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POLICY PERSPECTIVE

The IUCN Red List of Ecosystems: Motivations, Challenges, and Applications

David A. Keith[1,2,3,4,5], Jon Paul Rodríguez[4,5,6,7], Thomas M. Brooks[8], Mark A. Burgman[9], Edmund G. Barrow[10], Lucie Bland[11], Patrick J. Comer[12], Janet Franklin[13], Jason Link[14], Michael A. McCarthy[11], Rebecca M. Miller[15], Nicholas J. Murray[1], Jeanne Nel[16], Emily Nicholson[11,19,21], María A. Oliveira-Miranda[7], Tracey J. Regan[11,20], Kathryn M. Rodríguez-Clark[3], Mathieu Rouget[17], & Mark D. Spalding[18]

[1] Centre for Ecosystem Science, University of New South Wales, Sydney, New South Wales, Australia
[2] New South Wales Office of Environment and Heritage, Hurstville, New South Wales, Australia
[3] Long Term Ecological Research Network, Terrestrial Ecosystem Research Network, Australian National University, Canberra, Australian Capital Territory, Australia
[4] IUCN Commission on Ecosystem Management, Gland Switzerland
[5] IUCN Species Survival Commission, Gland, Switzerland
[6] Centro de Ecología, Instituto Venezolano de Investigaciones Científicas, Caracas, Venezuela
[7] Provita, Caracas, Venezuela
[8] IUCN Science and Knowledge Unit, Gland, Switzerland
[9] Centre of Excellence for Biosecurity Risk Analysis, School of Botany, The University of Melbourne, Victoria, Australia
[10] IUCN Global Ecosystem Management Programme, Nairobi, Kenya
[11] ARC Centre of Excellence for Environmental Decisions, School of Botany, The University of Melbourne, Victoria, Australia
[12] NatureServe, Boulder, Colorado, USA
[13] School of Geographical Sciences & Urban Planning, Arizona State University, Tempe, Arizona, USA
[14] NOAA Fisheries, Woods Hole, Massachusetts, USA
[15] IUCN Global Ecosystem Management Programme, Cambridge, United Kingdom
[16] Biodiversity & Ecosystem Services, Natural Resources & the Environment, CSIR, South Africa
[17] Land Use Planning and Management, School of Agricultural, Earth and Environmental Sciences, University of KwaZulu, Natal, South Africa
[18] The Nature Conservancy and Conservation Science Group, Department of Zoology, University of Cambridge, Cambridge, England
[19] School of Life and Environmental Sciences, Deakin University, Burwood, Victoria, Australia
[20] Arthur Rylah Institute for Environmental Research, Department of Environment, Land, Water and Planning, Heidelberg, Victoria, Australia
[21] Centre for Integrative Ecology, School of Life and Environmental Sciences, Deakin University, Burwood, Victoria 3125, Australia

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Correspondence
David Keith, Centre for Ecosystem Science, University of NSW, Kensington Sydney New South Wales Australia, 2052.
Tel: 612- 9585 6498; Fax: 612- 9585 6606.
E-mail: David.Keith@unsw.edu.au

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Abstract

In response to growing demand for ecosystem-level risk assessment in biodiversity conservation, and rapid proliferation of locally tailored protocols, the IUCN recently endorsed new Red List criteria as a global standard for ecosystem risk assessment. Four qualities were sought in the design of the IUCN criteria: generality; precision; realism; and simplicity. Drawing from extensive global consultation, we explore trade-offs among these qualities when dealing with key challenges, including ecosystem classification, measuring ecosystem dynamics, degradation and collapse, and setting decision thresholds to delimit ordinal categories of threat. Experience from countries with national lists of threatened ecosystems demonstrates well-balanced trade-offs in current and potential applications of Red Lists of Ecosystems in legislation, policy, environmental management and education. The IUCN Red List of Ecosystems should be judged by whether it achieves conservation ends and improves natural resource management, whether its limitations are outweighed by its benefits, and whether it performs better than alternative methods. Future development of the Red List of Ecosystems will benefit from the history of the Red List of Threatened Species which was trialed and adjusted iteratively over 50 years from rudimentary beginnings. We anticipate the Red List of Ecosystems will promote policy focus on conservation outcomes in situ across whole landscapes and seascapes.

Introduction

Policy imperatives to assess risks to biodiversity above the species level are underscored by IUCN's recent endorsement of new criteria and categories for Red Lists of Ecosystems (RLE) as a global standard for ecosystem risk assessment, and the inclusion of the RLE in the 2014 Horizon Scan of environmental issues (Sutherland *et al.* 2014). The RLE method is based on five quantitative criteria (Appendix 1) designed to evaluate symptoms of risk in terrestrial, subterranean, freshwater, and marine ecosystems (Keith *et al.* 2013). What are the practical challenges in applying such a tool in environmental policy and management for biodiversity conservation, and how can they be met?

In this perspective, our central purpose is to review the major challenges in developing, interpreting and applying the RLE method and consider trade-offs inherent in the design of solutions. To support our discussion, we first elucidate the motivations and goals for Red Listing ecosystems and later identify current and potential applications of the risk assessment products. Overall, we suggest that the IUCN Red List of Ecosystems should be judged by whether: it achieves conservation ends and improves environmental management; its limitations are acceptable given its benefits; and it outperforms alternative methods for similar tasks. We draw affirmative evidence from an extensive global consultation process including scientific workshops, stakeholder meetings, reviews, and published case studies (Nicholson *et al.* 2009; Keith 2009; Rodriguez *et al.* 2011, 2012; Keith *et al.* 2013; IUCN 2014a; Nicholson *et al.* 2014). We also recommend ongoing performance evaluations to support users who need to make judgements about RLE applications.

Motivations, goals, and trade-offs in ecosystem risk assessment

An IUCN tool to assess ecosystems has a sound conservation justification (Boitani *et al.* 2014). There are several well established imperatives to address multiple levels of biodiversity in conservation planning (Noss 1996). First, broad-scale features such as ecosystems identify major assemblages of biota requiring protection across landscapes and seascapes, and serve as surrogates for poorly known fine-scale features (Margules & Pressey 2000). Simultaneous evaluation of broad- and fine-scale features helps to ensure that key elements such as threatened species are not overlooked in planning decisions. Evaluation at both scales is more likely to achieve "comprehensive," "adequate," and "representative" (CAR) conservation outcomes (Margules & Pressey 2000).

Second, ecosystem-level assessment can address ecological processes, such as interactions among populations and with abiotic components of landscapes and seascapes, which are generally excluded by taxon-level assessments (Sabo 2008). Ecological processes are crucial in diagnosing threats to individual species and resolving potential management conflicts for coexisting species, and hence underpin the "adequacy" of conservation decisions.

Third, most ecosystem functions rely on common species (Gaston & Fuller 2007). These are central to description and risk assessment of ecosystems (Keith *et al.* 2013), but important changes in common species rarely feature in threatened species analysis.

Finally, people value and relate to ecosystems and places, even though they may not recognize many component species (Dallimer *et al.* 2012; Wunder *et al.* 2014). Human well-being depends on ecosystems for many ecological services (Costanza *et al.* 2014). These qualities make ecosystem-level assessments useful communication and education tools to support biodiversity conservation, macroeconomic planning and sustainable management of land and water.

These imperatives for ecosystem-level approaches to conservation underpin the central goal of the RLE criteria, which is to support conservation in resource use and management decisions by identifying ecosystems most at risk of biodiversity loss (Keith *et al.* 2013). To meet this goal, a balance of four qualities was sought in the design of the RLE method: generality, precision, realism, and simplicity (building on Levins 1966). Generality was seen as particularly important to ensure that the RLE criteria could be applied to terrestrial, subterranean, freshwater and marine ecosystem types, and sufficiently flexible to handle data of varying quality and detail (Keith *et al.* 2013). Precision promotes consistency, transparency, and repeatability in applications across assessors, regions and ecosystem types, and was a strong motivation for adopting quantitative listing criteria (Keith *et al.* 2013). Realism underpins reliable and accurate scientific assessments, and is enhanced in the RLE listing criteria by the ability to use direct ecosystem-specific diagnostic variables to assess ecological processes (Keith *et al.* 2013). We added simplicity to Levins' framework (after Einstein 1934) because it enhances clarity of purpose, accessibility to users and communication of resulting conservation messages. The direct focus of RLE criteria on symptoms of biodiversity loss contributes to simplicity by decoupling the risk assessment from priority setting and valuation of ecosystem services (Keith 2014). We further discuss relationships between biodiversity, ecosystem functions, and services below.

The four qualities sought in designing the RLE involve unavoidable trade-offs that are well known in model development across diverse disciplines (Levins 1966). For example, Red List criteria for both species and ecosystems sacrifice some precision (hence consistency) to achieve generality across species with different life histories, and ecosystems with different governing processes. The trade-offs and appropriate balance will depend on the goals and scope of risk assessment. When the scope is broad, as is the case with Red Lists, emphasis inevitably falls on generality with some trade-offs on other desired qualities. The issue is not whether imprecision and inconsistency exist (Boitani *et al.* 2014), but whether improving precision is worth the trade-offs in generality, realism, and simplicity. Hobday *et al.* (2011), in their risk assessment protocol for managed fisheries, use a hierarchical approach to manage such trade-offs; higher levels of assessment achieve more precise estimates of risk than lower levels, but at the cost of additional complexity and reduced generality. This approach may prove useful for Red Listing in cases where available data limit the certainty of assessment outcomes. Investments in more detailed risk assessments may be warranted when low-level assessments suggest risks may be high (Hobday *et al.* 2011).

Challenges and solutions in RLE design and application

Ecosystem types as assessment units

The RLE protocol defines ecosystems as units of assessment that represent complexes of organisms and their associated physical environment within an area (Keith *et al.* 2013). This definition incorporates four elements—biotic and abiotic complexes, interactions, and spatial location (after Tansley 1935, pp. 299-300)—to establish a framework for describing assessment units. These defining features set a framework to distinguish among ecosystem types, and also identify a suite of characteristics potentially sensitive to identified hazards. In RLE assessments, they set the requirements for description of ecosystem types, are used to establish the levels of change that signal ecosystem collapse, and thus underpin the generality of the RLE protocol (Keith *et al.* 2013). While any unit that is distinguishable on the basis of these defining features meets the conceptual definition of *ecosystems,* this should not be confused with the need for appropriately scaled descriptions and classifications of *ecosystem types* for application of the RLE criteria (Boitani *et al.* 2014).

Systematic classifications based on the defining features of ecosystem types place each unit in context and promote consistent identification and use. This is very important for comprehensive global and regional risk assessments aimed to support systematic conservation planning (e.g., Lindgaard & Henrisksen 2011), but less important where risk assessments are used individually to support strategic ecosystem management (Keith 2015). Two aspects of ecosystem typologies need to be addressed: (1) the classification method and resulting units; and (2) the extent to which these ecosystem types can be mapped. The latter is critical for assessment of RLE criteria A and B (Appendix 1), which address rates of change in distribution, and whether the extent is restricted and declining. A lack of spatial data for variables used in ecosystem classification can limit or preclude mapping of some ecosystem types (Mackey *et al.* 2008).

The development of suitable ecosystem typologies (or adaptation of existing ones) to support RLE assessments is challenging partly because biotic and abiotic features vary continuously (not discretely), and partly because the patterns of variation are scale-dependent (Keith 2009). A suitable typology would also need to be hierarchical or nested, allowing ecosystems assessed at national or finer scales to be related transparently to those applied in global-scale assessments. Two main lines of evidence suggest that these challenges can be met in workable applications of the RLE method.

First, risk assessments of ecosystem-type units have already been applied successfully at national and regional levels over several decades in Europe, the Americas, Australia, and Africa (Table 1). Systematic typologies support most assessments, but in Australia listings of ecosystem types (locally termed "ecological communities") derive from public nominations and each nominated unit is critically evaluated and adapted (where necessary to ensure robustness) by statutory committees of scientists as part of the assessment process (Keith 2009). Experience from this process shows that, even without a formal typology, red-listed ecosystem types can be reliably interpreted by consultants and agency staff and can withstand adversarial legal challenge, so long as their defining features are adequately described (Preston & Adam 2004). National applications involve a variety of listing criteria, typologies, and nomenclature for the assessment units (Table 1 and references therein), but they demonstrate that inherent uncertainties in unit delimitation can be managed and pose no barrier to productive risk assessment or application of its outcomes to real-world conservation problems.

Second, the RLE protocol can learn from the Red List of Threatened Species (RLTS) when dealing with uncertainties in classification of assessment units. Mace (2004), on species conservation, suggests that "Units for listing need to be consistent, change rarely and can be somewhat arbitrary." Yet even for species and extinction, the boundaries of taxa and the understanding of decline depend on context and on indeterminate, dynamic and

Table 1 Current legislative, policy, and reporting applications of ecosystem types and similar units

Jurisdiction	Application	Assessment unit	Goal and application	Reference
European Union	Habitats Directive 92/43/EEC (European Commission)	Habitat type. "Plant and animal communities as the characterizing elements of the biotic environment, together with abiotic factors operating together at a particular scale."	To maintain and restore, at favorable conservation status, natural habitats. Listed habitat types protected and the status of protected areas reported under Habitats Directive 1992.	Council of the European Communities (1992)
Germany	Red List of biotopes (Federal Environment Agency)	Biotope. "Habitat of a community of fauna and flora living in the wild."	To permanently safeguard biological diversity by countering threats to naturally occurring ecosystems, biotopes and species under the Federal Nature Conservation Act 2009	Riecken *et al.* (2006)
Finland	Red List of habitat types (Finnish Environment Institute)	Habitat type. "Spatially definable land or aquatic areas with characteristic environmental conditions and biota which are similar between these areas but differ from areas of other habitat types."	To provide a complete description of the current state of the habitat types found in Finland, their development during recent decades, and the threats they are likely to face in the near future.	Kontula & Raunio (2009)
Norway	Red List of ecosystems and habitat types (Norwegian Biodiversity Information Centre)	Habitat type. "A homogeneous environment, including all plant and animal life and environmental factors that operate there."	To preserve biodiversity and assess performance against national targets and international obligations. Listed "Nature types" protected under Nature Diversity Act 2009.	Lindgaard & Henrisksen (2011)
Venezuela	National Red List of ecosystems (Provita)	Major vegetation types for national assessment; satellite-derived land types for subnational assessments.	To support conservation policies and decision making.	Rodriguez *et al.* (2010)
Canada	State threatened species and ecosystems legislation (Manitoba Conservation and Water Stewardship Department)	Ecosystem. "A dynamic complex of plant, animal, and micro-organism communities and their nonliving environment interacting as a functional unit."	To conserve and protect endangered and threatened ecosystems in the province and promote the recovery of those ecosystems under the Endangered Species and Ecosystems Act 2013.	Government of Manitoba (2014)
Australia	Lists of threatened ecological communities at national and state levels (Federal Department of Environment, state environment agencies)	Ecological community. "An assemblage of native species that inhabits a particular area in nature."	To conserve biodiversity and support state of environment reporting. Listed "ecological communities" protected under Australian Environment Protection and Biodiversity Conservation Act 1999, and respective state legislation (see Keith 2009 for details)	Commonwealth of Australia (2000), Keith (2009), Nicholson *et al.* (2014)
South Africa	National biodiversity legislation (South African National Biodiversity Institute)	Ecosystem. "A dynamic complex of animal, plant, and micro-organism communities and their nonliving environment interacting as a functional unit."	To provide for the protection of ecosystems that are threatened or in need of protection to ensure the maintenance of their ecological integrity under the South African Biodiversity Act 2004 and give effect to the Republic's obligations under international agreements.	Republic of South Africa (2004), Driver *et al.* (2012)

Table 2 Sources of uncertainty in species and ecosystem classifications. Tools such as interval arithmetic, fuzzy arithmetic (Akçakaya *et al.* 2000) and indices of genuine change (Butchart *et al.* 2004) help to deal with these in Red List assessments of both species and ecosystems

Source of uncertainty	Species	Ecosystems
Legacies of species and ecosystem concepts that have evolved over time	Species classifications in current use include taxa whose names and circumscriptions are the cumulative legacy of several hundred years of classification and description. Hence they are based on a mix of contrasting species concepts including pre-Darwinian morphological, biological (gene flow barriers), evolutionary, phenetic morphometric, and phylomolecular concepts.	Although numerous variations on the ecosystem concept have been proposed, many retain the four elements in Tansley's original concept, albeit with different relative weightings depending on the application. The exposure of a global RLE to legacy effects will depend on how its adopted concept of an ecosystem evolves over time.
Alphataxonomic boundary delineations are subjective and vary between experts and lineages	There is no widely applied means of standardizing or calibrating levels of genetic variability within and between species. Degrees of splitting and lumping vary subjectively between lineages, despite nomenclatural rules and peer review. This directly determines which elements are eligible or ineligible for Red Listing.	Similar difficulties apply in constructing a classification to represent consistent levels of dissimilarity between ecosystem types. This would be analytically possible if a globally representative set of samples existed. In the absence of such data, calibration will require subjective judgements.
Numbers of described units are rapidly increasing, while large numbers of units remain undescribed	Reinterpretation, new analyses, and discoveries are rapidly increasing the number of described taxa at the rate of approximately 10,000 per year (May 2004). The deficit of undescribed taxa is biased across major taxa and will be prolonged by current declines in the taxonomic workforce and alphataxonomic research activity.	Numbers of ecosystem types may also increase over time as new variants are discovered, although the proportional rate of increase is likely to be slower than for species. The deficit of undescribed ecosystems is biased spatially and by environments (e.g., deep sea cf. terrestrial ecosystems).
Boundaries delimiting related units are routinely revised	Most taxonomic revisions involve new interpretations of species boundaries. Almost none find an existing classification to be an adequate representation of variation.	Improved inventory data is likely to lead to revised circumscriptions and spatial boundaries between ecosystem types.
Unrecognized cryptic units	An unknown number of biologically isolated and genetically discrete taxa are morphologically cryptic and hence unrecognized (e.g., Mant *et al.* 2002).	An unknown number of distinctive ecosystems may exist in cryptic environments such as subterranean and deep sea realms (e.g., Ramirez-Llodra *et al.* 2010).
New technology	The molecular revolution is resulting in radical reorganization of classifications at generic level and above, and has only just begun to have an impact on species delineation. Below the species level, molecular analyses have revealed signatures of deep coalescence, sporadic introgression, unidentified sympatry, and evidence of polyphyly that require taxonomic correction (Weston 2013).	New remote sensing and environmental DNA assay technologies (e.g., Jerde *et al.* 2011) may contribute to dynamism in classification of ecosystem types referred to above.

inconsistently applied theoretical concepts (Regan *et al.* 2002). The same is true of ecosystems and their collapse (Table 2). Although species and ecosystem concepts often impose essentially arbitrary boundaries on continua (Regan *et al.* 2002), they serve to identify sets of relatively unique, ecologically and evolutionarily important components of biodiversity with which scientists, managers, and the public identify. Methods are developing to strengthen the consistency of species delimitation (Brooks & Helgen 2010). The RLTS is equipped with tools to deal with classification uncertainties (Akçakaya *et al.* 2000; Butchart *et al.* 2004), and similar tools are appropriate for the RLE. Species and ecosystems are imprecise entities, but their applications to Red Lists of species (for some 50 years) and of ecosystem types (for more than 20 years at national scales) have shown them to be useful frameworks for biodiversity assessment, conservation, macroeconomic planning, and resource use decisions (Margules & Pressey 2000; Rodrigues *et al.* 2006).

While approaches to ecological classification are currently less unified than for species taxonomy, classification and mapping of broad landscape- and seascape-scale ecosystem types is likely to be burdened by fewer historical legacies than species classification (Table 2). Recent progress towards a global classification of terrestrial vegetation (Faber-Langendoen *et al.* 2014), and other frameworks of continental and national scope, indicate that development of hierarchical ecosystem typologies is feasible, although establishing clear relationships (crosswalks) with classifications in local use will be important for RLE applications. Vegetation types are often workable proxies for ecosystem types at terrestrial landscape scales; the macrogroup or group levels of classification could be appropriate for the global RLE (Faber-Langendoen *et al.* 2014; IUCN 2014a). However, restricting the RLE to vegetation types (as suggested by Boitani *et al.* 2014) would not resolve continuum issues which have long been recognized in plant community ecology (Austin 1985). This restriction would also reduce the generality of the RLE by precluding its application to many lentic, lotic, marine, subterranean, and some terrestrial ecosystems that cannot be characterized by vegetation alone.

Conceptualizing ecosystem dynamics and collapse

Ecosystems may decline in quality as well as quantity (Noss 1996). Erosion of the functional diversity and complementarity of biota is a particular concern because these features apparently support ecosystem function and resilience (Cardinale *et al.* 2012). Interpreting which aspects of ecosystem dynamics are relevant to conservation and defining collapse in an ecosystem-specific context are challenging prerequisites for assessing risks of ecosystem collapse. Contemporary theory on ecosystem resilience (Folke *et al.* 2004) and state-and-transition frameworks for ecosystem dynamics (Hobbs & Suding 2009) support an effective trade-off between realism and simplicity for this task in risk assessment, and have also proved useful in ecosystem management, restoration and other applications (e.g., Knapp *et al.* 2011). Tozer *et al.* (2014), for example, use a state-and-transition framework to identify processes that drive transitions between different states of a woodland ecosystem, and identify the states that represent ecosystem collapse (Figure 1). This and other examples may be generalized (Figure 2) to show how ecosystem dynamics can be interpreted for RLE assessment. A key task is to identify transitions between states either as part of natural variability within an ecosystem type, or as a process of collapse and replacement by a different or novel ecosystem type (Figure 2). The outcomes of this interpretive process may be uncertain (Figure 2), and depends on whether transitions to particular states involve loss of the defining features (characteristic biota and processes) that explicitly describe the ecosystem type (Table 1 in Keith *et al.* 2013). Once explicitly identified (Figure 2), model uncertainty can be incorporated into Red List assessments (Regan *et al.* 2002). This approach makes no assumptions about Clementsian climaxes, absolute stability, linearity between risk of collapse and diagnostic variables, or individualistic dynamics (cf. Boitani *et al.* 2014).

Ecosystem stability depends on scale and context (Wiens 1989). The RLE method requires assessors to explicitly define the scale, context and characteristics of alternative states, as well as appropriate scaling functions for standardization (Rodriguez *et al.* 2015). These requirements promote transparency and robustness, allowing assessments to be critically reviewed, updated and repeated, with a level of consistency not attained by previous protocols (Nicholson *et al.* 2009). Self-evidently, "states" are not useful for risk assessment when defined at such fine scales that they are always unstable or so broadly that they always appear stable.

Ecosystem responses to climate change should be interpreted as transitions to collapsed states only if defining features and functionality are lost (Figure. 2). If many ecosystem types qualify for threatened RLE status as a consequence of identified climate-related mechanisms, it suggests that the method is sensitive to this serious threat to persistence and functionality of ecosystems and their component biota. Existing case studies show that the RLE status of several ecosystem types is influenced by identified threats attributed to global climate change, including quantified trends in cloud cover, flood regimes, fire regimes, and extreme temperatures (Keith 2015).

Indeed, one important role of the RLE is to identify which ecosystems are at greatest risk of collapse from climate change. How this translates to priorities for action will depend on socioeconomic values, costs, and likelihood of management success (Carwardine *et al.* 2008), as well as the values of the novel systems that may replace those that collapse as the climate changes. Replacement by novel ecosystems may sometimes be interpreted as desirable adaptive responses. The role of Red Listing is to report on the risk of such transitions occurring, and the separate task of developing management strategies and priorities may or may not generate remedial management actions. The interpretation of novel ecosystems driven by no-analog climates nonetheless remains a major challenge for ecosystem risk assessment. The main contribution of the RLE approach is to provide a framework for transparent reasoning and treatment of uncertainty where climate change generates competing models of ecosystem dynamics (Figure 2).

Quantifying ecosystem degradation and collapse

Stochastic process-based ecological models provide the most powerful means of quantifying the risks of ecosystem collapse because they can incorporate multiple interacting mechanisms of ecosystem decline (e.g., Lester & Fairweather 2011; Plagányi *et al.* 2014). The RLE method provides for the application of such models through

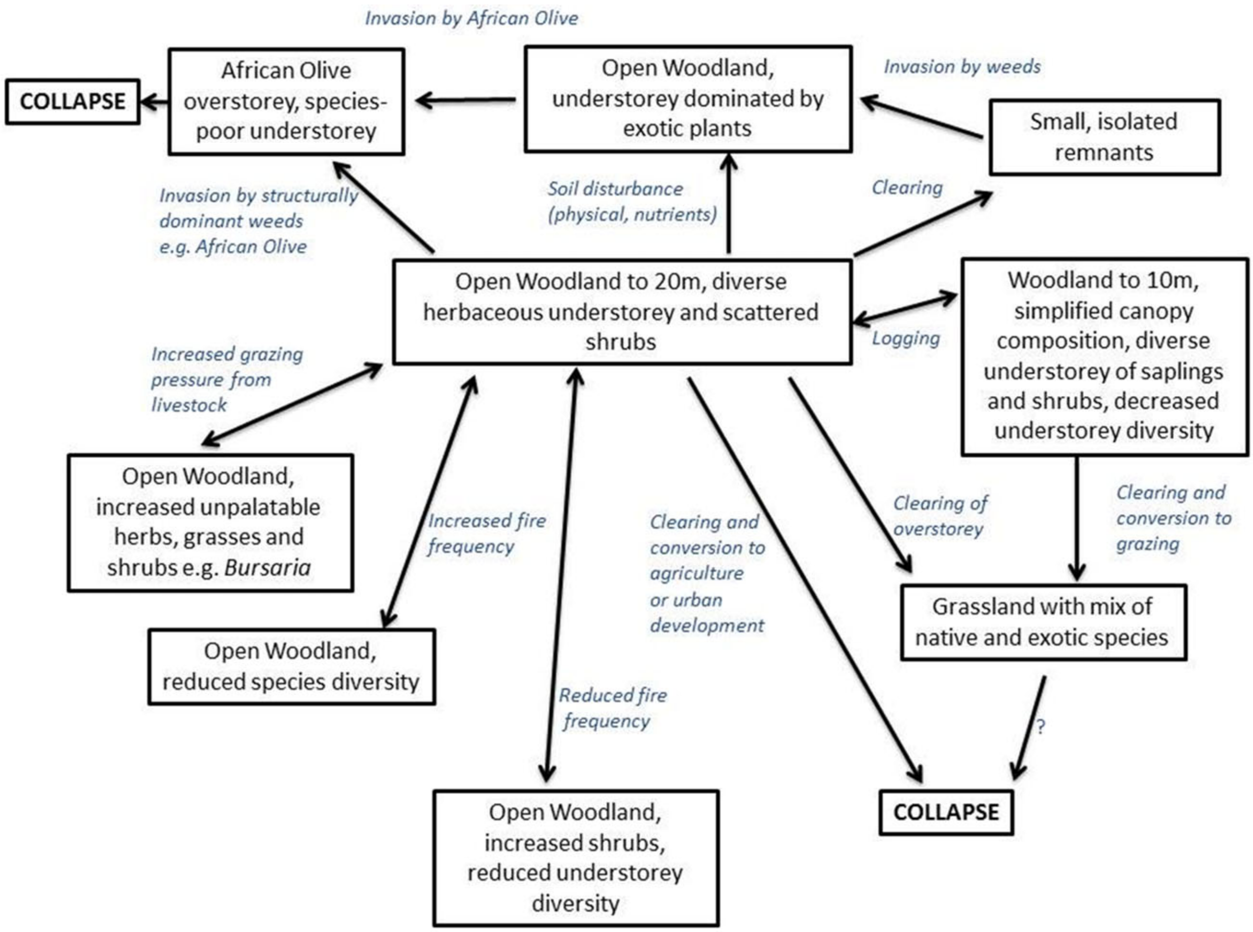

Figure 1 Example of a diagrammatic process model for a woodland ecosystem illustrating transitions between alternative states driven by different threatening processes. Collapsed states may be reached via multiple trajectories. Suitable diagnostic variables for assessing Red List criteria C and D are those that measure progression along these trajectories (from Tozer *et al.* 2014).

criterion E (Appendix 1; Keith *et al.* 2013; Burns *et al.* 2014). For many ecosystem types, however, there may be insufficient data on which to build models with adequate realism. Criteria C and D (Appendix 1) enable use of diagnostic variables that can be related directly to ecosystem degradation. Risk assessment protocols that incorporate both quantitative models and diagnostic variables provide the generality (hence flexibility) and simplicity needed to handle varying data quality and diverse symptoms of risk (IUCN 2001; Hobday *et al.* 2011; Keith *et al.* 2013). The trade-offs for realism and precision will depend on how well the diagnostic variables quantify degradation.

Four options for assessing the severity of functional decline were evaluated during RLE consultation workshops: unstructured qualitative ranking; aggregated indices of health/condition; one or a few prescribed generic ecosystem variables; and assessor-defined ecosystem-specific variables. Many existing protocols rank the severity of functional decline using unstructured qualitative methods (Nicholson *et al.* 2009). This is the least consistent, transparent and repeatable option because assessments cannot be effectively calibrated. Aggregated indices or multivariate measures of ecosystem health or condition may appear more consistent, but missing data, aggregation artefacts, threshold behavior and averaging effects reduce their sensitivity (e.g., Jennings 2005; Gorrod *et al.* 2013), and different types of ecosystems are likely to require structurally different indices (e.g., marine vs. terrestrial).

The other two options focus on particular variables correlated with functional decline. They have complementary strengths and weaknesses. The prescriptive option (as suggested by Boitani *et al.* 2014) requires estimates of trends in one or a few preset, generic

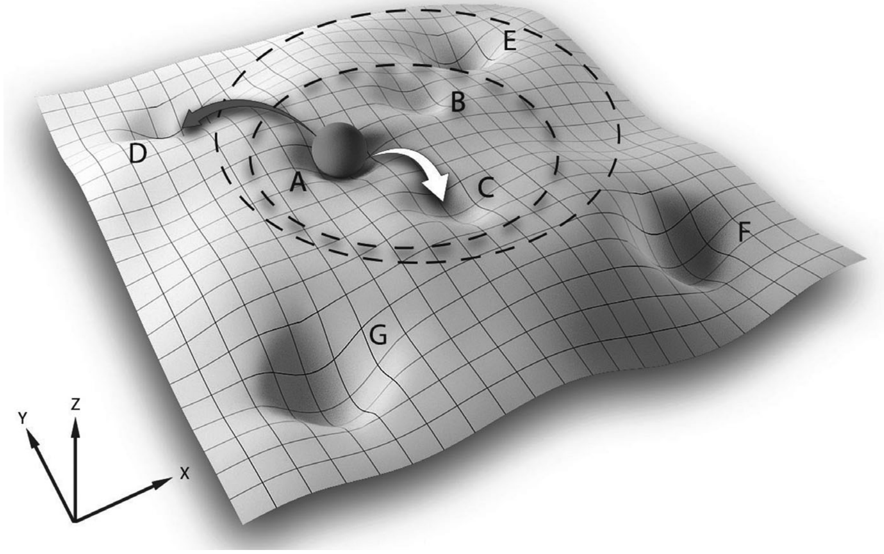

Figure 2 Generalized schematic representing a state and transition model of ecosystem dynamics (adapted from Folke *et al.* 2004). States A–G are defined by two state variables represented on the X and Y axes. The vertical axis (Z) represents potential for change. The RLE framework requires explicit description of state variables that represent defining features of the system (characteristic native biota and ecological processes) (Table 1 in Keith *et al.* 2013). The two broken lines represent alternative interpretations of ecosystem collapse. For the inner line, transitions between states A, B, and C (e.g., white arrow) represent natural variability without loss of key defining features, while transitions across broken lines (e.g., grey arrow) to states D, E, F, and G represent collapse and replacement by novel ecosystems that lack these features. Progression along different pathways of ecosystem collapse are assessed using variables X and Y or other ecosystem-specific diagnostic variables that reflect the loss of characteristic native biota and functionality. The outer broken line represents an alternative interpretation of ecosystem collapse in which state E is included within natural variation of the ecosystem type. Model uncertainty may be incorporated into Red List assessments by evidence-weighted model averaging in calculations of diagnostic variables based on the two alternative models.

variables such as productivity, species richness, or diversity. This appears to promote consistency, as all assessments use a common 'currency' to measure ecosystem degradation. The more flexible option, which permits assessor-defined ecosystem-specific variables, trades some of this apparent consistency for increased generality and realism, measuring more direct proxies of the ecological processes that threaten characteristic native biota. Thus, functionally contrasting systems may be assessed using different variables, which are scaled using an inferred threshold of collapse for ranking against a common set of ordinal threat categories (e.g., Figure 6 in Keith *et al.* 2013). Examples of suitable assessment variables include: measures of structural complexity or abundance of foundation species in forest or reef systems threatened by disturbance (Alvarez-Filip *et al.* 2009; Burns *et al.* 2014); trends in hydrological or water quality variables where there is evidence of biotic dependency in wetlands threatened by water extraction or pollution (Lester & Fairweather in Keith *et al.* 2013; Pisanu *et al.* 2014); and trends in the abundance of top predators or other species that provide crucial links in food webs in systems sustained by top–down regulatory processes (Schmitz *et al.* 2000; Estes *et al.* 2009).

The consultation process identified four reasons why gains in generality and realism of an ecosystem-specific approach should be worth an apparent reduction in consistency. First, scaled ecosystem-specific response variables should be more sensitive and consistent indicators of ecosystem degradation than generic variables when the relationship between the latter and the persistence of characteristic native biota is noisy or inconsistent across different ecosystems (e.g., Adler *et al.* 2011; Hooper *et al.* 2005). Prescribed response variables are workable for a narrow range of processes or ecosystem types (e.g., exploited fisheries; Hobday *et al.* 2011), but impose undue constraints for risk assessments across a wide range of ecosystems and processes. Second, the scaling of ecosystem-specific variables promotes consistency between assessments, so long as inferred thresholds of collapse are evidence-based and calculations of the relative severity of declines incorporate the uncertainties. Third, an ecosystem-specific approach is less limited by data availability than a prescriptive one-size-fits-all approach which requires the same variables to be quantified across all ecosystem types. Finally, ecosystem-specific approaches promote critical examination of the evidence in diagnosing causes of decline, the pathways of collapse, the most sensitive means of measuring decline along those pathways and setting an explicit threshold for unacceptable loss of characteristic native biota (Figures 1 and 2).

Further improvements in consistency should be possible by narrowing the range of ecological variables deemed appropriate for assessing particular types of ecosystem degradation, based on accumulating empirical experience. This will be increasingly possible as the RLE criteria are applied to many contrasting ecosystems by a diverse community of expert assessors. Imposing a prescriptive approach too early would stifle the exploration and evidence gathering essential to further development. In the meantime, published assessments provide guidance, and an IUCN peer-review process for global assessments will encourage consistency and promote rigor across assessments (IUCN 2014a; Rodriguez *et al.* 2015).

Delimiting ordinal categories of risk

The RLE criteria use numerical decision thresholds to assign ecosystem types to ordinal categories of risk (Appendix 1; Keith *et al.* 2013), a feature contributing to the simplicity of risk assessment outputs. A distinction exists between *decision thresholds*, which specify the conditions required to invoke a decision (e.g., categorization), and *ecological thresholds*, which are theoretical devices to explain nonlinear ecosystem behavior (Martin *et al.* 2009). The rationale for each RLE criterion derives from theory, while utility is the primary consideration for setting particular threshold values that delimit threat categories within each criterion, i.e., to make the categories informative about relative levels of risk (Keith *et al.* 2013, p. 10). Calls for a theoretical basis for decision threshold values (Boitani *et al.* 2014) thus confuse them with ecological thresholds. Utility was also important in setting thresholds in the RLTS. For example, the threshold delimiting Vulnerable from least concern for criterion A (population reduction) in the RLTS was increased from 20% (in v2.3) to 30% (v3.1, IUCN 2001) because of practical concerns that high proportions of some taxa would exceed the lower threshold, not because of new developments in population theory. Category thresholds may similarly require adjustments in the RLE as case studies accumulate.

In addition to considering utility in the RLE, the thresholds of spatial criteria (B1 and B2) are scale dependent (Nicholson *et al.* 2009). The scaling of these thresholds in different subglobal domains requires research to support subglobal-scale RLE assessments.

Current and potential RLE applications

The RLE criteria may be applied systematically to a set of ecosystem types within a specified domain (global or subglobal) or to single ecosystem types. The applications described below are for systematic assessments, but some may also be relevant to assessments of single ecosystems. The IUCN will lead the development of a global RLE, but will also support stakeholders to develop national and regional RLEs (IUCN 2014a), as these are the scales where many resource management and development decisions are made.

Red Lists of Ecosystems (sensu lato) are already part of the legislative or policy infrastructure for biodiversity conservation in some countries (Table 1). IUCN's new global standard is influencing federal jurisdictions such as Australia and the European Union, where member states are aiming to harmonize their listing methods (Nicholson *et al.* 2014). The IUCN RLE criteria also provide a benchmark for other countries wanting to develop their policy infrastructure (Government of Rwanda 2011).

The RLE can contribute significantly to monitoring and reporting the status of biodiversity. Some national reporting frameworks on threatened ecosystems are already in place (e.g., Riecken *et al.* 2006; Australian State of Environment Committee 2011; Driver *et al.* 2012). Tools such as the Red List Index (Butchart *et al.* 2004) promote temporal consistency. Internationally, global and subglobal RLEs could contribute to reporting instruments such as the Aichi Targets under the Convention on Biodiversity (especially target 5), the International Platform on Biodiversity and Ecosystem Services, and the evolving Sustainable Development Goals (IUCN 2014a).

A third application of RLEs is in the design of protected area networks, where they can identify locations of poorly reserved ecosystems that would benefit most (in terms of risk reduction) from inclusion in protected areas (Margules & Pressey 2000). The presence of threshold proportions of threatened and restricted-range ecosystems are proposed criteria for the identification of key biodiversity areas as sites contributing significantly to the global persistence of biodiversity (IUCN 2014b).

Fourth, RLEs can inform prioritization of investment decisions for conservation, restoration and development, for example, as inputs to cost efficiency analyses in allocating national or corporate budgets (Carwardine *et al.* 2008), and by contributing data to financial institutions for assessing investment risks for development projects (e.g., see Equator Principles, http://www.equator-principles.com/).

RLEs can support adaptive management strategies (Williams 2011), for example by informing sustainable allocations of water resources to environmental flows and agricultural production, or the zonation and quotas in fisheries management areas (Green *et al.* 2014). While the status of ecosystem types is the primary output of assessments, this is founded on an evidence-based diagnosis of threats and derivation of ecosystem-specific diagnostic variables that should be suitable for monitoring responses to alternative management actions.

RLEs may also inform the sustainable management and delivery of ecosystem services. The relationship between the RLE (for which the primary goal is to assess risks to ecosystem-level biodiversity) and ecosystem services is complex because the latter depend on both ecosystems and social environments (Costanza *et al.* 2014). To enhance clarity of purpose and simplicity of the RLE protocol, the RLE criteria focus on the loss of characteristic native biota (genes, populations, species, assemblages, and functional groups) and disruption to ecological processes that sustain it. Additional tools may thus be required to evaluate risks to ecosystem stocks, functions and services, especially if novel ecosystems deliver new or improved services relative to their collapsed predecessors.

Nonetheless, there are cases where the RLE assessments can provide important information about ecological changes that have major consequences for ecosystem services (Micklin 2006). The same causal factors that drive loss of biodiversity may also result in decline of ecosystem stocks, functions, and services (Cardinale *et al.* 2012). For example, Burns *et al.* (2014) assessed the Red List status of a forest ecosystem that had undergone major transformations in structure due to timber harvesting and successive canopy fires. At the crux of its Red List status was a loss of characteristic native biota associated with the declining abundance of large old trees, including mammals and birds that shelter and breed in tree cavities, insectivorous birds that forage in large canopies and under loose bark on large tree trunks, and poorly documented microbial and fungal diversity associated with deep forest litter. The same widespread transformation from old growth to regrowth that underpins the Red List status of this forest type has major implications for hydrological productivity (Kuczera 1987), carbon stocks and sequestration (Keith *et al.* 2009), timber production, as well as overall resilience of the system (Lindenmayer *et al.* 2011). Although the RLE criteria are focused on biodiversity, the outcomes of RLE assessments clearly have broader implications if interpreted with appropriate rigor and caution.

Finally, the RLE offers great potential for environmental education through extension programs, media articles, and resource materials for schools and community groups (http://www.iucnredlistofecosystems.org/). The RLE is at an early stage of development but has already generated considerable public interest (Marshall 2013).

Conclusions and future directions

In risk assessments generally, few hazards are defined precisely (Burgman 2005). Greater consistency could be achieved in the RLE through more prescriptive approaches to assessment (Boitani *et al.* 2014), but will these gains be worth the necessary trade-offs in generality and realism? The issue is not whether the concepts of ecosystems and collapse are imprecise or inconsistent (they are, undoubtedly) but whether the working definitions are useful and produce valuable outcomes. That is, will the RLE criteria (Keith *et al.* 2013) lead to better conservation decisions than would be achieved without them?

In sum, published assessments to date and our global consultation process suggest that the RLE framework achieves a workable balance among inevitable trade-offs (Keith *et al.* 2013), though we welcome empirical evidence indicating where a better balance is possible. We suggest a way forward by developing the RLE based on: (1) learning from systematic applications of the criteria across different ecosystem types and scales; (2) updating application guidelines regularly; (3) classifying and describing ecosystem types systematically as an integrated part of RLE assessments; (4) implementing an active research and development agenda; and (5) refining the conceptual framework and working definitions. In this, adaptive development of the RLE can benefit much from the history of the RLTS which, as one of IUCN's flagship products, was trialed and adjusted iteratively over 50 years from rudimentary beginnings (Scott *et al.* 1987).

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Supporting Information

Additional Supporting Information may be found in the online version of this article at the publisher's web site:

Appendix 1. Summary of IUCN Red List of Ecosystems categories and criteria v2.1.

References

Adler, P.B., Seebloom, E.W., Borer, E.T. *et al*. (2011). Productivity is a poor predictor of plant species richness. *Science*, **333**, 1750-1753.

Akçakaya, H.R., Ferson, S., Burgman, M.A. *et al*. (2000). Making consistent IUCN classifications under uncertainty. *Conserv. Biol.*, **14**, 1001-1013.

Alvarez-Filip, L., Dulvy N.K., Gill, J.A. *et al*. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proc. R. Soc. B*, **276**, 3019-3025.

Australian State of the Environment Committee (2011). Australian State of the Environment 2011. Independent report to the Australian Government Minister for Sustainability, Environment, Water, Population and Communities, Canberra.

Austin, M.P. (1985). Continuum concept, ordination methods, and niche theory. *Annu. Rev. Ecol. Syst.*, **16**, 39-61.

Boitani, L., Mace, G.M. & Rondinini, C. (2014). Challenging the scientific foundations for an IUCN Red List of Ecosystems. *Conserv. Lett.*, in press. [doi:10.1111/conl.12111].

Brooks, T. M. & Helgen, K. M. (2010) A standard for species. *Nature*, **467**, 540-541.

Butchart, S.H.M., Stattersfield, A.J., Bennun, L.A. *et al*. (2004). Measuring global trends in the status of biodiversity: Red List Indices for birds. *PLoS Biol.*, **2**, 2294-2304.

Burgman, M.A. (2005). *Risks and decisions for conservation and environmental management*. Cambridge University Press, Cambridge.

Burns, E.L., Lindenmayer, D.B., Stein J. *et al*. (2014). Ecosystem assessment of mountain ash forest in the Central Highlands of Victoria, south-eastern Australia. *Austral Ecol.*, **40**, in press. [doi:10.1111/aec.12200]

Cardinale, B.J., Duffy, J.E, Gonzalez, A. *et al*. (2012). Biodiversity loss and its impact on humanity. *Nature*, **486**, 59-67.

Carwardine, J., Klein, C.J., Roxburgh S.H. *et al*. (2008). Cost-effective priorities for global mammal conservation. *Proc. Natl. Acad. Sci. U.S.A.*, **105**, 11446-11450.

Commonwealth of Australia (2000). Environment Protection and Biodiversity Conservation Regulations, REG 7.02 Criteria for listing threatened ecological communities. Commonwealth of Australia, Canberra, ACT. http://www.frli.gov.au/ComLaw/Legislation/LegislativeInstrumentCompilation1.nsf/0/6762C4F433-643BDACA25728800075F3C?OpenDocument [Accessed 17 August 2014].

Costanza, R., deGroot, R., Sutton, P. *et al*. (2014). Changes in the global value of ecosystem services. *Glob. Environ. Change.*, **26**, 152-158.

Council of the European Communities (1992). Council Directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official J. Eur. Comm., series L* **206**, 7-50. URL: http://eur-lex.europa.eu/LexUriServ/LexUriServ.do?uri=CELEX:31992L30043:EN:HTML.

Dallimer, M., Irvine, K.N., Skinner, A.M.J. *et al*. (2012). Biodiversity and the feel-good factor: understanding associations between self-reported human well-being and species richness. *Bioscience*, **62**, 47-55.

Driver A., Sink, K.J., Nel, J.N. *et al*. (2012). National Biodiversity Assessment 2011: An assessment of South Africa's biodiversity and ecosystems. Synthesis Report. South African National Biodiversity Institute and Department of Environmental Affairs, Pretoria. Available http://bgis.sanbi.org [Accessed 17 August 2014].

Einstein, A. (1934). On the method of theoretical physics. *Philos. Sci.*, **1**, 163-169.

Estes, J.A., Doak, D.F., Springer, A.M. *et al*. (2009). Causes and consequences of marine mammal population declines in southwest Alaska: a food-web perspective. *Philos. Trans. R Soc. Lond. B*, **364**, 1647-1658.

Faber-Langendoen, D., Keeler-Wolf, T., Meidinger, D. *et al*. (2014). EcoVeg: a new approach to vegetation description and classification. *Ecol. Monogr.*, **84**, 533-561.

Folke, C., Carpenter, S.R., Walker, B. *et al*. (2004). Regime shifts, resilience and biodiversity in ecosystem management. *Annu. Rev. Ecol. Syst.*, **35**, 557-581.

Gaston, K.J. & Fuller, R.A. (2007). Commonness, population depletion and conservation biology. *Trends Ecol. Evol.*, **23**, 14-19.

Gorrod E.J., Bedward M, Keith D.A. *et al*. (2013). Systematic underestimation resulting from measurement error in score-based ecological indices. *Biol. Conserv.*, **157**, 266-276.

Government of Manitoba (2014). The Endangered Species and Ecosystems Act. http://web2.gov.mb.ca/laws/statutes/ccsm/e111e.php [Accessed 17 August 2014].

Government of Rwanda (2011). Rwanda Biodiversity Policy. Rwanda Environment Management Agency, Kigali. http://rema.gov.rw/rema.doc/pub/RWANDA%20BIODIVERSITY%20POLICY.pdf [Accessed 17 August 2014].

Green, A.L., Fernandes, L., Almany, G. *et al*. (2014). Designing marine reserves for fisheries management, biodiversity conservation, and climate change adaptation. *Coastal Manage.*, **42**, 143-159.

Hobbs, R.J. & Suding, K.N. (2009). *New models for ecosystem dynamics and restoration*. Island Press, Washington DC.

Hobday, A.J., Smith, A.D.M., Stobutzki, I.C. *et al*. (2011). Ecological risk assessment for the effects of fishing. *Fish. Res.*, **108**, 372-384.

Hooper, D.U., Chapin, III, F.S., Ewel, J.J. *et al*. (2005) Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecol. Monogr.*, **75**, 3-35.

IUCN (2001). *Red List categories and criteria*. International Union for Conservation of Nature, Gland.

IUCN (2014a). *Red List of Ecosystems Workshop Report. Cambridge, United Kingdom, 22–23 January 2014.* Commission on Ecosystem Management and Species Survival Commission, International Union for Conservation of Nature, Gland.

IUCN (2014b) *Consultation document on an IUCN standard for the identification of key biodiversity areas*. International Union for Conservation of Nature, Gland.

Jennings, S. (2005). Indicators to support an ecosystem approach to fisheries. *Fish Fish.*, **6**, 212-232.

Jerde, C.L., Mahon, A.R., Chadderton, W.L. *et al*. (2011). "Sight-unseen" detection of rare aquatic species using environmental DNA. *Conserv. Lett.*, **4**, 150-157.

Keith, D.A. (2009). The interpretation, assessment and conservation of ecological communities and ecosystems. *Ecol. Manage. Restor.*, **10**, S3-S15.

Keith, D.A. (2014). Separating risks from values in setting priorities for plant community conservation. *Appl. Veget. Sci.*, **17**, 384-384.

Keith, D.A. (2015). Assessing and managing risks to ecosystem biodiversity. *Austral Ecol.* **40**, in press [doi:10.1111/aec.12249].

Keith, D.A., Rodríguez, J.P., Rodríguez-Clark, K.M. *et al*. (2013). Scientific foundations for an IUCN Red List of Ecosystems. *PLoS One*, **8**(5), e62111.

Keith, H., Mackey, B.G. & Lindenmayer, D.B. (2009). Re-evaluation of forest biomass carbon stocks and lessons from the world's most carbon-dense forests. *Proc. Natl. Acad. Sci. U.S.A.*, **106**, 11635-11640.

Knapp, C.N., Fernandez-Gimenez, M.E., Briske, D.D. *et al*. (2011). An assessment of state-and-transition models: perceptions following two decades of development and implementation. *Rangeland Ecol. Manage.*, **64**, 598-606.

Kontula, T. & Raunio, A. (2009). New method and criteria for national assessments of threatened habitat types. *Biodivers. Conserv.*, **18**, 3861-3876.

Kuczera, G. A. (1987). Prediction of water yield reductions following a bushfire in ash-mixed species eucalypt forest. *J. Hydrol.*, **94**, 215-236.

Lester, R.E. & Fairweather, P.J. (2011). Creating a data-derived, ecosystem-scale ecological response model that is explicit in space and time. *Ecol. Model.*, **222**, 2690-2703.

Levins, R. (1966). The strategy of model building in population biology. *Am. Sci.*, **54**, 421-431.

Lindenmayer D.B., Hobbs R.J., Likens G.E., Krebs C. & Banks, S. C. (2011) Newly discovered landscape traps produce regime shifts in wet forests. *Proc. Natl. Acad. Sci. U.S.A.*, **108**, 15887-15891.

Lindgaard, A. & Henriksen, S. (2011). *The 2011 Norwegian Red List for ecosystems and habitat types*. Norwegian Biodiversity Information Centre, Trondheim.

Mace, G.M. (2004). The role of taxonomy in species conservation. *Philos. Trans. R. Soc. Lond. B*, **359**, 711-719.

Mackey, B., Berry, S. & Brown, T. (2008). Reconciling approaches to biogeographic regionalization: a systematic and generic framework examined with a case study of the Australian continent. *J. Biogeogr.*, **35**, 213–229.

Mant, J.G., Shiestl, F.P., Peakall, R. *et al*. (2002). A phylogenetic study of pollinator conservatism among sexually deceptive orchids. *Evolution*, **56**, 888-898.

Margules, C. & Pressey, R. (2000) Systematic conservation planning. *Nature*, **405**, 243-253.

Marshall, M. (2013). Forget the fish. Save the seas! *New Sci.*, **218**(2917), 6-7.

Martin, J., Runge, M.C., Nichols, J.D. *et al*. (2009). Structured decision making as a conceptual framework to identify thresholds for conservation and management. *Ecol. Appl.*, **19**, 1079-1090.

May, R.M. (2004). Tomorrow's taxonomy: collecting new species in the field will remain the rate-limiting step. *Philos. Trans. R. Soc. Lond. B*, **359**, 733-734.

Micklin, P. (2006). The Aral Sea crisis and its future: an assessment in 2006. *Eurasian Geogr. Econ.*, **47**, 546-567.

Nicholson, E., Keith, D.A. & Wilcove, D.S. (2009). Assessing the conservation status of ecological communities. *Conserv. Biol.*, **23**, 259-274.

Nicholson, E., Regan, T.J. & Auld, T.D. (2014). Towards consistency, rigour and compatibility of risk assessments for ecosystems and ecological communities. *Austral Ecol.*, **40**, in press [doi:10.1111/aec.12148].

Noss, R.F. (1996). Ecosystems as conservation targets. *Trends Ecol. Evol.*, **11**, 351.

Pisanu P., Kingsford R.T., Wilson B. & Bonifacio R. (2014) Status of connected wetlands of the Lake Eyre Basin, Australia. *Austral Ecol.*, **40**, in press [doi:10.1111/aec.12203].

Plagányi, E.E., Punt, A.E, Hillary, R. *et al*. (2014). Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish Fish.*, **15**, 1-22

Preston B.J. & Adam P. (2004). Describing and listing threatened ecological communities under the Threatened Species Conservation Act 1995 (NSW). *Environ. Plann. Law J.*, **21**, 250-263.

Ramirez-Llodra, E., Brandt, A., Danovaro, R. *et al*. (2010). Deep, diverse and definitely different: unique attributes of the world's largest ecosystem. *Biogeosciences*, **7**, 2851-2899.

Regan, H.M., Colyvan, M. & Burgman, M.A. (2002). A taxonomy of and treatment of uncertainty for ecology and conservation biology. *Ecol. Appl.*, **12**, 618-628.

Republic of South Africa (2004). National environmental management: Biodiversity Act. *Gov. Gazette Repub. South Africa Cape Town*, **467**, 1-84.

Riecken, U. Finck, P., Raths, U. *et al*. (2006). *Rote Liste der gefhahrdeten biotop-typen Deutschlands*. Bundeesamt fur Naturschutz, Bonn.

Rodrigues, A.S.L., Pilgrim, J.D., Lamoreux, J.F. *et al.* (2006). The value of the IUCN Red List for conservation. *Trends Ecol. Evol.*, **21**, 71-76.

Rodriguez J.P., Keith, D.A., Rodriguez-Clarke, K.M. *et al.* (2015). A practical guide to the application of the Red List of Ecosystems criteria. *Philos. Trans. R. Soc. B*, **370**, 20140003. [doi: http://dx.doi.or/10.1098/rstb.2014.0003]

Rodriguez, J.P., Rodriguez-Clark, K., Baillie, J.E. *et al*. (2011). Establishing IUCN Red List Criteria for threatened ecosystems. *Conserv. Biol.*, **25**, 21-29.

Rodríguez, J.P., Rodríguez-Clark, K.M., Keith, D.A. *et al*. (2012). IUCN Red List of ecosystems. *S.A.P.I.EN.S* [Online] 5.2: http://sapiens.revues.org/1286.

Rodrıguez, J.P., Rojas-Suárez, F. & Giraldo Hernández, D. (2010). *Libro Rojo de los Ecosistemas Terrestres de Venezuela*. Provita, Shell Venezuela y Lenovo (Venezuela), Caracas.

Sabo, J.L. (2008). Population viability and species interactions: life outside the single-species vacuum. *Biol. Conserv.*, **141**, 276-286.

Schmitz, O.J., Hambäck P.A. & Beckerman, A.P. (2000). Trophic cascades in terrestrial ecosystems: a review of the effects of carnivore removals on plants. *Am. Nat.*, **155**, 141-153.

Scott, P., Burton, J.A. & Fitter R. (1987). Red Data Books: the historical background. Pages 1-5 in R. Fitter, M. Fitter, editors. *The road to extinction: problems of categorizing the status of taxa threatened with extinction*, IUCN, Gland.

Sutherland, W.J., Aveling, R., Brooks, T.M. *et al*. (2014). A horizon scan of global conservation issues for 2014. *Trends Ecol. Evol.*, **29**, 15-22.

Tozer, M.G., Leishman, M.R. & Auld, T.D. (2014). Ecosystem risk assessment for Cumberland Plain Woodland, New SouthWales, Australia. *Austral Ecol.*, **40**, in press. [doi:10.1111/aec.12201]

Weston, P.H. (2013). What has molecular systematics contributed to our knowledge of the plant family Proteaceae? Pages 365-397 in P. Besse, editor. *Molecular plant taxonomy: methods and protocols*. Springer, New York .

Wiens, J.A. (1989). Spatial scaling in ecology. *Func. Ecol.*, **3**, 385-397.

Williams, B.K. (2011). Adaptive management of natural resources—framework and issues. *J. Environ. Manage.*, **92**, 1346-1353.

Wunder, S., Angelsen, A. & Belcher, B. (2014). Forests, livelihoods, and conservation: broadening the empirical base. *World Dev.*, S1-S11.

