## 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 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

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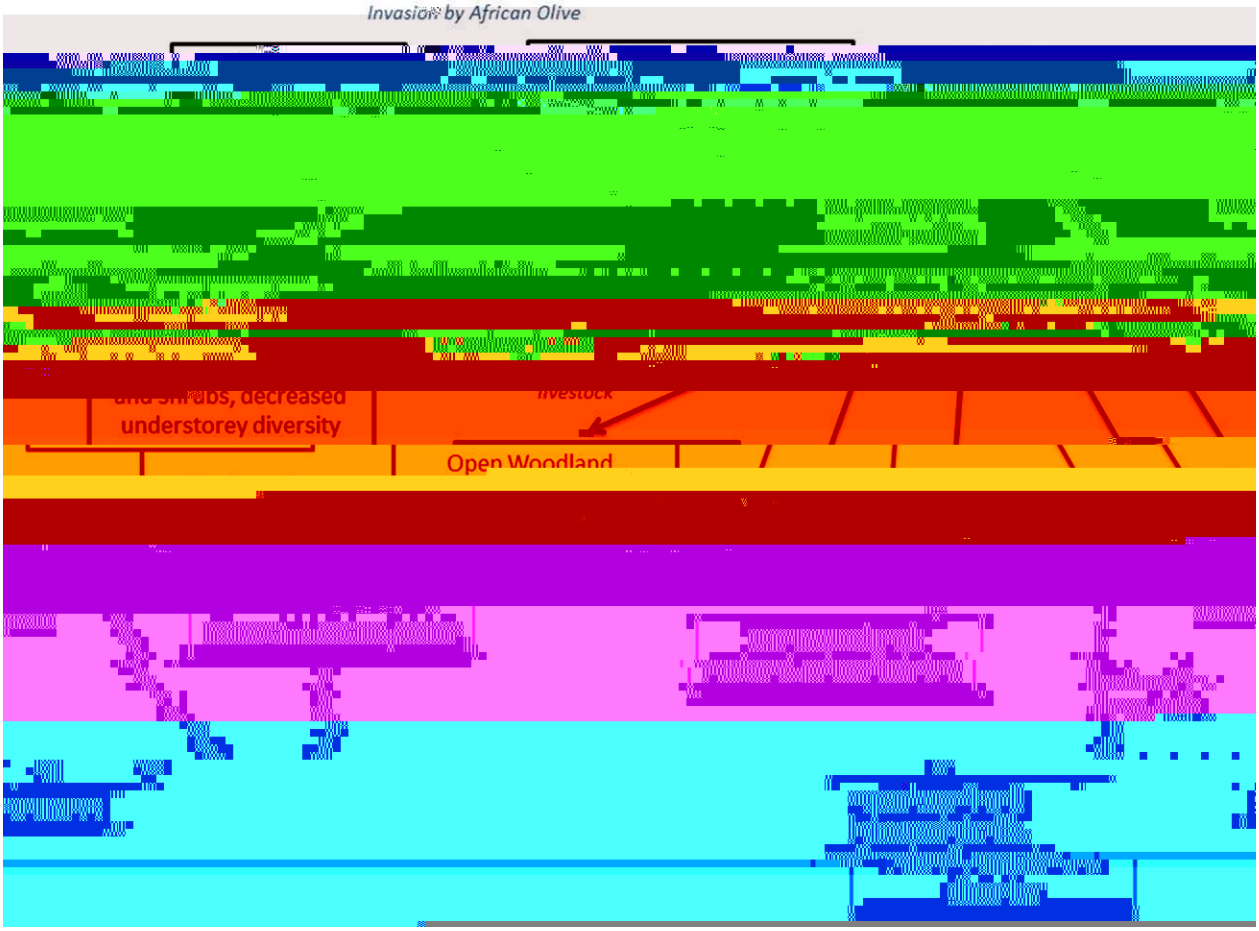


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criterion E (Appendix 1; Keith *et al.* 2013; Burns *et al.* 2014). For many ecosystem types, however, there may be insufficient data on which to build models with adequate realism. Criteria C and D (Appendix 1) enable use of diagnostic variables that 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

**F[illegible] 2** [illegible] *et al.* [illegible] *et al.* [illegible] 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 contribsi2 ofof

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

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