities in northeast Germany. Their method explicitly separates community endangerment from conservation value, and derives its plant communities from a sound regional classification. It could be improved by incorporating ecological processes into risk assessment, and socio-political constraints, economic costs and the likelihood of success into priority setting.

Information about trends and risks to biodiversity is fundamental to well-justified and cost-effective conservation action. Red lists of species have helped meet this need for decades, and have motivated development of analogous tools for higher levels of biodiversity, ranking of ecologically-defined units, such as ecosystems, ecological communities and habitat types.

Nicholson et al. (2009) reviewed a sample of 12 protocols designed for this purpose, of which few were published in scientific literature. They found that all protocols included criteria to quantitatively assess spatial extent and trends in distribution, although implementation details differed. The common challenges identified include: embedding the method in an explicit theoretical framework, controlling for variations in thematic and spatial scales, quantifying the contribution of declines in ecological function to risks, and justifying the thresholds that delineate different levels of threat. Many existing protocols are implicitly local in focus to suit patterns of ecological variation, social and scientific settings and perceived information needs within particular countries or sub-national jurisdictions.

Berg et al. (2014) propose a protocol to assess the status of plant communities and set priorities for conservation in northeast Germany. Their scheme explicitly separates risk assessment from valuation and priority setting. This is a clear strength, avoiding well-recognized problems stemming from conflation of scientific and value-based components of assessment and decision making (Rodriguez et al. 2011).

Another strength of their framework is that their assessment units derive from a quantitative and comprehensive regional classification of plant communities. A relatively fixed level of classification allows spatial thresholds delimiting the threat categories to be set commensurately, thus minimizing scale-related artifacts of the assessment process (Nicholson et al. 2009). A standard map base of plant community types would further reduce these effects. Although Berg et al. acknowledge limitations and uncertainties in the classification and needs for further development, its derivation from a large set of relevés ensures a degree of realism in representing on-ground variation in biodiversity. A future challenge will be to incorporate this uncertainty explicitly into the risk assessment and prioritization process. Emerging methods for representing uncertainty in classifications (Pillar 1999; Duff et al. 2014) offer great promise for this process.

Berg et al. propose a risk assessment based on three criteria pertaining to the past, present and future spatial extents of plant communities. These time scales were set pragmatically considering availability of data and 'manageability', with assessment of future trends limited to 10 yrs. Although this limits scope for prediction error, it precludes evaluation of slower-acting threats such as climate change, which have established and rapidly improving methods for projection over longer time scales (Hannah 2012).

The reliance on spatial criteria to the exclusion of an explicit evaluation of ecological processes is a conscious and, in my view, serious and unconvincing omission. Berg et al. point to the difficulties of defining when a plant community has vanished due to gradual change. They suggest that processes such as floristic impoverishment and physiognomic change are more easily assessed as transitions between types in a "sufficiently fine typology." The drawback to this approach is that ecological changes associated with declines and losses of biodiversity may not be expressed as detectable transitions between recognized assemblages until a late (and perhaps irreversible) stage.

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The capacity to track and re-map distributions of fine-scale community types at regular intervals must surely be limited, especially if fine distinctions between units are undetectable with remote methods.

Although challenging, flexibility to assess a wide range of ecological processes would be a welcome extension to the proposed framework, increasing its sensitivity to detect biodiversity declines and enhancing its value as an early warning system (e.g. Keith et al. 2013). Furthermore, this would make better use of the developing compositional time series in relevé databases, as well as Europe's major body of work on vegetation dynamics, restoration ecology, plant–environment–disturbance relationships, species migration and climate change responses. It is better to incorporate these data where they are available than exclude them because they are not available for all community types. We have shown elsewhere that such an approach is workable, even in data-poor parts of the world (Keith et al. 2013).

An interesting structural aspect of the Berg et al. framework is its factorial method of aggregating information from different components into ordinal categories of threat, value and priority. This minimizes loss of information, especially where there may be interactions between the components. Thus, for example, communities with highly restricted current distributions are categorized as 'very rare but not currently threatened', so long as rates of past and future decline are slow. An aggregation method based on averaged scores would have difficulty in discriminating such communities.

The normative component of the framework includes subjective valuation criteria based on representation of threatened (plant) species, subjective ranks of naturalness, and proportion of global range represented in the study area. Other valuation criteria could be added, depending on the socio-political setting of conservation objectives. Examples include phytosociological uniqueness (dissimilarity to other communities), contributions to ecosystem services and cultural values.

The 'need for action' on each community is derived by factorially combining its threat and value categories. Some further steps are needed to turn 'need for action' outcomes into practical priorities for allocating conservation resources. Two critical factors are often overlooked in prioritization approaches: the cost of implementing the action, and the likelihood that the action will succeed (Joseph et al. 2009). Ignoring these can lead to ineffective resource use and avoidable losses of biodiversity. Simple objective methods exist to incorporate risk, value, cost and likelihood of success into a single prioritization process (Joseph et al. 2009). Prioritizing actions, rather than objects, thus can avoid loss of opportunities to apply cheap preventative actions to 'low priority' communities, and also design and assess the worth of actions that potentially benefit multiple communities.

Finally, Berg et al. evaluate the performance of their framework by considering whether it produces 'meaningful' ranks. Such intuitive evaluations are a useful first step to identifying obvious discrepancies. A more searching evaluation would extend to independent comparisons between predicted and observed outcomes, either through simulation or monitoring, to determine whether communities ranked at highest risk decline and vanish at the fastest rates. The general applicability of the framework will ultimately be determined by its uptake in other parts of Central Europe and beyond.

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