POLICY PERSPECTIVE

Challenging the scientific foundations for an IUCN Red List of Ecosystems

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Abstract

The International Union for Conservation of Nature (IUCN) is currently discussing the development of a Red List of Ecosystems (RLE) that would mirror the categories and criteria used to assess the conservation status of species. The suggested scientific foundations for the RLE are being considered by IUCN for adoption as the backbone of the RLE. We identify conceptual and operational weaknesses in the draft RLE approach, the categories, and criteria. While species are relatively well-described units, there is no consistent means to classify ecosystems for assessing conservation status. The proposed RLE is framed mostly around certain features of ecosystems such as broad vegetation or habitat types, and do not consider major global change drivers such as climate change. We discuss technical difficulties with the proposed concept of ecosystem collapse and suggest it is not analogous to species extinction. We highlight the lack of scientific basis for the criteria and thresholds proposed by the RLE, and question the need to adopt the structure of the Red List of Species for an RLE. We suggest that the proposed RLE is open to ambiguous interpretations and uncertain outcomes, and that its practicality and benefit for conservation should be carefully evaluated before final approval.

Introduction

The IUCN (International Union for Conservation of Nature) has been pursuing the development of a Red List of Ecosystems (RLE) for over 5 years. Resolution 4.020 approved at the 4th IUCN World Congress in Barcelona in 2008 called for the initiation of "a consultation process for the development, implementation and monitoring of a global standard for the assessment of ecosystem status, applicable at local, regional and global levels." Draft categories and criteria were first proposed in 2008, and Resolution 055 at the 5th IUCN World Congress in Jeju, Korea in 2012 called for their formal approval by the IUCN Council, "once [they] have been rigorously tested." In 2013, Keith *et al.* (2013) published a paper presenting scientific foundations for an IUCN RLE.

The intent of Keith *et al.* (2013) is that the RLE becomes a risk assessment tool that would mirror the IUCN Red List for Species (RLS) (IUCN 2001), with analogous scientifically based categories and criteria, and be adopted by the IUCN and other conservation organizations to assess the conservation status of ecosystems. An IUCN tool to assess ecosystems has a sound conservation justification. The RLS has had a major role in directing conservation action on endangered species (Rodrigues *et al.* 2006; Lacher *et al.* 2012; Rondinini *et al.* 2013). The RLE could have a similar impact and reduce the loss of biodiversity by focusing on levels of ecological complexity poorly served by species-specific lists, by assessing conservation status of ecological communities and ecosystems. However, species and ecosystems are not comparable entities.

Keith *et al.* (2013) presented the concept of ecosystem collapse as analogous to species extinction and then derived five quantitative criteria to reflect the risks of collapse in an ecosystem, mirroring the criteria used by the RLS. We question the strength of these criteria for ecosystem collapse on several different grounds, but most fundamentally because "ecosystem" and "collapse" are not well defined. The foundation of any risk assessment is a clear definition of the entities that are at risk (e.g., species or ecosystems) and of the undesirable condition for which the risk is to be estimated (e.g., extinction, loss for ecosystem, collapse). This article identifies conceptual and operational weaknesses in the draft RLE approach, in the identification of the ecological entities and units for assessment that the RLE is expected to address, in the definition of their desirable/undesirable conservation status, in the selection of the categories and criteria used to assess their conservation status.

Species versus ecosystem concepts

Conceptual issues

Species (even accounting for the intrinsic fuzziness of the species concept, Hey *et al.* 2003) are well-identified units whose identity is relatively stable for conservation purposes: lists of species are published, validated, and available, and many species are listed in national and international legislation and policy. Names and identities of species do change, but the process of changing names occurs through established scientific processes, namely per reviewed publication. Although biodiversity can be monitored using a large number of indicators (Noss 1990), species are a fundamental component of biodiversity and one of the most important "currencies" for biodiversity conservation (Mace 2004).

Ecosystems, by contrast to species, are ecological systems consisting of all the organisms in an area and the physical environment with which they interact (Chapin *et al.* 2002). They are open systems, with flows of energy and matter in and out, and a set of characteristic functions. Ecosystems are dynamic with components, features, and functions changing over time, and can range in size across 10 orders of magnitude (Chapin *et al.* 2002). Since Tansley (1935) defined ecosystems to be highly dynamic systems, operating at many scales, multifunctional, nested, overlapping, and interconnected, his broad definition has remained substantially unchanged (cf. Chapin *et al.* 2002). Ecosystems are a fundamental concept for the scientific study of ecological processes and function, and incorporate biodiversity in senses that go well beyond richness and composition of species. Ecosystem science is directed toward understanding the processes, flows, and interactions among the components of ecosystems (Golley 1993). Since Clements' (1916) early definition of stable climax, more recent work by ecologists has shown that climax communities rarely occur and there is no stable state or equilibrium for an ecosystem (Chapin *et al.* 2002).

Keith *et al. (*2013) referred to Tansley (1935) to define four key characteristics of an ecosystem (i.e., native biota, abiotic environment, key processes and interactions, and spatial distribution) but we could not find these characteristics in Tansley's original paper. Tansley (1935) stated that ecosystems are defined by climatic conditions, soil composition, and the complex of species living in an area. Concepts such as "key processes and interactions" and "spatial distribution" are absent from Tansley's description of ecosystems. It can be argued that other ecologists (cf. Chapin *et al.* 2002 for an historical account) later added other features of ecosystems that can be summarized in the four proposed key characteristics. However, in spite of the extensive discussion by Keith *et al.* (2013), we find that the concepts of these ecosystem features are extremely difficult to apply in practice. There is no means to consistently define ecosystems for management and conservation because ecosystems operate at a range of scales and contain a range of features that could be employed given the ecological underpinnings of the concept.

Keith *et al.* (2013) proposed to define ecosystems by describing their "*functional component of characteristic biota.*" These are species that have major influences on ecosystem dynamics, such as ecosystem engineers, trophic or structural dominants, or functionally unique elements, including predators that have an overarching role in structuring animal communities. These concepts are far from settled scientifically. Many alternative species and functional groups could be selected, with different implications for the list, and there are no objective methods to determine which species could be selected to be functional component of characteristic biota.

Keith *et al.* (2013) proposed that ecosystems can be described by identifying and measuring key processes and interactions. With the exception of very few specific and well-studied systems, this must soon encounter the formidable challenges of maximizing consistency and minimizing controversy in describing ecological dynamics and applying theoretical concepts to the real world (Peters 1991), again with no clear principles by which differences of opinions can be settled. Ecosystems are defined through a set of parameters that have different significance depending on scale, ecological context, functional relationships, and species assemblages. Hence it is not surprising that no list of recognized types of ecosystems is currently available.

Operational implications

Keith *et al.* (2013) recognized the difficulties in defining the physical boundaries of ecosystems, and implicitly acknowledged the lack of a workable definition of ecosystem (they "*. . . regard other terms applied in conservation assessments, such as 'ecological communities,' 'habitats' and 'biotopes,' as operational synonyms of 'ecosystems'*"). Their proposed guideline for a workable definition of ecosystem is de facto based mostly on vegetation types and water bodies (and exceptional cases such as caves) and closely resembled the approach used to nominate "Threatened Ecological Communities" for listing in Australia (TSSC 2013). Keith *et al.* (2013) suggested that "*.. development of a global taxonomy for ecosystems can proceed contemporaneously with risk assessment*." They seemed to have a precise target of ecosystem scales for their system for the RLE: they affirm, for example, that "*To provide initial guidance, we suggest that a classification comprising a few hundred ecosystem types on each continent and in each ocean basin will be a practical thematic scale for global assessment.*" This suggestion has no obvious scientific foundation and it appears to be justified only by the need to find a practical framework (for a very specific geographical scale and thematic units, e.g., vegetation types) for an otherwise theoretical concept.

Keith *et al.* (2013) proposed criteria for assessment and examples of applications at very specific scales, i.e., at human landscape scale (hectares to square kilometers) for spatial scale and decades for temporal scales. At these scales, the only workable definition of a unit of ecosystem assessment that is applicable across large geographic areas is broad vegetation type (cf. Maes *et al.* 2013). This is the proxy the authors chose for most of their examples of ecosystems (Keith *et al.* 2013, supplementary material). The RLE model proposed by Keith *et al.* (2013) appears to be applicable only at these specific scales, not to all ecosystems at any scale or complexity. The criteria for assessing ecosystems, the threshold values of the variables, and the time scales of the proposed RLE model are tailored for application to landscape-scale vegetation types or abiotic units such as special geologies, floodplains (with spatially variable vegetation), or well-defined water bodies. The name "RLE" is misleading, because the model applies only to a spatially and temporally (and arbitrarily) defined subset of ecosystem proxies. While limiting the applicability to one specific scale can make the proposed model easy to operate, it also arbitrarily constrains the nature and scale of the components (species, ecological communities), the processes, and the functions that can be measured.

Climate change is happening (IPCC 2013) and will cause vegetation types and ecosystems to shift and change in composition (Scholze *et al.* 2006). Decades of research in community ecology have shown that species respond individually (not as communities) to environmental gradients and changing environmental conditions (Lavergne *et al.* 2010) across a wide range of temporal and spatial scales. The broad vegetation types usable for assessment may lose their current identity in a few decades lapse (Williams *et al.* 2007). In practical and theoretical terms, therefore, the model proposed by Keith *et al.* (2013), if accepted by IUCN, would provide the conservation community with a tool intended to assess something that is not well defined, understood, or stable over time.

Biodiversity decline and undesirable endpoints

Conceptual issues

Species can decline in abundance and go extinct locally or globally. Although sometimes difficult to assess with a high degree of confidence, extinction of species has a clear meaning—there are no entities of the taxon left. There is consensus that extinction is (with few exceptions for organisms causing deadly disease) an undesirable endpoint for species management and can usually be estimated unambiguously. Extinction of species is not equivalent to collapse of ecosystems, because ecosystems are dynamic composites that change in time and space. The species composition and internal processes of ecosystems can change rapidly, sometimes without loss of the system properties, and ecosystem properties can change with small variations to the biotic component (Chapin *et al.* 2002). The internal functions of ecosystems change. There are multiple possible endpoints for a changing ecosystem, with little consensus on what is desirable or undesirable. Furthermore, the concept of stable ecosystem state is discredited, as most ecosystems exist in alternative stable states (e.g., Hilderbrand *et al.* 2005). Keith *et al.* (2013) did not discuss climate change impacts, but the extensive modifications to ecosystems anticipated under future climate change (Williams *et al.* 2007; Fagre *et al.* 2009) make it even more difficult still to define an hypothetical stable or climax ecosystem state from which departures are presumed undesirable.

Operational implications

Keith *et al.* (2013) defined ecosystem collapse as "*A theoretical threshold, beyond which an ecosystem no longer sustains most of its characteristic native biota or no longer sustains the abundance of biota that have a key role in ecosystem organization [. . .]. Collapse has occurred when all occurrences of an ecosystem have moved outside the natural range of spatial and temporal variability in composition, structure and function.*"

This definition of collapse is problematic for several reasons. Ecosystems change over a range of time scales, from seasonal to evolutionary. Therefore, it is difficult to quantify the natural range of temporal variability. This is another important difference as compared to species, where the time frame for decline generally scales to generation length. Although sometimes difficult to assess (but see, e.g., Pacifici *et al.* 2013 for mammals), generation length of species has a clear definition, and the meaning of population decline of species from one generation to the next is well understood in conservation biology and risk assessment thanks to the well-established theory of population dynamics (Mace *et al.* 2008). The definition of collapse is so vague that in practice it will be possible (and often necessary) to define collapse separately for each ecosystem, using a variety of attributes and threshold values. Because such a fundamental concept as collapse is ecosystem-specific, ecosystem attributes will not be able to be defined consistently and comparisons of the risk levels of different ecosystems will be difficult. Comparability will be limited to different states of the same ecosystem in time and will be difficult between ecosystems. It will not be possible to know, for instance, whether an ecosystem listed as Critically Endangered is any more "at risk" than one listed as Vulnerable or as Least Concern. Comparability across all "units" of assessment (ecosystems) and consistency through the assessment process are essential to ensure that the results of the assessment are useful in guiding conservation action (Mace *et al.* 2008).

Criteria

Keith *et al.* (2013) proposed a specific set of quantitative criteria for listing ecosystems according to their risk of collapse. Given the difficulties in determining the units for assessment (ecosystems) and the features of collapse, it is even more difficult to base the criteria upon consistent ecological theory and we argue that the theoretical basis for most of the five criteria proposed for the RLE are generally unsatisfactory or absent.

Criteria A (reduction in geographic distribution) and B (restricted geographic distribution) are based on spatial extent of the ecosystem and likely to be the most frequently used criteria because of the availability of spatial data. Keith *et al.* (2013) discussed the theory for criteria based on spatial decline, centered on the species-area models and the vulnerability of small patches of ecological communities. They acknowledged the species-area quantitative relationship does not support universal threshold values of decline for assessing ecosystem status. While intuitively appealing for its analogy to population size and decline, there is little scientific ground for the application of these criteria and their thresholds to large-scale ecosystems (see, e.g., Radford *et al.* 2005 for the relationship between habitat loss and bird species richness at landscape level). Two key spatial parameters that are used by the RLS (IUCN 2001) are the Extent of Occurrence (EOO, the area encompassing all known occurrences of a species) and the Area of Occupancy (AOO the area occupied by a species; Gaston 1991). Although their application has generated some confusion, the reasons why these variables are used in the RLS are clear (EOO for risk spreading and AOO for population size and habitat specificity) (Mace *et al.* 2008). It is not clear what the basis is for including them in RLE: what are the system- or process-level phenomena that inform the areas used for the proposed thresholds? The relationship between ecosystem area and species presence or ecosystem functions has empirical support at small patch and landscape levels (e.g., Bender *et al.* 1998) but is not informative when applied at larger scale (Huggett 2005). It may be argued that the EOO can be used to inform on risk spreading, but it remains obscure how the AOO analogue can be applied to ecosystems. It is conceptually hard to accept that the patches of an ecosystem type are separate ecological entities isolated from the wider (and different) ecosystem in which they are nested. While the logic for criterion B (restricted distribution) might be good, its content and the proposed quantitative thresholds are not linked to collapse by ecological theory that can be applied consistently at all scales, and have no justification other than a suggested analogy with the RLS.

Criteria C (environmental degradation based on change in an abiotic variable) and D (disruption of biotic processes or interactions based on change in a biotic variable) suffer from the intrinsic subjectivity of the chosen variable. Keith *et al.* (2013) offered a list of examples of potentially suitable variables linked to various abiotic features/process and ecosystem functions. Each variable would have its own methods for measuring it and thresholds, adding to the problem of comparability of risk level among different ecosystems. The criteria use the strength of the pressure rather than its impact on ecosystem properties and functions. This makes it impossible to calibrate across ecosystems, and also makes the criteria vulnerable to false information when pressure and response become decoupled for any reason.

Criterion E (quantitative analysis that estimates the probability of ecosystem collapse) is analogous to a full PVA for species. It is likely to be used rarely because of lack of sufficient data, and if used would suffer from the same uncertainties about definitions of ecosystem and collapse described above.

Thresholds

In the RLS, thresholds are set at arbitrary points but informed by underlying theory, and the parameters and scales used are based on sound theory and refer to an underlying process that is explicit (Brook *et al.* 2002; Mace *et al.* 2008). To reduce the subjectivity of most thresholds for assessing the conservation status of Australian "ecological communities," the guidelines of the Australian government (TSSC 2013) suggest indicative thresholds that refer either to a well-defined benchmark "original" condition (i.e., pre-European time) or to thresholds to assess key species of the communities, which are linked to the RLS criteria and thresholds.

There is theoretical evidence of ecological thresholds and some empirical evidence limited to few ecological systems but the concept and the supporting evidence are still controversial and lack full generality (see Huggett 2005 for a review). There is no theory for the proposed quantitative thresholds of the RLE, as clearly stated by Keith *et al.* (2013): ".. *in the absence of a clear theoretical foundation for setting particular thresholds . . . , we set threshold values at relatively even intervals for current and future declines.*" This setting is acceptable only under the assumption of a linear function between probability of collapse and values of the criteria, obviously a very difficult assumption and broadly disproved by most ecological theory (e.g., Saunders *et al.* 1991, for ecosystem fragmentation). The relationship between "thresholds" and loss of ecosystem composition and function is highly context-specific and can hardly be generalized with quantitative and general values (Huggett 2005).

While Keith *et al.* (2013) provided extended discussion of the proposed criteria A–E, we found no evidence supporting the selection of thresholds values. For example, we found no justification for the use of (1) 2,000–20,000–50,000 km^2 as thresholds of threat categories of an ecosystem's spatial distribution, (2) 80% extent and 50% severity as thresholds of loss, (3) 50 years as the time period to assess change. Keith *et al.* (2013) proposed to extrapolate the occurrence of an ecosystem in 1 km^2 to the entire 100 km^2 grid cell, which they suggested as the standard cell for distributional data. For criterion B (restricted distribution), Keith *et al.* (2013) stated that they "..*recognise that such thresholds are somewhat arbitrary*.." and acknowledged that ".. *the proposed thresholds are based on* [their] *collective experience . . .*" We do not think this is an acceptable basis for a system aiming at becoming a general world standard. It is not clear what the experience relates to and therefore the circumstances under which these values could be justified.

There needs to be much stronger science to support the proposed values of Keith *et al.* (2013) for the thresholds of the five criteria for ecosystem assessment. Such science would provide the necessary scientific foundation to avoid subjective and gross thresholds with little basis other than mimicking the RLS. However, with the lack of a clear endpoint for collapse, and lack of a general theory linking ecosystem reduction (of area, components, and functions) to collapse and applicable at all scales and all ecological systems, setting scientifically based thresholds for decline is likely to remain a very difficult task.

Relationship between RLE and RLS

Keith *et al.* (2013) suggested that the RLE could be used as a surrogate model for species assessment when data on individual species are hard to identify. However, this argument is hard to understand because it seems to imply that there is some general and linear relationship between species decline and ecosystem degradation. The evidence for links from biodiversity to ecosystem function is based on more ecological features than species (often functional types), and the generally positive relationship is highly context-dependent (Loreau *et al.* 2001). In closely controlled experiments, about half of the variation in function depends on species diversity and a similar proportion on the identity of the species present in the assemblage (Cardinale *et al.* 2012; Hooper *et al.* 2012). How many species and processes can change before an ecosystem changes its identity and is called something else is often of little interest for species conservation, because the species involved are not necessarily those of highest conservation concern (Mace *et al.* 2012).

Way forward

We suggest the following steps that could strengthen the process of developing an RLE:

- Identify the entities to be assessed in a clear, unambiguous, and consistent way. Ecosystems are equivocal units and are subject to a variety of interpretations in their spatio-temporal scales, identity, and functions. The term "ecosystem type" or something equivalent could be used to indicate that the proposed model is not about all ecosystems but a classification of them into discrete units based on one or more dominant features. Perhaps a workable solution to the major problem of ecosystem identification is to change the focus of the unit of assessment for threat and state that the RLE model is proposed for assessing vegetation types (or, e.g., "habitats" sensu the European Habitats Directive where habitats are indeed habitat types defined by phyto-sociological criteria, i.e., assemblages of botanical species) or "ecological communities" (sensu TSSC 2013). It should be clearly stated, however, that there are larger scale and smaller scale processes and ecosystems that will be missed necessarily and that this classification operates under the assumption that we are dealing with time scales too short for climate change to alter these units, i.e., the vegetation types.

- Most of the problems with the proposed RLE caused by the lack of clarity on what an ecosystem is, and how it can be defined, would then be resolved. The model proposed here is theoretically more defensible than the model proposed by Keith *et al.* (2013) because it would deal with vegetation types (which have their own four essential elements implicit in the definition of ecosystem). Assessing broad vegetation types may be useful to conservationists and policy makers who commonly operate at landscape/vegetation scale (1-10,000 ha). Criteria A (declining distribution) and B (restricted distribution) of the model proposed by Keith *et al.* (2013) implicitly confirm its applicability only to the landscape scale, as it requires that units be measured in km^2.
- Ensure that the area-based units for risk assessment are identified at the scale and by attributes that have the same conceptual and theoretical basis across all global assessment projects suggested by Keith *et al.* (2013). The goal to producing a global List of Threatened Ecosystem can only be achieved if there is one global classification of units of assessment across all continents and at one desired scale.
- Whatever the rules are for determining the ecosystem types for the list, they need to be operationally straightforward (so that the system can be applied globally) but must also result in units that are relevant and consistent considering the definition of collapse. This is where the conceptual linkages between the units for assessment and the undesirable endpoints will need to draw on relevant ecological science.
- Identify a clear and unambiguous definition of the undesirable endpoints that the proposed model is tracking trends toward, and be clear about why they matter for conservation responses. Substantial effort should be spent in standardizing the concept of collapse for the chosen units of assessment. As a suggestion, a collapse-equivalent concept should probably focus more on the loss of species and less on the disruption and loss of ecological processes, which are more vulnerable to opportunistic selection and subjective description.
- Make sure the categories are relevant to the goal of the assessment, conceptually and nomenclaturally. Mimicking the terms in the RLS is not necessarily useful when the concept is different.
- Make sure the criteria are relevant to the goal of the assessment. There is no need to replicate the criteria of the RLS. An effort should be made to design a new and independent conceptual framework, and learn from the limitations of the RLS (Possingham *et al.* 2002). It might be useful to instead frame the variables in the criteria around the response of the ecosystem to the pressures, rather than the pressure itself. This would support consistency across assessed ecosystems in the same way that the RLS uses standardized measures of population decline and fragmentation that are comparable across very different species.
- The revised set of attributes (of the units of assessment) should ensure that they can be practically and unambiguously measured through standard and stable quantitative methods (e.g., species diversity, species composition, area, services such as carbon storage or water quality, flow of nutrients).
- Define thresholds that are supported by the best ecological theory or clearly linked to the nature of the variables: absolute quantities whenever possible and relative percentages for trends and losses. Arbitrary thresholds are acceptable on condition that they are based on solid scientific foundation and not primarily on personal experience.

In conclusion, the RLE model proposed by Keith *et al.* (2013) has fundamental weaknesses as well as practical difficulties. We are concerned that introducing the RLE on this basis will risk misinterpretations, ambiguities, and approximations that will at best be a distraction and cause confusion, and at worst lead to poorly based conservation priorities and completely discredit the RLE process.

Furthermore, we are concerned that much effort is going in defining a new assessment tool and will likely go to applying it to ecosystems in various parts of the world without a clear definition of the expected conservation benefits. If resources were invested by IUCN in the RLE, it would be useful to know in advance what outcomes can be expected over and above investment in the RLS. If the RLE is defined at a particular thematic resolution (and we have the means to measure information content in relation to thematic resolution), how much better or worse might conservation outcomes be? Putting resources into promoting and discussing the details of the RLE without a clear scientific foundation and without a cost-benefit assessment of the conservation outcomes is not an effective use of the limited resources available for conservation.

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