Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria

Edited by D.A. Keith, J.R. Ferrer-Paris, S.M.M. Ghoraba, S. Henriksen, M. Monyeki, N.J. Murray, E. Nicholson, J. Rowland, A. Skowno, J.A. Slingsby, A.B. Storeng, M. Valderrábano and I. Zager

Version 2.0

INTERNATIONAL UNION FOR CONSERVATION OF NATURE

CEM

RED LIST OF ECOSYSTEMS

About IUCN

IUCN is a membership Union uniquely composed of both government and civil society organisations. It provides public, private and non-governmental organisations with the knowledge and tools that enable human progress, economic development and nature conservation to take place together.

Created in 1948, IUCN is now the world's largest and most diverse environmental network, harnessing the knowledge, resources and reach of more than 1,400 Member organisations and around 16,000 experts. It is a leading provider of conservation data, assessments and analysis. Its broad membership enables IUCN to fill the role of incubator and trusted repository of best practices, tools and international standards.

IUCN provides a neutral space in which diverse stakeholders including governments, NGOs, scientists, businesses, local communities, indigenous peoples organisations and others can work together to forge and implement solutions to environmental challenges and achieve sustainable development.

Working with many partners and supporters, IUCN implements a large and diverse portfolio of conservation projects worldwide. Combining the latest science with the traditional knowledge of local communities, these projects work to reverse habitat loss, restore ecosystems and improve people's well-being.

www.iucn.org
https://x.com/IUCN

Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria

Edited by D.A. Keith, J.R. Ferrer-Paris, S.M.M. Ghoraba, S. Henriksen, M. Monyeki, N.J. Murray, E. Nicholson, J. Rowland, A. Skowno, J.A. Slingsby, A.B. Storeng, M. Valderrábano and I. Zager

Version 2.0

IUCN is pleased to acknowledge the support of its Framework Partners who provide core funding: Ministry of Foreign Affairs, Denmark; Ministry for Foreign Affairs, Finland; Government of France and the French Development Agency (AFD); Ministry of Environment, Republic of Korea; Ministry of the Environment, Climate and Sustainable Development, Grand Duchy of Luxembourg; the Norwegian Agency for Development Cooperation (Norad); the Swedish International Development Cooperation Agency (Sida); the Swiss Agency for Development and Cooperation (SDC) and the United States Department of State.

Published by: IUCN, Gland, Switzerland

Produced by: IUCN Commission on Ecosystem Management (CEM)

Recommended citation: IUCN (2024). *Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria,* Version 2.0. Keith, D.A., Ferrer-Paris, J.R., Ghoraba, S.M.M., Henriksen, S., Monyeki, M., Murray, N.J., Nicholson, E., Rowland, J., Skowno, A., Slingsby, J.A., Storeng, A.B., Valderrábano, M. & Zager, I. (Eds.). Gland, Switzerland: IUCN.

ISBN: 978-2-8317-2281-8 (PDF)

DOI: https://doi.org/10.2305/CJDF9122

Cover photos: © David Keith, Maxwell Ridgway, Daniel Pelaez Duque via Unsplash

Contents

Executive summary

The IUCN Red List of Ecosystems is the global standard for ecosystem risk assessment and a framework for monitoring the status of the world's ecosystems. It is part of the growing toolbox for assessing risks to biodiversity and aims to support conservation, resource use and management decisions by identifying ecosystems most at risk of biodiversity loss. By targeting a level of biological organisation above species, the IUCN Red List of Ecosystems complements The IUCN Red List of Threatened Species™ in supporting biodiversity conservation decision making and action. The *IUCN Red List of Ecosystems Categories and Criteria* are designed to be widely applicable across ecosystem types and geographical areas, transparent and scientifically rigorous, and easily understood by policy makers and the public.

The IUCN Red List of Ecosystems Categories and Criteria

The basis of the IUCN Red List of Ecosystems is the *IUCN Red List of Ecosystems Categories and Criteria*, a set of eight categories and five criteria that provide a consistent method for assessing the risk of ecosystem collapse. The eight categories of ecosystem risk are: Collapsed (CO), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC), Data Deficient (DD) and Not Evaluated (NE).

The IUCN Red List of Ecosystems protocol comprises five rule-based criteria (A–E) for assigning ecosystems to a risk category. Two of these criteria assess spatial symptoms of ecosystem collapse: declining distribution (A) and restricted distribution (B). Two criteria assess functional symptoms of ecosystem collapse: environmental degradation (C) and disruption of biotic processes and interactions (D). Multiple threats and symptoms can be integrated in a model of ecosystem dynamics to produce quantitative estimates of the risk of collapse (E). The Guidelines include comprehensive sections to support application of each of the five criteria, including information on relevant theory, thresholds and examples.

Application and documentation standards

The Guidelines assist correct application of the *IUCN Red List of Ecosystems Categories and Criteria* by providing background on the development and scientific foundations of the categories and criteria, identifying appropriate input data, elucidating the interpretation of the listing criteria and associated concepts, and detailing methods for assessing the listing criteria. Section 1 Introduction describes the objectives, development and governance of the Red List of Ecosystems. Section 2 Categories of the IUCN Red List of Ecosystems explains the structure of the risk assessment protocol. Section 3 Scientific foundations defines foundational concepts including assessment units (ecosystem types), ecosystem collapse, the multiple dimensions of ecosystem scale, and standards of evidence in the context of uncertainty. The Guidelines also provide a Glossary of the terms used in the *IUCN Red List of Ecosystems Categories and Criteria*.

The Guidelines aim to support the practical implementation of the *IUCN Red List of Ecosystems Categories and Criteria* from sub-national to global areas of assessment. Section 4 Assessment process outlines the necessary steps in the assessment process from defining the assessment area and the assessment units to documentation requirements. Section 5 Criterion A. Reduction in geographic distribution, Section 6 Criterion B. Restricted geographic distribution, Section 7 Criterion C and D. Environmental degradation and disruption of biotic processes and Section 8 Criterion E. Quantitative risk analysis provide detailed technical information on the application of

each criterion. Section 9 Guidance on specific drivers of ecosystem collapse gives additional guidance on special topics including climate change and ecosystem fragmentation. All the steps are illustrated with examples spanning a wide range of ecosystem types, geographical localities and levels of data availability. Finally, Section 10 Databasing, peer review and publication outlines processes for databasing, peer review and publication.

The future of the IUCN Red List of Ecosystems

The IUCN Red List of Ecosystems programme will assess the global status of the world's terrestrial, marine, freshwater and subterranean ecosystems. In addition, the programme aims to support the development of national and regional Red Lists to inform conservation planning and sustainable development. For more information on the IUCN Red List of Ecosystems please consult the IUCN Red List of Ecosystems website (www.iucnrle.org).

Acknowledgements

IUCN gratefully acknowledges the dedication and efforts of the hundreds of scientists and practitioners who have contributed to the scientific development and practical testing of the *IUCN Red List of Ecosystems Categories and Criteria* since 2008. These experts have cumulatively participated in more than 100 meetings and workshops in more than 50 countries around the world, as well as giving input and suggestions remotely. To all we are most indebted.

We especially thank Dr Andrés Etter and Dr Tracey Regan, who undertook thorough peer reviews of Version 2.0 of the Guidelines. We also thank Dr Emily Botts for diligent editing and layout of the final manuscript.

The process to develop Red List criteria for ecosystems was launched with Resolution 4.020 at the Fourth IUCN World Conservation Congress in 2008 and consolidated with Resolution 5.055 adopted by the Fifth World Conservation Congress in 2012. This process culminated in the adoption of the *IUCN Red List of Ecosystems Categories and Criteria* by the IUCN Council in May 2014. Further information on the development of the IUCN Red List of Ecosystems is available in Section 1.2 Development of the IUCN Red List of Ecosystems.

The following organisations are gratefully acknowledged for their support for the revised Version 2.0 of the Guidelines for the IUCN Red List of Ecosystems: Australian Research Council, Provita, South African National Biodiversity Institute, Norwegian Biodiversity Information Centre, IUCN Commission on Ecosystem Management, IUCN Secretariat, University of New South Wales, James Cook University, University of Melbourne, and Deakin University.

Contributing authors

- David A. Keith
- José R. Ferrer-Paris
- Somaya Magdy M. Ghoraba
- Snorre Henriksen
- Mapale Monyeki
- Nicholas J. Murray
- Emily Nicholson
- Jessica Rowland
- Andrew Skowno
- Jasper A. Slingsby
- Anne B. Storeng
- Marcos Valderrábano
- Irene Zager

IUCN Red List of Ecosystems
Steering Committee

- David A. Keith, Chair, Committee for Scientific Standards
- Angela Andrade Perez
- Thomas M. Brooks
- Kendall Jones
- Richard Kingsford
- Emily Nicholson
- David Obura
- Beth Polidoro
- Irene Zager

Staff
- Marcos Valderrábano, Red List of Ecosystems Programme Officer

IUCN Commission on Ecosystem Management (2021–2025)
Steering Committee

- Angela Andrade Perez, Chair
- Mariana Rivera, Assistant to Chair
- Liette Vasseur, Deputy Chair
- Shalini Dhyani, Vice Chair for Asia
- Bernal Herrera-Ferandez, Vice Chair for Latin America and the Caribbean
- Jonathan Hughes, Vice Chair for Europe
- Doris Mutta, Vice Chair for Africa
- Peter Smith, Vice Chair for Oceania

Red List of Ecosystems Thematic Group
- David A. Keith, Lead
- Emily Nicholson, Co-lead

Acronyms

Acronym	*Definition*
AOO	Area of occupancy
ARD	Absolute rate of decline
BAKT	IUCN Biodiversity Assessment and Knowledge Team
CEM	IUCN Commission on Ecosystem Management
CO	Collapsed
CR	Critically Endangered
DD	Data Deficient
DOI	Digital Object Identifier
EA	Ecosystem Accounting
EFG	Ecosystem Functional Group
EN	Endangered
EOO	Extent of occurrence
GET	IUCN Global Ecosystem Typology
GIS	Geographic Information System
GSP	Global Species Programme
IUCN	International Union for Conservation of Nature
LC	Least Concern
NE	Not Evaluated
NT	Near Threatened
PRD	Proportional rate of decline
RLE	IUCN Red List of Ecosystems
RLTS	The IUCN Red List of Threatened Species™
SDC	Science and Data Centre
SEEA	System of Environmental Economic Accounting
SSC	IUCN Species Survival Commission
UN	United Nations
VU	Vulnerable

Glossary

Term	*Definition*
Abiotic ecosystem properties	The non-living chemical and physical properties of an ecosystem.
Anthropogenic ecosystems	Ecosystem types created and sustained by intensive human activities. For some of these systems, cessation of those activities leads to transformation into ecosystem types with different properties and organisational processes (from Keith et al., 2022).
Area of assessment	Defines the spatial bounds of an IUCN Red List of Ecosystems assessment (e.g. a country, marine region, continent, the world, etc.).
Area of occupancy (AOO)	A standardised measure of the area that is occupied by an ecosystem type.
Biotic ecosystem properties	The living components or properties (organisms) of an ecosystem.
Characteristic native biota	Biological features that define the identity of a natural or semi-natural ecosystem type and distinguish it from other ecosystem types and/or drive ecosystem dynamics and function, e.g. ecological processes, ecosystem engineers, trophic or structural dominants, functionally unique elements, species interactions.
Continuing decline	A gradual or episodic decline in geographic distribution, ecological process, biotic or abiotic environment that is likely to continue into the future and is non-trivial in magnitude, and its effect on the sustainability of characteristic native biota.
Ecosystem collapse	A transformation of identity, a loss of key defining biotic or abiotic features, and a replacement by a different ecosystem type or anthropogenic environments.
Ecosystem integrity	The degree to which the current composition, structure and function of an ecosystem resemble that of its reference states (after Nicholson et al., 2021). Reference states should be based on replicated samples of the ecosystem type with minimal exposure to threatening processes, such as agriculture, mining, invasive species, alteration to disturbance regimes, timber harvest and fishing (Nicholson et al., 2021).
Ecosystem type	The unit of assessment for the IUCN Red List of Ecosystems (see Section 3.1 Ecosystem types: the units of assessment).
Extent of occurrence (EOO)	A standardised measure of the area within which all occurrences of an ecosystem type exist.
Geographic distribution	Represents all known spatial occurrences of an ecosystem type at a specified time (see Section 3.3.1 Time frames for time frames of assessment).
Grain size	The size of the spatial unit (e.g. grid cell, polygon segment) used to measure a distribution.
Location	See Threat-defined location.
Natural ecosystems	Ecosystems that have not been substantially transformed into semi-natural or anthropogenic ecosystem types by human activity, but may have undergone varying degrees of degradation, and hence declines in ecosystem integrity.

Term	Definition
Reference state (of an ecosystem)	The state of an ecosystem in which ecosystem properties (biotic and abiotic features, ecological processes and geographic distribution) are largely unaffected by intensive or broad-scale human activity. For practical purposes, the year 1750 marks a reference date at the beginning of the industrial era when broad-scale exploitation of ecosystems began to accelerate markedly, noting that major ecosystem transformations had occurred in some areas at earlier times (Section 5.3 Applying criterion A). Properties of reference states exhibit natural variability in space and time that should be considered in description and analysis. The properties of reference states may be inferred from historical, relictual and modelled information.
Relative severity	The estimated magnitude of past or future environmental degradation or disruption to biotic processes, expressed as a percentage relative to a change large enough to cause ecosystem collapse.
Semi-natural ecosystems	Ecosystems that have been partially but substantially transformed by human activity (e.g. by elimination of a major structural or functional component), retaining some properties of the native or natural ecosystem type from which they were derived, as well as some novel properties (i.e. those that were not characteristic of ecosystem types from which they were transformed). Typically, they have anthropogenic origins that extend earlier than the onset of the industrial era (c. 1750), although this generalisation is yet to be tested. They may be important for biodiversity conservation in some extensively transformed anthropogenic landscapes and seascapes.
Spatial extent	The total area of an ecosystem type estimated with a specified metric.
Spatial scale of ecosystem units	An umbrella term describing measures of the resolution of spatial information (maps or digital data) for the geographic distribution of an ecosystem unit (see grain size).
Temporal variation within ecosystem units (temporal scale)	The changes in ecosystem properties (composition, structure or function) that occur as part of natural dynamics and turnover within an ecosystem type. The trajectories of these changes are typically reversible or cyclical over specific time frames. For example, temporal variability may occur within a year (e.g. seasonal cycles in freeze-thaw streams and lakes), across multiple years (e.g. deserts and ephemeral wetlands with boom-bust dynamics driven by interannual weather cycles) or across repeatable successional pathways (e.g. decadal or century-scale fire cycles in pyric forest ecosystems).
Thematic scale of ecosystem units or classification	The resolution at which an ecosystem unit is classified, which may be represented by the specific level of a hierarchical classification and the relative degree of variation represented within and between the units at that level. A coarse thematic scale describes a classification level with few heterogeneous units, whereas a fine thematic scale describes a classification level with more homogeneous units.
Threat-defined location	A geographically or ecologically distinct area in which a single threatening event can rapidly affect all occurrences of an ecosystem type.

Term	*Definition*
Threat event	An incidence of a threatening process, capable of causing decline or degradation of an ecosystem's properties that occurs independently of other such events in time and space. Examples include a heat wave event causing coral bleaching that is part of the overall process of climate change, a short fire interval causing tree decline that is part of an overall process of changing fire regimes, and construction of a stream barrier that is part of an overall process of water (flow and flood) regime change. See Threat-defined locations (Section 6.3 Applying criterion B) for further detail.
Threatening process (plausible)	An agent or causal factor that negatively affects the properties of an ecosystem type. Its action may be expressed as a discrete event, a series of events or a continuous process. Examples include forest clearing, introduction of invasive species that cause decline in native species, climate change, changes in fire regimes, etc. A threatening event is 'plausible' if it has a non-negligible probability (e.g. > 1%) of occurring within the next 20 years.
Time frame (of assessment)	The total period over which ecosystem change is assessed. This varies between subcriteria for criteria A, C, D and E.

1. Introduction

The IUCN Red List of Ecosystems (RLE) was developed to promote a consistent global framework for monitoring the status of ecosystems (Keith et al., 2015). It is part of the growing toolbox for assessing risks to biodiversity and aims to support conservation, resource use and management decisions by identifying ecosystems most at risk of biodiversity loss. By targeting a level of biological organisation above species, the IUCN Red List of Ecosystems complements The IUCN Red List of Threatened Species™ (IUCN, 2012), together providing simultaneous assessment of broad- and fine-scale biodiversity. A combined approach is more likely to achieve the aim of comprehensive, effective and representative conservation outcomes, and will improve the ability to monitor the status of biodiversity on Earth.

The basis of the RLE is the *IUCN Red List of Ecosystems Categories and Criteria* (Appendix 1), a set of five criteria and associated thresholds that provide a repeatable, globally consistent method for classifying the risk of ecosystem collapse (Rodríguez et al., 2015; Keith et al., 2013). Ensuring accurate and comparable assessments for all ecosystem types included on the RLE is a key challenge for the RLE programme. These Guidelines provide the information required to meet this challenge.

The Guidelines assist users to correctly implement the *IUCN Red List of Ecosystems Categories and Criteria* by accompanying the assessor through the RLE assessment process, from understanding the scientific foundations through to finalising assessments for publication.

1.1. Objectives of the IUCN Red List of Ecosystems

The primary goal of the RLE is to support conservation in resource use and management decisions by identifying ecosystems most at risk of biodiversity loss (Keith et al., 2013; Keith et al., 2015). By assessing relative risks of biodiversity loss at the ecosystem level, the RLE accounts for broad-scale ecological processes and important dependencies and interactions among species (Keith et al., 2015). The RLE also shines a light on common species, which define the identity of many ecosystems, are involved in key interactions with large numbers of co-occurring species and can have major influences on ecosystem form and function (Gaston & Fuller, 2007). To achieve the primary goal of the RLE, listing categories and criteria were designed to be:

1. A standard method for assessing and comparing risks of ecosystem collapse.
2. Easily understood by policy makers and the public.
3. Transparent, objective and scientifically rigorous.
4. Applicable to terrestrial, marine, freshwater and subterranean systems.
5. Applicable to risk assessments of local to global areas.
6. Flexible to use data of varying quality and coverage.
7. Consistent with, and complementary to, The IUCN Red List of Threatened Species.

Although the primary goal of the RLE is focused on biodiversity conservation, the data associated with the RLE may inform a wide range of other activities, including the sustainable management of ecosystem services. Of themselves, the risk categories that constitute the primary output of RLE assessments are not designed to be priority setting tools for ecosystem conservation or management, or to reflect the ability of ecosystems to provide ecosystem services. However, the RLE status, and associated products of RLE assessments (e.g.

descriptions, maps, threat data), can inform such applications, which will usually require additional tools to achieve effective planning outcomes (Keith et al., 2015).

1.2. Development of the IUCN Red List of Ecosystems

The global need for an international standard for ecosystem risk assessment was recognised through successive IUCN resolutions. In 2008, Resolution 4.020 on *Quantitative Thresholds for Categories and Criteria of Threatened Ecosystems* (Fourth World Conservation Congress, Barcelona, 2008) actively promoted the development of formal categories and criteria, requesting that IUCN "initiate 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." The Fifth World Conservation Congress (Jeju, 2012) adopted Resolution 5.055 on the *Consolidation of the IUCN Red List of Ecosystems*, requesting IUCN Council to "take the necessary steps for formal approval of the categories and criteria as an official IUCN data analysis protocol for use by the Members and any other stakeholder interested in ecosystem risk assessment".

Between 2007 and 2013, with significant contributions from the scientific, government and conservation sectors, the IUCN Red List of Ecosystems Thematic Group of the Commission on Ecosystem Management (CEM) developed and iteratively refined a set of criteria for assessing risks of ecosystem collapse. Justification for the RLE and initial criteria were published in 2011 (Version 1.0; Rodríguez et al., 2011). These were revised as the scientific foundations of the RLE were further developed, through review of literature and extensive consultation. This revised set of *IUCN Red List of Ecosystems Categories and Criteria*, their conceptual foundations and 20 application cases from around the world were published in 2013 (Keith et al., 2013). IUCN Council examined the *IUCN Red List of Ecosystems Categories and Criteria* and on 21 May 2014 adopted them as the official global standard for assessing the risk to ecosystems. Guidelines were first published in 2016, with updated versions based on continued application of the criteria at national, regional and local levels, spanning many ecosystem types worldwide (Keith et al., 2015; Bland et al., 2019; Nicholson et al., 2024).

The RLE has since seen rapid uptake and impact in global, national and local policy and practice. There is a growing body of evidence of its value and impact in ecosystem management and conservation, particularly at the national level (e.g., Bland et al., 2019; Botts et al., 2020; Salomaa & Arponen, 2023; Keith et al., 2023a; Nicholson et al., 2024). In 2022, the RLE was adopted as a Headline Indicator in the Kunming-Montreal Global Biodiversity Framework under the Convention on Biological Diversity. It supports monitoring and implementation of the Framework's explicit, ecosystem-focussed goals and targets at global and national levels (CBD, 2022). The RLE is also recommended for assessing risks and impacts across private sector mechanisms (e.g. under the fourth criterion of the International Finance Corporation's *Performance Standard 6* (IFC, 2019), triggering more stringent biodiversity requirements for financing).

The RLE complements and contributes to other global standards for assessing biodiversity and ecosystem change. By addressing a higher level of biological organisation, the RLE complements The IUCN Red List of Threatened Species (IUCN, 2022a), which assesses species extinction risk. RLE assessments contribute to identification of Key Biodiversity Areas (KBAs), through criteria relating to threatened ecosystems (IUCN, 2022b). It provides a complementary framework for assessing ecosystem change to the UN Statistical Commission's standard for natural capital accounting, the System for Environmental Economic Accounting Ecosystem Accounting (SEEA EA, Edens et al., 2022; UNCEEA, 2021). SEEA EA is a statistical framework for quantifying changes in ecosystem extent and condition, ecosystem service provision, and contributions of ecosystems to people and the economy. The RLE and SEEA EA draw on similar concepts and data for classifying, mapping and quantifying ecosystem change (Xiao et al., 2022). However, the RLE identifies threatened ecosystems by evaluating the impacts of change in ecosystem distribution and integrity on the risk of ecosystem collapse.

1.3. Governance of the IUCN Red List of Ecosystems

The RLE is jointly coordinated by two IUCN bodies, the Commission on Ecosystem Management (CEM) and the Science and Data Centre (SDC). It is governed by two interacting committees with specific functions: (i) the Steering Committee, and (ii) a Committee for Scientific Standards. It is supported by the Red List of Ecosystems Thematic Group of the CEM, which is a group of volunteer experts that undertake diverse duties in support of the objectives of the RLE. The RLE Programme Unit administers the RLE and ensures global coordination of the experts involved in research, implementation and peer review activities.

1.3.1. The Steering Committee

The RLE Steering Committee oversees the implementation of the *IUCN Red List of Ecosystems Categories and Criteria* at global and sub-global levels. The Steering Committee is composed of the Lead (and if applicable, the Co-lead) of the Red List of Ecosystems Thematic Group of the CEM (appointed by the Chair of the CEM), the Chair of the RLE Committee for Scientific Standards, the Chair of the CEM, the Head of the Biodiversity Assessment and Knowledge Team (BAKT), the Chief Scientist and up to four RLE Partners appointed by the Chair of the CEM because of their specific technical or organisational expertise. The Head of BAKT and the Chief Scientist represent the IUCN Secretariat.

The Steering Committee has the following functions:

1. Develop and manage the strategy and work plan for the implementation of the RLE worldwide, to achieve the goal of assessing all ecosystems at a global level by 2025.
2. Establish a mechanism for periodically updating global assessments.
3. Identify and approach potential sources of financial support for assessments and their dissemination.
4. Supervise a team of professional staff within the RLE Programme Unit, and build a network of volunteers to implement the RLE work plan both within the CEM and the IUCN Secretariat.
5. Actively engage the CEM in developing and peer reviewing assessments at the global and sub-global levels.
6. Develop training materials and guidelines in the three official IUCN languages to support assessments.
7. Recommend appointments to the RLE Committee for Scientific Standards.
8. Ensure that progress of the RLE is reported back to the IUCN Council and Secretariat senior management.
9. Ensure that progress and outcomes of the RLE are well communicated in the scientific literature and media.
10. Ensure the execution of the RLE work plan and maintain cooperation among collaborating organisations.
11. Actively engage with others involved in the development, testing and applications of knowledge products mobilised by IUCN.

1.3.2. The Committee for Scientific Standards

The RLE Committee for Scientific Standards is the principal scientific body that provides expertise in the development, application and review of all issues related to the RLE. The Committee consists of scientific experts with balanced expertise spanning a range of skills, including risk assessment, ecological modelling, remote sensing, ecosystem classification and mapping, decision theory, and ecology of terrestrial, freshwater, marine and subterranean ecosystems. The combined expertise of the members of the Committee for Scientific Standards covers the full diversity of ecosystem types and geographical regions.

Members of the Committee for Scientific Standards, including the Chair and Deputy Chair, are proposed by the RLE Steering Committee. The Chair of the CEM is ultimately responsible for appointing members to a maximum four-year term, which expires at the following session of the IUCN World Conservation Congress. One seat of the Committee for Scientific Standards is reserved for a representative of The IUCN Red List of Threatened Species designated by the Species Survival Commission (SSC) and the Global Species Programme (GSP).

The Committee for Scientific Standards promotes the application of high scientific standards to the implementation of the *IUCN Red List of Ecosystems Categories and Criteria*, and ensures that the intent of the categories and criteria is not compromised. The specific functions of the Committee for Scientific Standards are:

1. Develop and maintain technical guidelines in the three IUCN official languages to support the application of the *IUCN Red List of Ecosystems Categories and Criteria*, including details on implementation standards and data quality.
2. Provide scientific advice on the categories and criteria to the RLE Steering Committee and the Programme Unit.
3. Provide scientific advice and support to the Programme Unit on the development of databases, training materials and other resources.
4. Provide scientific advice on the design and implementation of systematic ecosystem risk assessment projects that could contribute to the global RLE.
5. Manage a peer review process of all classifications and maps of ecosystem types proposed for use in the global RLE.
6. Manage a peer review process for all assessments proposed for inclusion in the global RLE and, subject to the outcomes of the review process, submit recommendations to the Steering Committee on the inclusion or rejection of these assessments.
7. Critically review all applications of criterion E.
8. Provide scientific support and training for sub-global assessments of ecosystem types via the RLE Programme Unit and other RLE partners.
9. Promote and undertake research to improve ecosystem risk assessment methodologies underpinning the *IUCN Red List of Ecosystems Categories and Criteria*.
10. Submit all formal decisions and recommendations of the RLE Committee for Scientific Standards to the Steering Committee for review and formal adoption.

1.4. Structure of the Guidelines

The *Guidelines for the application of IUCN Red List of Ecosystems Categories and Criteria* provide the information necessary to conduct a robust and repeatable ecosystem risk assessment suitable for inclusion on the RLE. Section 1 Introduction offers an overview of the motivation and history of the RLE, describing its general objectives and governance structures. Section 2 Categories of the IUCN Red List of Ecosystems presents the categories. Section 3

Scientific foundations summarises the science underlying the categories and criteria, and presents the RLE risk assessment model.

Section 4 Assessment process guides assessors through a full assessment suitable for submission, including the steps required to define the area and units of assessment, and the key ecosystem processes that will permit accurate application of the five criteria. The following four sections outline the scientific theory underpinning each criterion, the estimation of variables for assessment, and the values of the thresholds for each category – Section 5 Criterion A. Reduction in geographic distribution, Section 6 Criterion B. Restricted geographic distribution, Section 7 Criterion C and D. Environmental degradation and disruption of biotic processes and Section 8 Criterion E. Quantitative risk analysis.

Section 9 Guidance on specific drivers of ecosystem collapse gives supplementary guidance on climate change and fragmentation as drivers of ecosystem collapse. Section 10 Databasing, peer review and publication describes the standards for evaluating the quality of a risk assessment and the process of preparing an assessment for peer review and publication. Throughout, a series of worked examples and case studies are provided to assist assessors with the implementation of the categories and criteria.

A summary sheet of the current version of the *IUCN Red List of Ecosystems Categories and Criteria* is included as Appendix 1. More information on the RLE, links to relevant documents, and summaries of case studies are available in multiple languages on the RLE website (www.iucnrle.org).

2. Categories of the IUCN Red List of Ecosystems

The IUCN Red List of Ecosystems (RLE) includes eight categories: Collapsed (CO), Critically Endangered (CR), Endangered (EN), Vulnerable (VU), Near Threatened (NT), Least Concern (LC), Data Deficient (DD) and Not Evaluated (NE; Figure 1). The first six categories (CO, CR, EN, VU, NT and LC) are ordered in decreasing risk of collapse. The categories Data Deficient and Not Evaluated do not indicate a level of risk.

The categories Critically Endangered, Endangered and Vulnerable indicate threatened ecosystems and are defined by quantitative and qualitative criteria described in Appendix 1. These categories are nested, so that an ecosystem type meeting a criterion for Critically Endangered will also meet the criteria for Endangered and Vulnerable. The three threatened ecosystem categories are complemented by several qualitative categories that accommodate: (i) ecosystem types that almost meet the quantitative criteria for Vulnerable (Near Threatened); (ii) ecosystems that unambiguously meet none of the quantitative criteria (Least Concern); (iii) ecosystems for which too few data exist to apply any criterion (Data Deficient); (iv) ecosystems that have not yet been assessed (Not Evaluated). Following the precautionary principle (Precautionary Principle Project, 2005), the overall status of an ecosystem type is the highest risk category obtained through any criterion.

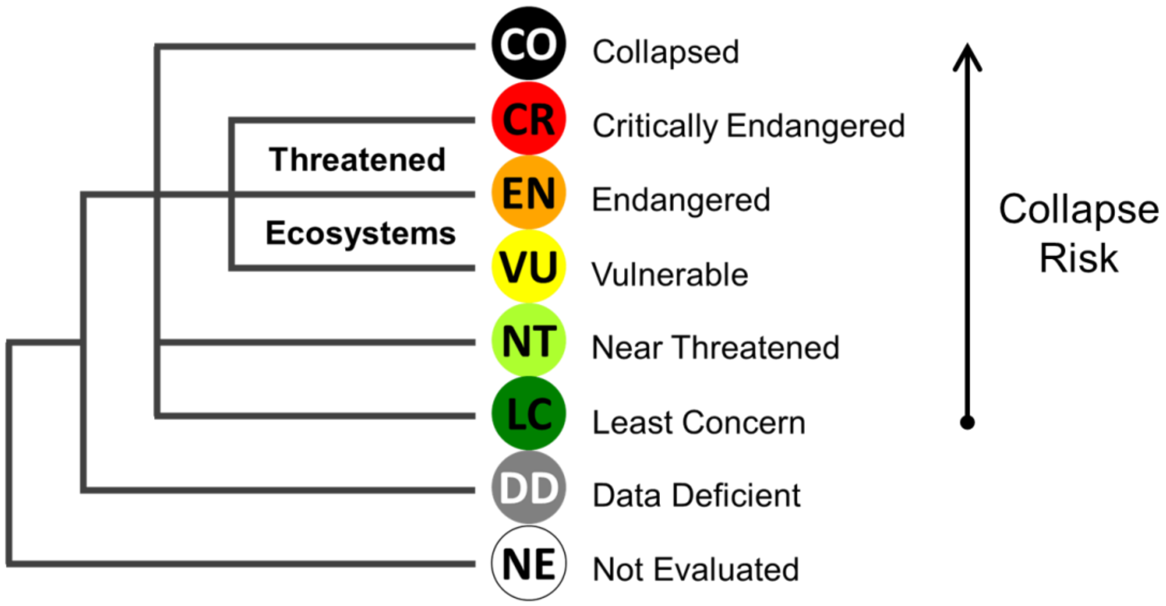


Figure 1. Structure of the IUCN Red List of Ecosystem categories. Source: Authors.

Collapsed (CO)

An ecosystem is Collapsed when it is virtually certain (Table 4) that its defining biotic or abiotic features are lost from all occurrences, and the characteristic native biota are no longer sustained. Collapse may occur when most of the diagnostic components of the characteristic native biota are lost from the system, or when functional components (biota that perform key roles in ecosystem organisation) are greatly reduced in abundance and lose the ability to recruit.

Critically Endangered (CR)

An ecosystem is Critically Endangered when the best available evidence indicates that it meets any of the criteria A to E for Critically Endangered. It is therefore considered to be at an extremely high risk of collapse.

Endangered (EN)

An ecosystem is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered. It is therefore considered to be at a very high risk of collapse.

Vulnerable (VU)

An ecosystem is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable. It is therefore considered to be at a high risk of collapse.

Near Threatened (NT)

An ecosystem is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.

Least Concern (LC)

An ecosystem is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widely distributed and relatively undegraded ecosystems are included in this category.

Data Deficient (DD)

An ecosystem is Data Deficient when there is inadequate information to make a direct, or indirect, assessment of its risk of collapse based on decline in distribution, disruption of ecological function or degradation of the physical environment. Data Deficient is not a category of threat, and does not imply any level of collapse risk. Listing of ecosystems in this category indicates that their situation has been reviewed, but that more information is required to determine their risk status.

Not Evaluated (NE)

An ecosystem is Not Evaluated when it has not yet been evaluated against the criteria.

3. Scientific foundations

3.1. Ecosystem types: the units of assessment

The IUCN Red List of Ecosystems (RLE) protocol is a robust and generic risk assessment framework that can be applied to internally consistent classifications of ecosystem types. It has flexibility to assess risks to ecosystems that vary greatly in biological and environmental characteristics, scales of organisation and amounts of available data. The clear definition and description of ecosystem types is therefore an essential first step to RLE assessment.

Ecosystems are complexes of organisms and their associated physical environment within a specified area (Tansley, 1935). They have four essential elements: a biotic complex, an abiotic environment, the interactions within and between them, and a physical space in which these operate (Pickett & Cadenasso, 1995). Guidance on how to apply these concepts to define and describe suitable units for RLE assessment is given in Section 4.2 Describing the unit of assessment.

3.1.1. Ecosystem typologies

IUCN's global standard for ecosystem classification, the Global Ecosystem Typology (IUCN, 2020; Keith et al., 2022; https://global-ecosystems.org/) provides an overarching global framework for contextualising and comparing units of assessment in Red Listing. It is a hierarchical classification that defines major groups of ecosystems distinguished by their functional properties in three upper levels and different compositional expressions of these ecosystem functional groups in three lower levels (see Section 4.2.1 Classification). Red List assessments may be carried out on ecosystem units defined at any thematic scale equivalent to levels 4–6 of the Global Ecosystem Typology, i.e. they should be compositionally distinctive expressions of ecosystems within an Ecosystem Functional Group (level 3 of the typology).

Assessors should identify the respective Ecosystem Functional Group (EFG) to which each of their assessment units belongs by comparing their descriptions to those of EFGs (Keith et al., 2022; https://global-ecosystems.org/). For units that share properties of more than one EFG, the EFG with the most similar properties should be identified and other candidate EFGs should be noted.

The *IUCN Red List of Ecosystems Categories and Criteria* may be applied systematically to a set of ecosystem types within a specified area of assessment (global or sub-global), or strategically to single ecosystem types to inform ecosystem management (Keith et al., 2015). The units of standalone strategic assessments should be referrable to EFGs, so long as the unit of assessment is clearly defined and delineated. Systematic assessments of multiple ecosystems are typically based on a classification of ecosystem types that ensures consistent and comparable ecosystem risk assessments across the area of assessment. The classification may simply delineate units at a particular thematic scale, or may describe their relationships using hierarchies or nested arrangements that span a range of thematic scales (Rodríguez et al., 2011). Where no classification currently exists, the IUCN Global Ecosystem Typology (Keith et al., 2020) can guide the development of a suitable set of assessment units, first by identifying EFGs represented in the area of assessment, and second by using available information to delineate different compositional expressions of each EFG at the desired level of detail (see Murray et al., 2019, 2020 for an example).

Sub-global assessments may be based on established national or regional ecosystem classifications, providing the units of assessment conform to the definition of ecosystem types (see Section 3.1 Ecosystem types: the units of assessment). These units should be justified as suitable proxies for ecological assemblages and should be cross-referenced to the IUCN Global Ecosystem Typology. A number of jurisdictions have developed suitable typologies to support national RLE assessments (Table 1; Kontula & Raunio, 2009; Lindgaard & Henriksen, 2011; Driver et al., 2012). These inventories are outcomes of ongoing investment in classification, mapping, validation and revision. Their units represent level 6 of the Global Ecosystem Typology. Typically, however, classifications and maps of adjoining jurisdictions may not align with one another due to different methodological approaches, scales, data availability and stages of development. Attribution of all classification units to EFGs is therefore a basic requirement to facilitate contextualisation and comparison across boundaries. Similarly, broader classifications for larger regions (e.g. Ferrer-Paris et al., 2019) should be attributed to EFGs by comparing descriptions of their units as described above.

Globally recognisable ecosystem types should not be confused with biogeographic or biophysical ecoregions (Spalding et al., 2007), or biomes (Allen & Hoekstra, 1990). Ecoregions and biomes are areas that share common macro-environmental or biogeographical features and contain complexes of contrasting, but co-occurring ecosystem types (Spalding et al., 2007; Keith et al., 2022). The potential heterogeneity of ecoregions and biomes makes them unsuitable units for most RLE applications (Rodríguez et al., 2015; Keith et al., 2015; Keith et al., 2013). The same is true for units in the upper levels (realms, biomes and EFGs) of the IUCN Global Ecosystem Typology (Keith et al., 2022). Other terms applied in conservation assessments such as ecological communities, habitat types, biotopes and vegetation types (largely in the terrestrial context) – may be regarded as operational synonyms of ecosystem types (Nicholson et al., 2009) providing they meet the requirements of the RLE standard for describing ecosystem assessment units (Section 4.2 Describing the unit of assessment). Although vegetation or habitat types are often suitable units for RLE assessments, their descriptions typically do not include sufficient information on ecosystem processes and fauna components and therefore need supplementation to meet the requirements.

3.1.2. Assessments of anthropogenic and semi-natural ecosystem types

In theory, risks to any natural, semi-natural or anthropogenic ecosystem type can be assessed using the RLE criteria and categories. For example, some systematic Red List projects assess risk of collapse for agricultural systems together with natural ecosystems (e.g. Delarze et al., 2016). Anthropogenic ecosystem types are created and sustained by intensive human activities (see Glossary; Keith et al., 2022), primarily through manipulation of resources, abiotic or biotic features and processes (e.g. through actions such as vegetation clearing, earthworks, addition of artificial substrates, introduction or harvest/control of biota, drainage, irrigation, fertiliser addition, etc.) (Nicholson et al., 2021).

Semi-natural ecosystems (see Glossary) are of particular interest because, although partially transformed, they may have important conservation values and consequently have been included within several national and regional Red List assessments. The national Red List of Norwegian ecosystems (NBIC, 2018), for example, includes several semi-natural grassland ecosystem types (e.g. Box 1) that are attributable to Ecosystem Functional Group T7.5 'Derived semi-natural pastures and old fields' within the IUCN Global Ecosystem Typology (Keith et al., 2022). Novel ecosystems that replace fully collapsed antecedent natural ecosystem types (e.g. Box 2) may similarly be assessed as semi-natural or anthropogenic ecosystem types, subject to the precautions outlined below.

Box 1: Red List assessments of semi natural ecosystems in Norway.

The Norwegian national Red List for ecosystems (NBIC, 2018) includes semi-natural ecosystems which are defined as “holistic ecosystems with a species composition that combines human influence with significant variation due to naturally occurring environmental variables”. This definition excludes anthropogenic systems with significant human impact and without significant variation in species composition along natural local environmental gradients.

Coastal heathland is a semi-natural ecosystem type in Norway shaped through the clearing of forest and several thousand years of use. It is a near-coastal system characterised by treeless vegetation with heather (*Erica* spp.), grasses and herbs. The most important anthropogenic processes for maintaining coastal heathland are burning of heather and frequent grazing through most or all of the growing season. Burning prevents tree encroachment and transition to forest, and also removes old heather and benefits a number of herbs and grasses, creating more productive foraging conditions for sheep.

The main cause of biotic degradation of coastal heathland is the rapid decline in traditional use from around 1900 and especially since 1950, with reduced burning and grazing enabling trees and shrubs to establish. However, several additional factors contribute to or speed up the degradation of coastal heathland. Exotic conifers such as sitka spruce *Picea sitchensis* and mountain pine *Pinus mugo*, planted in coastal heathland or adjacent areas for timber production or as windbreaks, are becoming established in large parts of the area, changing the heathland into forest. A warming climate is expected to accelerate this process. In southern Norway, abiotic degradation is caused by air-born nitrogen input. This leads to an increased dominance of grass and sedge and causes the coastal heaths to change in the direction of grass heaths. Coastal heathland is therefore assessed as Endangered (EN) according to criteria D2 and D3.

Because it is threatened and a large proportion of the global area is found in Norway, coastal heathland was designated as a ‘selected nature type’ in accordance with the Norwegian Nature Diversity Act. This designation required the state to develop and implement an action plan to safeguard the nature type and to avoid habitat loss or deterioration of the ecological status of the areas. Further information on ‘selected nature types’ is available at: https://www.regjeringen.no/en/dokumenter/nature-diversity-act/id570549/

Assessors should very carefully distinguish these semi-natural ecosystem types from degraded states of natural ecosystems, which may form part of the assessment of natural ecosystems under criteria C or D. Unlike semi-natural ecosystem types, degraded states of natural ecosystem types should be clearly associated with their undegraded states, which remain extant in the same landscapes or seascapes. Semi-natural ecosystem types may be derived from multiple different natural ecosystem types and their characteristic biota may include biota that are native to the region in which they occur, but not necessarily to the natural ecosystem types from which they were derived. Semi-natural ecosystems typically have anthropogenic origins that extend earlier than the industrial era, whereas degraded states of natural ecosystems typically originated since the onset of industrialisation (c. 1750).

This distinction is critical to ensure that RLE assessment outcomes are valid for reporting (and other uses) because if degraded ecosystems are incorrectly reclassified as semi-natural then they may be assessed at much lower risk than their degraded natural ecosystem would be, resulting in an underestimate of biodiversity risks and loss.

The methods for applying RLE criteria to semi-natural and anthropogenic ecosystem types are the same as for any other ecosystem type (see Section 4 Assessment process). However, some precautions are needed when using the results of systematic Red List assessments if they include anthropogenic or semi-natural ecosystem types. If results from RLE are used for priority setting (e.g. for management actions or designation of new protected areas), the inclusion of anthropogenic ecosystem types with 'natural' ecosystems in the same prioritisation may produce perverse results that conflict with biodiversity conservation objectives. Some actions to reduce risks to plantations may increase risks to certain natural systems, or divert scarce resources needed to reduce risks to unique natural ecosystem types.

To avoid unintended outcomes, assessors should consider whether a Red List assessment for anthropogenic and semi-natural ecosystems is needed, and if so, how the results will be applied in a way that does not compromise biodiversity conservation objectives. Three particular questions should be addressed:

1. **Which candidate ecosystem types in the area of assessment are 'natural', 'semi-natural' and 'anthropogenic'?** Assessors should consult definitions of anthropogenic and semi-natural ecosystems (Glossary) and descriptions of relevant anthropogenic EFGs in the IUCN Global Ecosystem Typology (Keith et al., 2022). Inevitably, there will be some uncertainty in interpreting whether a particular ecosystem type is anthropogenic or semi-natural. Assessors are advised to consider the degree of direct human intervention on ecosystem properties and their maintenance, as well as methods for dealing with uncertainty and expert opinion (see Section 3.2.2 Uncertainties in units and endpoints and Section 3.3.4 Quantitative data and expert knowledge). In general, assessors should interpret ecosystems as anthropogenic if the major defining properties and processes are derived from human intervention (e.g. based on the relative abundance of native and introduced biota, resemblance of structural properties to the untransformed state, etc.), or as semi-natural if those properties exhibit substantial influence from both natural and anthropogenic interventions. The latter could result from centuries of traditional management or relatively recent management practices.
2. **Why is a Red List assessment needed?** The answer depends on the objectives and context of the assessment, as well as whether unique ecological properties and processes are represented in the ecosystem type. Assessments of anthropogenic or semi-natural ecosystem types may be needed if they have biodiversity conservation values that would benefit from a risk assessment. Semi-natural ecosystems should be included in Red List assessments if they provide significant biodiversity values that sustain biota or ecological processes that are no longer represented or rarely represented in natural landscapes or seascapes. Anthropogenic ecosystems may be included in Red List assessments if policy requires a comprehensive assessment of all ecosystem types within an area (e.g. a country). For example, this may be required for comparisons with ecosystem accounts (UNCEEA, 2021). To support the justification for Red List assessments of semi-natural or anthropogenic ecosystems, their descriptions (Section 4.2 Describing the unit of assessment) should articulate their key features that contribute to major values in terms of biota and ecological processes. For example, Red List assessments of traditional agroecosystems may be relevant where they contribute to agrobiodiversity and associated use knowledge. In general, however, trade-offs should be considered to optimise the use of scarce resources available for assessments, and this will often result in pragmatic decisions to include important semi-natural ecosystem types and to exclude anthropogenic ecosystem types from Red Listing. Red List assessments of anthropogenic ecosystems should not be necessary if there is no identified need for the output.

3. **Does an intended application of Red List data, if applied to an anthropogenic or semi-natural ecosystem type, create potential conflicts with ecosystem management actions that may be required for conservation of other ecosystem types?** If so, then the anthropogenic ecosystem should be excluded from that application or the application should be modified to avoid such a conflict. This requires careful consideration of the application of Red List data and perhaps a trial that enables any unintended outcomes or potential conflicts to be identified. For example, actions to reduce risks to semi-natural grasslands threatened by tree encroachment should focus on management of existing grasslands, rather than conversion of threatened forest types to increase the area of grasslands. In such cases, management priorities should be guided by the status of both the semi-natural system and others in its vicinity. Anthropogenic ecosystem types, or semi-natural types with low conservation values should generally be excluded from spatial conservation planning, e.g. for protected areas, unless their management affects the viability of nearby natural or semi-natural ecosystems. If particular anthropogenic or semi-natural ecosystem types are excluded from a systematic Red List assessment, their status should be designated as 'Not Evaluated' (see Section 2 Categories of the IUCN Red List of Ecosystems).

It is important to consider the role of other tools in the management of anthropogenic and semi-natural ecosystems, especially those assigned to the 'Not Evaluated' category in the RLE. All such ecosystems should be included within ecosystem accounts for a given area of assessment (UNCEEA, 2021), and this will enable their extent and condition to be tracked, even though their risk of collapse remains unevaluated.

3.1.3. The influence of scale on assessment outcomes

The RLE risk assessment protocol was designed to be flexible for application at multiple spatial scales and with a range of data types (Rodríguez et al., 2015; Keith et al., 2015; Keith et al., 2013). However, there are practical limits to the spatial, temporal and thematic scales of units that can be assessed (see Glossary), and within these limits the assessment outcomes are sensitive to scale. Assessments of ecosystem types that are too broadly or narrowly defined, or failure to implement methods or standardisation procedures could lead to scale mismatches, incomparable assessments across different areas and times, or invalid assessment outcomes (Keith et al., 2013). It is also important to consider dependencies between different dimensions of scale. For example, assessments of ecosystem units defined at a fine thematic scale (i.e. fine level of classification) will require high spatial resolution of mapping to distinguish their distribution from other ecosystem units within the classification. The points below discuss frequently raised interpretive issues and the range of measures in the RLE protocol to address the influence of scale:

1. **Size of area of assessment**. The RLE criteria may be applied to assess the risk of ecosystem collapse within geographic areas of different sizes. Areas of assessment may vary in size from global extent to much smaller sub-global areas, such as biogeographic regions, countries and many sub-national jurisdictions. Some sub-global assessments will work within areas defined by ecologically arbitrary boundaries (e.g. national borders), and therefore will consider only parts of the global distribution of some ecosystem types, in some cases representing a very small portion of their global extent. Related scenarios include assessments within very small areas, such as small islands or local government areas. Assessments of different ecosystem types within these small areas may result in similarly high threat categories for all ecosystem types. This occurs because the resulting threat categories are more strongly reflecting the small area of

assessment, rather than the properties of individual ecosystem types. The risk of ecosystem collapse will always be greater within a small part of the range of an ecosystem type than across its entire distribution. Therefore, high threat categories are not unrealistic in such circumstances and no adjustments should be made to the criteria, thresholds or categories. Section 4.1 Area of assessment elaborates on the area of assessment. Appendix 4 offers further guidance on using risk assessment outcomes to set priorities for ecosystem protection, regulation, management and restoration where the overall threat categories fail to distinguish subtle differences in risks between ecosystem types within the same small area of assessment.

2. **Thematic scale of assessment units**. Thematic scale (classification level or strength; Hermoso et al., 2013) refers to the number and internal heterogeneity of units in an ecosystem classification (Glossary). The thematic scale of ecosystem types affects the outcomes of assessments because of averaging effects and fixed threshold effects.

   Averaging effects become more prominent in broadly defined units. For example, a set of related ecosystem types may return assessment outcomes that range from Critically Endangered (CR) to Least Concern (LC) categories, but if lumped together, the overall status of the combined unit may be Vulnerable (VU). This is because high rates of decline in some of the finer units are averaged out by lower rates of decline in others, and because estimates of spatial indicators (extent of occurrence, area of occupancy) are aggregated. This 'averaged' status of the combined unit masks the fact that it contains more threatened subunits recognisable at finer thematic scales. Therefore, the internal heterogeneity within the unit of assessment should be described (Section 4.2 Describing the unit of assessment), including any recognised subunits that may have a different status if assessed separately.

   Fixed threshold effects are specific to criterion B and are expressed at both very coarse and very fine thematic scales (i.e. broadly and narrowly defined units). At very coarse thematic scales, few or none of the assessment units are likely to fall within the spatial thresholds that define threatened categories under criterion B. Conversely, at very fine thematic scales, more assessment units are likely to fall within the spatial thresholds that define threatened categories under criterion B. A taxonomic analogue can be imagined by comparing assessments of subspecies, varieties or populations with those for species, and higher taxonomic entities such as genera or families.

   These scaling effects highlight the importance of using assessment units that are defined at an appropriate thematic scale for application of the RLE criteria. Unfortunately, there is no universal quantitative means of measuring thematic scale of classifications (unlike spatial scales of maps, which can be communicated as various numerical metrics). The IUCN Global Ecosystem Typology (Section 3.1.1 Ecosystem typologies) provides very general guidance that RLE assessments should be carried out on units at levels 4–6 of the hierarchy (i.e. levels 1–3 are too coarse for RLE application). As guidance on appropriate thematic scales for ecosystem risk assessment, Table 1 lists exemplar inventories and associated Red List assessments designed to support different regulatory frameworks and conservation planning applications in national or sub-national jurisdictions where most ecosystem management is implemented.

3. **Spatial scale, map resolution and spatial metrics.** The spatial resolution, map scale or grain size (e.g. pixel size in raster maps, and minimum polygon size or vector length in vector maps) at which an ecosystem distribution is mapped can greatly affect the estimate of ecosystem extent or area. When ecosystem types are mapped at high spatial resolution, they appear to occupy a smaller extent than if mapped at lower resolution (Figure 2). This has major implications for assessing the size of a geographic distribution against fixed area thresholds, as required for assessments under criterion B in the IUCN Red List protocols for both ecosystems and species (Keith et al., 2018). For example, when maps with different spatial resolution are used to estimate spatial parameters in adjacent countries, this can limit the comparability of Red List status and bias the outcomes of assessments. Similar biases arise when maps of different resolutions are used in time series to estimate rates of change in ecosystem extent.

   To maintain consistency of assessments against the fixed thresholds for distribution size (criterion B), estimates of extent of occurrence must be made using a standard geometric method – a minimum convex polygon which encompasses all known spatial occurrences including the fragmented natural patches of an ecosystem type (Section 6.3 Applying criterion B). Similarly, estimates of the area of occupancy must be made at a standard grain size by counting the number of 10 x 10 km grid cells that contain the current mapped distribution of the ecosystem type, irrespective of the resolution of the primary map data (Section 6.3 Applying criterion B). This relatively coarse grain of assessment takes into account the spatial scale of threats, which are more influential on the risk of collapse than spatial properties of ecosystem distributions (Keith et al., 2018). It also allows a wide range of base maps to be used for assessment, including those of coarse spatial resolution. A range of tools are available to assist with upscaling and downscaling distribution data, and completing assessments under criteria A and B (Section 5 Criterion A. Reduction in geographic distribution and Section 6 Criterion B. Restricted geographic distribution).

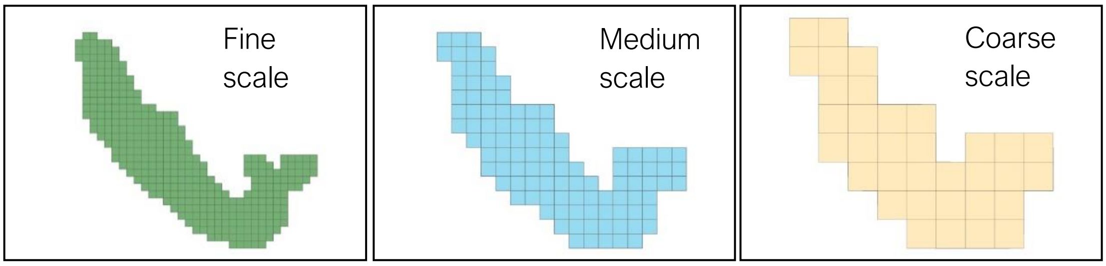


Figure 2. Distribution of the same ecosystem type mapped at three different spatial scales represented by different grain sizes (see Glossary) to demonstrate the sensitivity of area of occupancy (AOO) estimates to spatial scale. Source: Authors.
Coarser map resolutions produce larger estimates. IUCN Red List protocols for both ecosystems and species require use of standard grid sizes to avoid these geometric artefacts.

4. **Temporal scales and standard time frames for assessment.** Different ecosystems change at different rates and may appear stable at some temporal scales, while undergoing trends or fluctuations at others (Wiens, 1989; Carpenter & Turner, 2001). To ensure consistency of assessments over time frames relevant to ecosystem management, while also accommodating legacies and lags of historical events, the RLE criteria assess ecosystem change over standard time frames that represent trends over present, future and historical time scales (Section 3.3.1 Time frames). Present and future time frames are set at 50 years to balance the need to diagnose trends with reasonable certainty (requiring long time frames) with the need for timely responses to adverse trends. Historical time frames are included to accommodate the effects of ecological lags in assessments.

Table 1. Examples of ecosystem typologies and similar classifications supporting national ecosystem risk assessments for various conservation planning and regulatory applications. Source: Adapted from Keith et al., 2015.

Jurisdiction	*Application*	*Assessment unit*	*Reference*
European Union	Habitats Directive 92/43/EEC (European Commission)	Habitat type. 'Plant and animal communities as the characterising elements of the biotic environment, together with abiotic factors operating together at a particular scale.'	Council of the European Commission (1992)
Germany	Red List of biotopes (Federal Environment Agency)	Biotope. 'Habitat of a community of fauna and flora living in the wild.'	Riecken et al. (2006, 2009)
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.'	Kontula & Raunio (2009, 2019)
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.'	Lindgaard & Henriksen (2011); NBIC (2018)
Italy	Red List of Ecosystems (Ministero dell'Ambiente e della Sicurezza Energetica)	Ecosystems. Units of potential natural vegetation in different ecoregions, characterised by biogeographic and physiognomic features, as well as structurally dominant plants.	Blasi et al. (2023); Capotorti et al. (2023)
Colombia	Colombian ecosystems Red List (Pontificia Universidad Javeriana and Conservación Internacional-Colombia)	Ecosystems. 'Ecological assemblages differentiated by climatic zoning and azonal/intrazonal differences resulting from the interaction between climate, topography and soils.'	Etter et al. (2020)

Jurisdiction	*Application*	*Assessment unit*	*Reference*
Venezuela	National Red List of Ecosystems (Provita)	Major vegetation types for national assessment; satellite-derived land types for sub-national assessments.	Rodríguez et al. (2010)
Canada	State threatened species and ecosystems legislation (Manitoba Conservation and Water Stewardship Department)	Ecosystem. 'A dynamic complex of plant, animal and microorganism communities and their non-living environment interacting as a functional unit.'	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.'	Commonwealth of Australia (2000); Keith (2009); Nicholson et al. (2015)
South Africa	National biodiversity legislation (South African National Biodiversity Institute)	Ecosystem. 'A dynamic complex of animal, plant and micro-organism communities and their non-living environment interacting as a functional unit.'	Republic of South Africa (2004); Driver et al. (2012); Skowno & Monyeki (2021)
Congo Basin	Regional Red List of forest ecosystems in the Congo Basin (Wildlife Conservation Society)	Forest ecosystems. 'Unique forest types determined by phenology, climate regime, flooding dynamics and biogeographical zone.'	Shapiro et al. (2021)
China	National Red List of forest ecosystems (Chinese Academy of Science)	Forest formations. An 'assemblage of plant communities with the same dominant species in the dominant stratum.'	Guo et al. (2018); Chen et al. (2020)
Myanmar	Myanmar National Ecosystem Assessment (Collaboration of government, non-government organisations and academic institutions)	Ecosystem types based on a hierarchical typology consistent with the IUCN Global Ecosystem Typology (Realm, Biome, Ecosystem Functional Groups and Ecosystem type).	Murray et al. (2020)

3.2. Ecosystem collapse

To estimate risk – the probability of an adverse outcome over a specified time frame – it is necessary to define the endpoint of ecosystem decline, the point at which an ecosystem is considered collapsed. The definition of the endpoint to ecosystem decline must be sufficiently discrete to permit an assessment of risk, but sufficiently general to encompass diverse causes, mechanisms and pathways of ecosystem decline and the broad range of contexts in which risk assessments are needed.

Within the *IUCN Red List of Ecosystems Categories and Criteria*, "an ecosystem is Collapsed when it is virtually certain (Table 4) that its defining biotic or abiotic features are lost from all occurrences, and the characteristic native biota are no longer sustained" (IUCN, 2016; Keith et al., 2013).

3.2.1. Characteristics and pathways of ecosystem collapse

Collapse is a transformation of identity, a loss of defining features, and/or replacement by a different ecosystem. An ecosystem is collapsed when all occurrences lose defining biotic or abiotic features, no longer sustain the characteristic native biota, and have moved outside their natural range of spatial and temporal variability in composition, structure and/or function. This can be illustrated by the familiar 'marble' model of state and transition theory (Figure 3) and by key examples, including the Aral Sea (Box 2) and other published cases (e.g. Scheffer et al., 2001; Bergstrom et al., 2021; Keith et al., 2023b).

Unlike species, ecosystems do not disappear; rather they transform into novel ecosystems with different characteristic biota and mechanisms of organisation (Hobbs et al., 2006; Keith et al., 2015; Keith et al., 2013). The novel systems may retain some of the characteristic biota of the collapsed systems that they replace, but the abundance of those species, their interactions or ecological functions are altered and they may lose the ability to recruit. Thus, collapse may occur either when most of the characteristic native biota are lost from the system, or when key functional components that perform key roles in ecosystem organisation are greatly reduced in abundance and lose the ability to recruit, i.e. loss of key properties and collapse may occur long before assemblage extinction when the last characteristic species disappears from the last ecosystem occurrence (Gaston & Fuller, 2007).

Transitions to collapse may be gradual, sudden, linear, non-linear, deterministic or highly stochastic. These include regime shifts (Scheffer et al., 2001), but also other types of transitions that may not involve reinforcing feedbacks (Bergstrom et al., 2021; Keith et al., 2023b). Ecosystem collapse may in theory be reversible – given a long-time frame, or via the reintroduction of characteristic biota and/or the restoration of ecosystem function – but in many systems recovery will not be possible. In practice, reversal of ecosystem collapse rarely occurs, either because: (i) some components of the reference state no longer exist; (ii) threatening processes that drove ecosystem collapse are still operating; (iii) there are ecological barriers, homeostasis or long lags that impede reversal of collapse; (iv) restoration actions are technically infeasible or prohibitively expensive; or (v) ongoing intervention is essential, yet rarely sustained to prevent the system converging on a novel equilibrium that differs from the reference state.

The dominant dynamic in an ecosystem will depend on abiotic or external influences (e.g. weather patterns or human disturbance), internal biotic processes (e.g. competition, predation or epidemics), historical legacies (e.g. climatic history, extinction debts or exploitation), and spatial context (e.g. whereabouts, size and dispersion of distribution). An ecosystem may thus be driven to collapse by different threatening processes and through multiple pathways. Trophic cascades (Estes et al., 2011), loss of foundation species (Diamond, 2007), environmental degradation (UNEP, 2001; Keith et al., 2023b), and climatic forcing (Grebmeier et al., 2006) are common pathways to ecosystem collapse. Symptoms of collapse may differ depending on the characteristics of the ecosystem, the nature of threatening processes and the pathways of decline that these generate.

In Red List assessments, the risk of ecosystem collapse is measured by quantifying the transition beyond a bounded threshold in one or more state variables (indicators) that define the identity of the ecosystem (Figure 3). The RLE protocol has flexibility to allow thresholds of collapse to be expressed in appropriate terms for very different kinds of ecosystems (see Section 4.2.7 Describing collapsed states). The development of a conceptual model (Section 4.2.5 Conceptual models) for specific ecosystem types or groups of functionally similar ecosystem types can help to identify the pathways, expected symptoms and indicators of ecosystem collapse.

Ecosystems that have already collapsed and for which time series data exist for relevant state variables provide a strong evidence base for understanding and measuring trajectories towards collapse. It will often be possible to infer characteristics of collapse and informative state variables from local occurrences of an ecosystem type that have lost defining features, even if the majority of the ecosystem remains extant. Major changes in functionally similar ecosystems can also provide guidance for defining the symptoms of collapse in systems of interest.

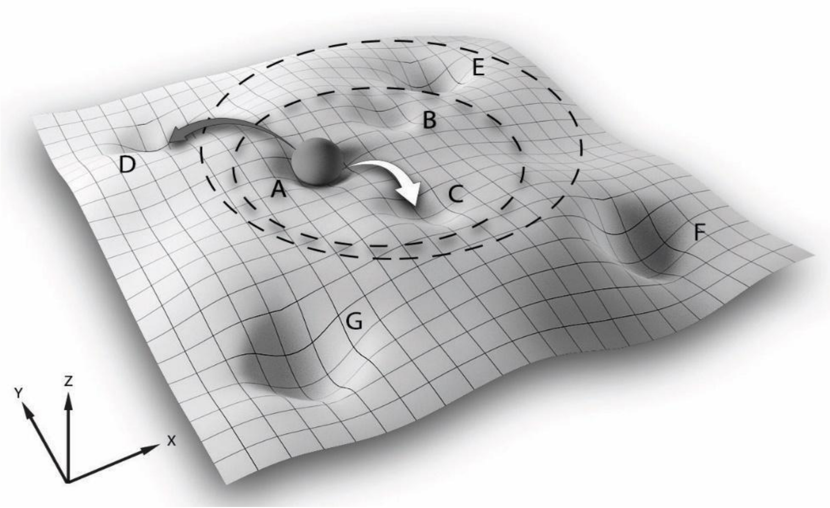


Figure 3. Generalised schematic illustrating the interpretation of ecosystem collapse in a state and transition framework. Source: Keith et al., 2015.
States A-G are defined by two state variables represented on the X and Y axes. The vertical axis (Z) represents potential for change. The two broken lines 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. Progression along different pathways of collapse is assessed with variables X and Y, or other ecosystem-specific diagnostic variables that reflect the loss of characteristic native biota and function. The outer broken line represents an alternative interpretation of ecosystem collapse in which state E is included within natural variation of the ecosystem type (see Section 3.2.2 Uncertainties in units and endpoints).

Box 2. Ecosystem collapse in the Aral Sea.

The **Aral Sea** – the world's fourth largest continental water body – is fed by two major rivers, the Syr Dar'ya and Amu Dar'ya (Aladin & Plotnikov, 1993). Its characteristic native biota includes freshwater fish (20 species), a unique invertebrate fauna (> 150 species) and shoreline reed beds, which provide habitat for waterbirds, including migratory species (Keith et al., 2013). Hydrologically, the sea was approximately stable during 1911–1960, with inflows balancing net evaporation (Micklin & Aladin, 2008). Intensification of water extraction to support expansion of irrigated agriculture led to shrinkage and salinisation of the sea. By 2005, only 28 aquatic species (including fish and invertebrates) were recorded, reed beds had dried and disappeared, the sea had contracted to a fraction of its former volume and surface area, and salinity had increased tenfold (Micklin & Aladin, 2008).

Consistent with the definition of ecosystem collapse, these changes suggest the Aral Sea has undergone a transformation of identity, lost many of its defining features (aquatic biota, reed beds, waterbirds, hydrological balance and brackish hydrochemistry) and has been replaced by novel ecosystems (saline lakes and desert plains). Under this interpretation, collapse occurred before the volume and surface area of standing water declined to zero. Although the exact point of ecosystem collapse is uncertain, time series data for several variables are suitable for defining a functional reference state (prior to onset of change from 1960) and a bounded threshold of collapse, assuming this occurred sometime between 1976 and 1989 when most biota disappeared (Keith et al., 2013).

The choice of available variables for assessing the status of the ecosystem will depend on how closely they represent the ecosystem's defining features, the quantity and quality of the data, and the sensitivity of alternative variables to ecological change. Of those listed above, fish species richness and abundance may be the most proximal biotic variable to the features that define the identity of the Aral Sea ecosystem. Sea volume may be a reasonable abiotic proxy, because volume is functionally linked with salinity, which in turn mediates persistence of the characteristic freshwater/brackish aquatic fauna. Sea surface area is less directly related to these features and processes, but can be readily estimated by remote sensing and may be useful for assessment when data are unavailable for other variables.

Collapse of the Aral Sea ecosystem may or may not be reversible. While it may be possible to restore the hydrological regime over a small part of the former sea (Micklin & Aladin, 2008), some components of the characteristic biota are apparently extinct (e.g. the Aral salmon, *Salmo trutta aralensis*), preventing reconstruction of the pre-collapse ecosystem. Images: © NASA

3.2.2. Uncertainties in units and endpoints

Risk assessment relies on the definition of an adverse outcome, typically a discrete endpoint or event that affects the asset under evaluation. The implementation of risk assessment confronts uncertainties in two key areas: the definition of the asset itself, and the definition of the endpoint. The boundary which delineates an ecosystem type may be uncertain due to imperfect knowledge of natural variability within the ecosystem, continuous patterns of variability with other ecosystems, and changes in ecosystem classification through time, as well as uncertainties associated with mapping distributions (Keith et al., 2013). Defining ecosystem collapse is also subject to uncertainty which can affect the estimation of spatial and functional symptoms of collapse (Figure 4). All applications of the *IUCN Red List of Ecosystems Categories and Criteria* should consider these sources of uncertainty and discuss them in the assessment documentation. Examples of how uncertainties can be dealt with through the assessment process are described below, acknowledging that uncertainties in spatial and functional systems are often related.

Uncertainty in spatial symptoms

During decline, an ecosystem may transition to a collapsed state(s) in some parts of its distribution before others. In areas where these transitions have occurred, the ecosystem may be described as 'locally collapsed'. Spatially, an ecosystem is considered collapsed when all extant occurrences of the ecosystem have collapsed (i.e. area of occupancy = 0 10 x 10 km grid cells and extent of occurrence = 0 km^2). To quantify past declines in distribution and declines in function, assessors must identify where the ecosystem type is currently extant, and where it was previously extant (within the time frame of assessment) and is now in a collapsed state. Similarly, to quantify future declines in distribution and function, assessors must project the area in which the ecosystem will collapse during the future time frame of the assessment. All of these estimations and projections involve uncertainties. Epistemic uncertainty (i.e. uncertainty due to a lack of knowledge, as opposed to inherent uncertainty due to variability in the system; Regan et al., 2002) exists due to a range of measurement and classification errors:

1. Thematic uncertainties (vagueness) related to boundaries between units or categories e.g. when an ecosystem type is considered to have moved outside of its natural bounds of variability, or where spatial boundaries between different ecosystem types are placed (Payet et al., 2013).
2. Measurement error due to imperfect measurements or mapping techniques resulting in area estimates that are not precisely repeatable and randomly fluctuate (Elith et al., 2002; Olofsson et al., 2014; Fuller et al., 2003).
3. Systematic error due to mapping methods that consistently produce biased area estimates (Congalton & Green, 2008).
4. Classification errors that result in misclassification of pixels or polygons in a distribution map, generally termed omission or commission errors (Congalton & Green, 2008; Foody, 2011).
5. Errors of scale where the grain size at which an ecosystem is mapped results in area estimates that are dependent on the scale at which they are mapped (Hartley & Kunin, 2003; Gaston & Fuller, 2009).

Uncertainty in functional symptoms and thresholds of collapse

A collapsed ecosystem may be replaced by a novel ecosystem with strongly contrasting features. When grasslands replace forests, the change in vegetation structure is readily detected by a range of proximal and remote sensing methods. In other cases, ecosystems may lose defining features and collapse, but the novel system may resemble the antecedent one, making symptoms of collapse more difficult to detect. Burns et al. (2015) describe an example of a forest ecosystem characterised by biota associated with large old trees. When timber harvest reduces densities of large old trees below a critical level, some elements of the characteristic native biota are lost from the system. This includes birds and mammals that nest or shelter in tree hollows, and invertebrates that live under loose bark and in deep leaf litter beds. After such transitions, the novel ecosystem still retains a forest structure, albeit one characterised by smaller trees and lacking biota associated with large trees (secondary in comparison to primary forest). Similarly, Barrett & Yates (2015) described collapse of a species-rich shrubland as the elimination of groups of plant species by a soil-borne disease. The novel ecosystem replacing the antecedent one was a structurally similar, but compositionally and functionally different shrubland. These and other examples illustrate uncertainties in delineating extant and collapsed states, which depend on the features of the antecedent ecosystem, the pathway of collapse and the features of the novel ecosystem. Sources of uncertainty include:

1. Definition of reference ecosystem states, and the natural variability within those.
2. Definition of collapsed ecosystem states, which represent critical deviations from natural variability. Transition points from original to novel ecosystems are inherently uncertain but can be estimated within plausible bounds (Figure 4). The first value represents no doubt that the ecosystem has collapsed, whereas the second is a plausible value based on observations or inferences.
3. Variation in collapsed states related to different pathways of decline. Catastrophic threats (e.g. transformational events such as intensive land-use change, sea-level change, transformational disturbance events) may cause total functional and spatial collapse of the ecosystem. Other threats, such as environmental degradation or the spread of invasive species may cause different functional changes in characteristic biota. These different pathways of collapse should be reflected in the documentation (as part of the definition of collapse; see Section 4.2.7 Describing collapsed states).
4. Uncertainty in the measurement of variables representing ecosystem function and collapse. As with spatial variables, measurement error in functional variables may affect the assessment of ecosystem collapse through random errors or systematic bias.

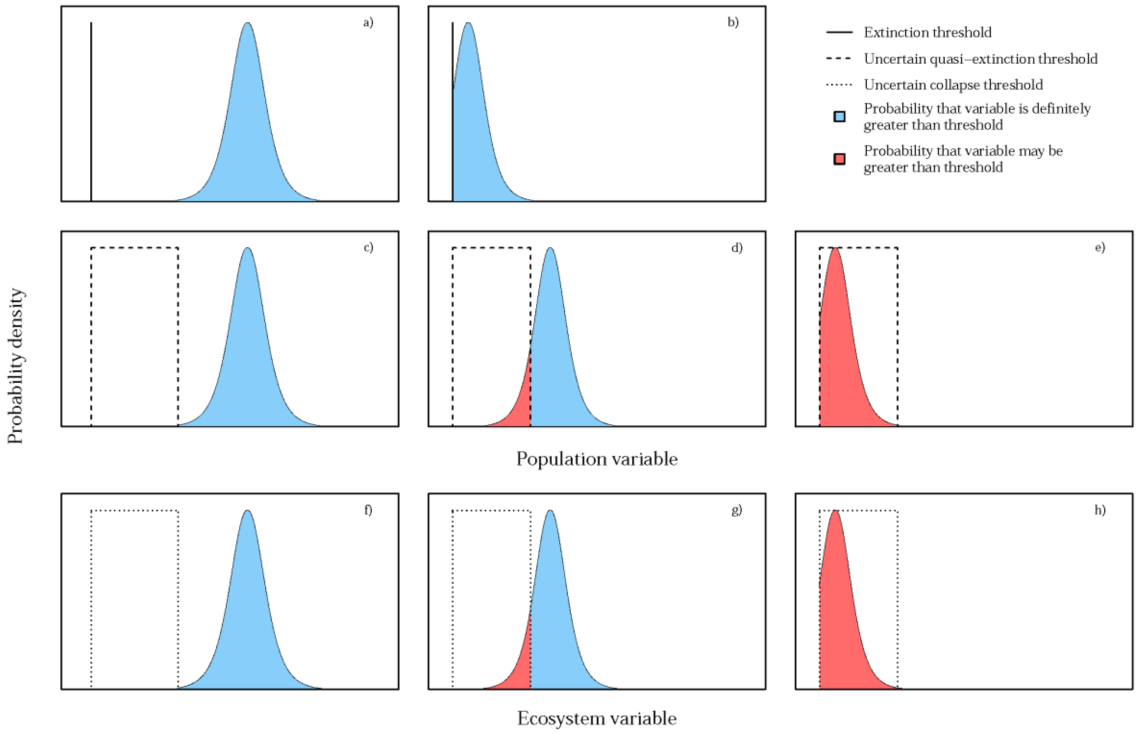


Figure 4. Probability density functions for the population and ecosystem variables that measure proximity to the thresholds that define species extinction (a, b), species quasi-extinction (c–e), and ecosystem collapse (f–h). Source: Adapted from Keith et al., 2013. For species, the population threshold that defines extinction is known with certainty (e.g. zero abundance, described by the vertical line in (a) and (b)). In practice, Population Viability Analyses are calibrated on a quasi-extinction threshold higher than the extinction threshold, to account for prediction and management uncertainty. A lower bound on the value of extinction (zero abundance), and a putative upper bound for the value of quasi-extinction can be depicted as a dashed box (c–e). For ecosystems (f–h) the x-axis could represent key features or processes (e.g. spatial distribution, number of species, water quality). The bounded definition of collapse is analogous to the definition of quasi-extinction in species. The width of the dashed box represents uncertainty in the collapse definition. The blue area represents the probability that the ecosystem is definitely extant, whereas the red area represents the probability that the ecosystem may be extant.

3.3. Risk assessment protocol

The RLE protocol comprises five rule-based criteria for assessing risks to ecosystems (Table 2). Risks to ecosystems may be caused by a variety of threatening processes that are expressed through different symptoms of ecosystem collapse (Keith, 2015).The RLE protocol groups symptoms of ecosystem collapse into four major types and identifies the corresponding mechanisms that link the symptoms to the risk that an ecosystem will lose its defining features (Figure 5). Two of the four mechanisms produce distributional symptoms: (A) declines in distribution, which reduce carrying capacity for dependent biota; and (B) restricted distribution, which predisposes the system to spatially explicit threats. Two other mechanisms produce functional symptoms: (C) degradation of the abiotic environment, reducing habitat quality or abiotic niche diversity for component biota; and (D) disruption of biotic processes and interactions, resulting, for example, in the loss of mutualisms, biotic niche diversity, or exclusion of some component biota by others. Interactions between two or more of these four contrasting mechanisms may produce additional symptoms of transition towards ecosystem collapse. Multiple mechanisms and their interactions may be integrated into a simulation model of ecosystem dynamics to produce quantitative estimates of the risk of collapse (E). These five groups of symptoms form the basis of the RLE criteria. An ecosystem type under assessment should be evaluated using all of the criteria for which data are available. The overall risk status of the ecosystem type is assigned as the highest category of risk obtained through any criterion.

A summary table of the current *IUCN Red List of Ecosystems Categories and Criteria* is provided in Appendix 1.

Table 2. Purpose of the IUCN Red List of Ecosystems criteria.

Criterion		*Purpose*
A	Reduction in geographic distribution	Identifies ecosystems that are undergoing declines in area, most commonly due to threats resulting in ecosystem loss and fragmentation.
B	Restricted geographic distribution	Identifies ecosystems with small distributions that are susceptible to spatially explicit threats and catastrophes.
C	Environmental degradation	Identifies ecosystems that are undergoing environmental degradation.
D	Disruption of biotic processes or interactions	Identifies ecosystems that are undergoing loss or disruption of key biotic processes or interactions.
E	Quantitative analysis that estimates the probability of ecosystem collapse	Allows for an integrated evaluation of multiple threats, symptoms and their interactions.

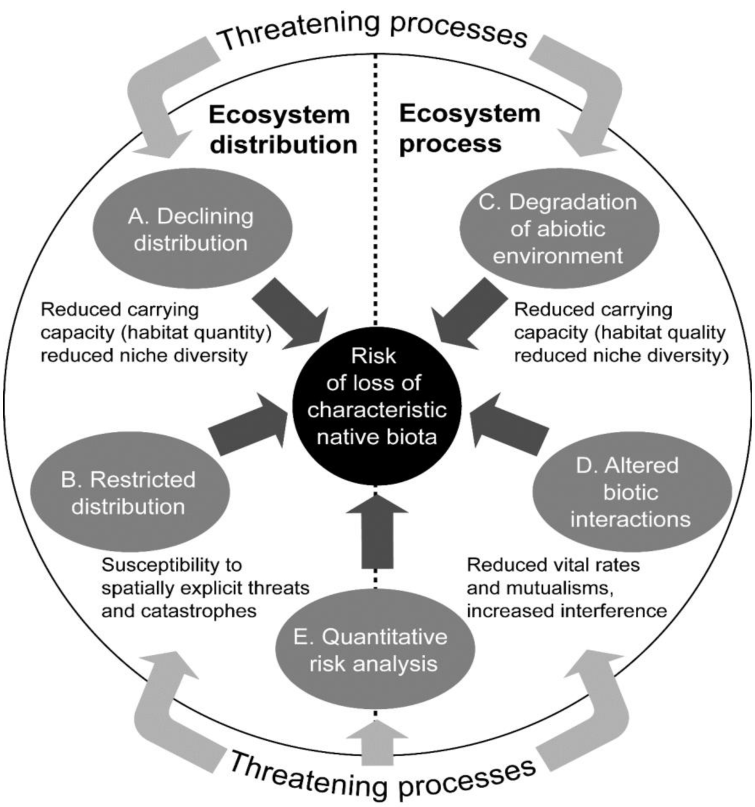


Figure 5. Mechanisms of ecosystem collapse and symptoms of collapse risk.
Source: Keith et al., 2013.

3.3.1. Time frames

The criteria assess declines over four specified time frames (Figure 6): (i) the recent past (last 50 years); (ii) any 50-year period including the recent past, present and future; (iii) the future (next 50 years); and (iv) the historical past (since 1750). The 'recent past' time frame encompasses the last 50 years. Although different ecosystem types may change at different rates depending on disturbance regimes and life histories of their functionally dominant biota, 50 years is a pragmatic assessment time frame that is sufficiently recent to capture current trends but long enough to distinguish directional change from natural variability. The RLE protocol assumes that declines over this time frame are indicative of future risk irrespective of cause.

Assessment of future declines requires predictions of changes over the next 50 years or any 50-year period including the present and future (Figure 6). Past declines may provide a basis for such predictions by extrapolation, but other information may support predictions and inferences about rates of future decline even when the ecosystem is currently stable. Extrapolations from the past into the future should be done by fitting statistical or other suitable models with explicitly stated and defensible assumptions about the pattern of future change (e.g. accelerating, constant, decelerating). Plausible alternative models of change should be explored where appropriate, but a constant proportional rate of decline is often a reasonable default assumption (Section 7.3.5 Calculating extent of degradation). Projections of future declines may also be made with the aid of simulation models, where the mechanisms of ecosystem dynamics are sufficiently known to support the structure of such models (e.g. Burns et al., 2015; Bland et al., 2017).

Historical declines are assessed over a long-time frame that extends from the onset of industrialisation to the present day, when there was a substantial intensification of pressures on natural and semi-natural ecosystems. The onset of industrialisation commenced at different times around the world, however, a consistent baseline year should be used for historical assessments to ensure consistency of assessment outcomes. Therefore, a notional reference date of 1750 provides an indicative bound that marks the historical baseline recommended for Red List assessment.

Assessments of historical declines are essential for ecosystems containing biota with long generation lengths and slow population turnover (Mace et al., 2008). They are also essential for foundation species with short generation lengths which may have suffered extensive historical declines (e.g. oyster reefs: Kirby, 2004; Beck et al., 2011). Even where future rates of decline abate, historical reductions in distribution or function may create legacies that predispose an ecosystem to additional risks and reduce its ability to absorb adverse changes (Folke et al., 2004). Extinction debts and collapse debts are expressions of such legacies.

To assess declines in ecosystem properties over historical time frames, the present-day ecosystem state needs to be compared with that at the historical baseline date (notionally 1750). This may be done with the aid of models or inferences that draw evidence from relictual present-day distributions and/or historical information, including that available in historical documents, maps, oral histories and expert elicitation.

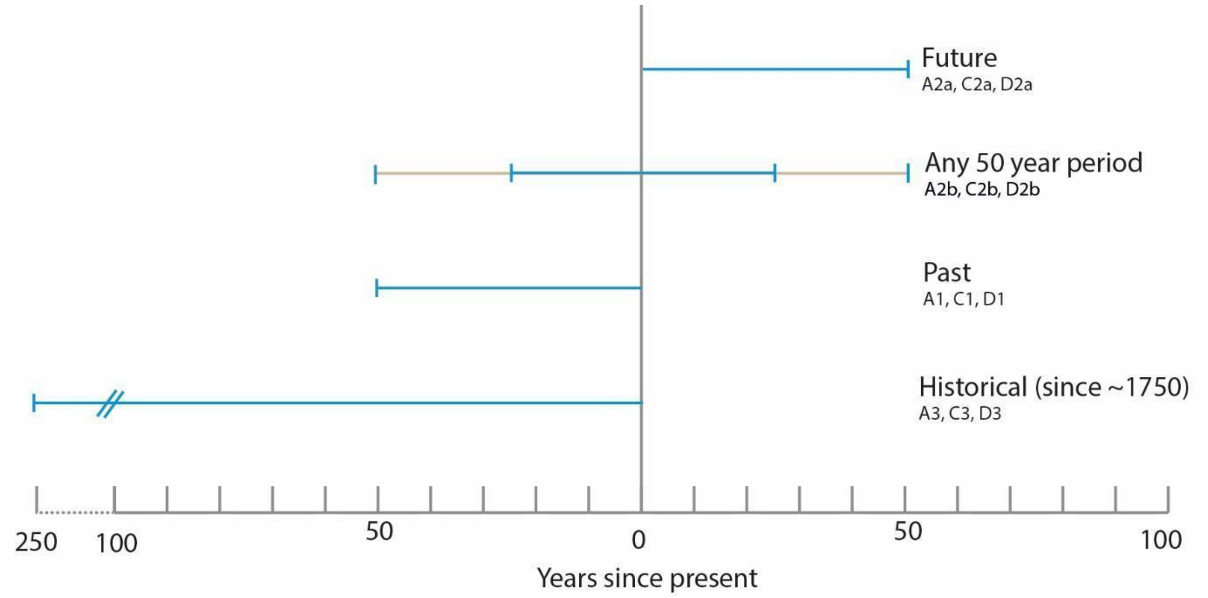


Figure 6. Time frames for assessment of change under criteria A, C and D.
Source: Adapted from Keith et al., 2013.
Note: the historical baseline is notionally fixed at the year 1750 and the illustration here at 250 years before present assumes the present at year 2000.

3.3.2. Decline thresholds

The ordinal categories of risk (Section 2 Categories of the IUCN Red List of Ecosystems) are delimited by thresholds defined in the *IUCN Red List of Ecosystems Categories and Criteria* (Appendix 1). The rationale for the criteria and ordinal categories is grounded in theory (Keith et al., 2013). However, the threshold values that delimit categories are based partly on theoretical considerations and partly on utilitarian considerations (Keith et al., 2015). Theory provides a qualitative basis for ordered thresholds for decline, but offers limited guidance for setting their absolute values. The purpose of these decision thresholds is to rank ecosystems in informative ordinal categories of risk, rather than estimate precise probabilities of collapse. Consequently, for criteria A, C and D, threshold values were set at relatively even intervals for current and future declines in ecosystem distribution or function (Vulnerable: 30%, Endangered: 50%, Critically Endangered: 80%). The range of thresholds between 0 and 100% seeks to achieve an informative rather than highly skewed ranking of ecosystems among categories. The lowest threshold for a threatened ecosystem type (30%) recognises that evidence of an appreciable decline in ecosystem distribution or function is necessary to support listing in a threatened category. These thresholds are consistent with thresholds for population reduction in The IUCN Red List of Threatened Species (IUCN, 2001, 2012). Thresholds for historical declines are higher (A3, C3, D3: 50%, 70%, 90%) because time frames for assessment are longer.

Declines within 5–10% of thresholds for the Vulnerable category may warrant listing as Near Threatened, although there are no quantitative thresholds for this category (Section 2 Categories of the IUCN Red List of Ecosystems). For example, an ecosystem type with an extent of occurrence of 50,000 to 55,000 km^2 that qualifies for at least one of the three subcriteria of criterion B could qualify for listing as Near Threatened. An ecosystem type with a decline in an abiotic variable of 20% to 30% relative severity and 100% extent could qualify as Near Threatened under subcriteria C1 or C2.

3.3.3. Standards of evidence and dealing with uncertainty

Achieving a robust and repeatable assessment for an ecosystem type requires extensive data, often from disparate sources. The categories and criteria were specifically designed to allow the inclusion of various data types from a range of sources, but the onus is on the assessor to critically evaluate whether data quantity and quality are sufficient to support a determinate outcome of an assessment. For guidance on this evaluation, assessors are referred to the principles adopted by the Intergovernmental Panel on Climate Change for consistent treatment of uncertainty (Mastrandrea et al., 2010). In summary, key principles include:

1. Evaluating the type (Table 3), amount, quality and consistency of evidence (summary descriptors: 'limited', 'medium' or 'robust').
2. Evaluating the degree of agreement between different sources of evidence (summary descriptors: 'low', 'medium', or 'high').
3. Providing a traceable account describing the evaluation of evidence and agreement.
4. Evaluating the likelihood (Table 4) of alternative categories as outcomes of an assessment.
5. Communicating the uncertainty in the outcomes of an assessment by reporting the most likely category, as well as categories that represent plausible upper and lower bounds of the assessment outcome (Section 4.4.1 Dealing with uncertainty).

The standard of evidence for the RLE must be sufficient to support inferences that:

1. Some categories (LC, NT, VU, EN or CR) are 'very unlikely' outcomes of assessment (i.e. probability < 10%, Table 4). If no category can be excluded with that level of certainty, then the status should be assigned as Data Deficient (DD).
2. The plausible bounds of assessment outcomes include all categories necessary to ensure that collectively they are 'very likely' to encompass the true status (i.e. probability > 90%, Table 4). If all categories (LC–CR) are within the plausible bounds, then the status should be assigned as Data Deficient (DD).
3. The best overall status (i.e. categorisation of an ecosystem) is more likely than any alternative categorisation and within the plausible bounds.
4. All categorisations of overall status in the Collapsed category (CO) are 'virtually certain' (i.e. > 99% certain, Table 4). Where this is not the case and CO is the most likely category, the best overall status should be assigned to Critically Endangered (CR), and CO reported as the upper bound of the assessment outcome.

Table 3. Descriptors for types of evidence will typically support inferences during an assessment. Source: IUCN, 2001, 2012.
These apply to quantitative variables (such as rates of change in distribution) and binary inferences (such as whether or not there is a continuing decline in distribution).

Descriptor	*Explanation*
Observed	Information that is directly based on well-documented records of all known occurrences of the ecosystem (IUCN SPC, 2024).
Estimated	Information that is based on calculations that may include statistical assumptions about sampling, or biological assumptions about the relationship between an observed variable and the variable of interest (e.g. relationship between an index of abundance and the number of mature individuals; IUCN SPC, 2024). These assumptions should be stated and justified in the assessment documentation. Estimation may also involve interpolation in time to calculate the variable of interest for a particular time step (e.g. a 50-year reduction in distribution based on observations of distribution 40 and 60 years ago).
Inferred	Information that is based on indirect evidence and on variables that are indirectly related to the variable of interest, but in the same general type of units (IUCN SPC, 2024). Inferred values rely on more assumptions than estimated values. For example, inferring disruption of biotic interactions from catch statistics not only requires statistical assumptions (e.g. random sampling) and biological assumptions (about the relationship of the harvested section of the population to the total population), but also assumptions about trends in effort, efficiency, and the spatial and temporal distribution of harvest in relation to the population. Inference may also involve extrapolating an observed or estimated quantity from known ecosystem occurrences to calculate the same quantity for other occurrences. Whether there are enough data to make such an inference will depend on how large the known occurrences are as a proportion of the whole distribution, and the applicability of threats and trends observed in the known occurrences to the rest of the ecosystem.
Projected	Same as estimated, but the variable of interest is extrapolated in time towards the future (IUCN SPC, 2024). Projected variables require a discussion of the method of extrapolation (e.g. justification of the statistical assumptions or the ecosystem model used) as well as the extrapolation of current or potential threats into the future, including their rates of change.

Table 4. Calibrated language for describing quantified uncertainty.
Source: Mastrandrea et al., 2010.
It can be used to express a probabilistic estimate of a quantity, a binary inference or an assessment outcome (e.g. a magnitude of change in distribution, whether or not there has been a change, whether the status of an ecosystem is within a given range). Likelihood may be based on statistical or modelling analyses, elicitation of expert views, or other quantitative analyses. The categories defined in this table can be considered to have 'fuzzy' boundaries (Kauffman & Gupta, 1991).

Term	*Likelihood of outcome (probability)*
Virtually certain	99–100%
Very likely	90–100%
Likely	66–100%
More likely than not	50–100%
About as likely as not	33–66%
Unlikely	0–33%
Very unlikely	0–10%
Exceptionally unlikely	0–1%

3.3.4. Quantitative data and expert knowledge

The Red List criteria require calculations based on quantitative estimates of variables such as areas and rates of change in biotic and abiotic features of ecosystems. Quantitative estimates of these variables are ideally based on systematic measurements acquired in a sampling design that permits valid statistical inferences across the geographic range of the ecosystem type under evaluation. In reality, relevant and useful evidence on ecosystem status includes a range of incomplete, patchy and subjective observations.

Scientific judgements are required to decide which pieces of information meet the standard of evidence to infer the status of an ecosystem. For example, a particular forest ecosystem may never have been mapped at an appropriate resolution to quantify the proportional change in its distribution over the past 50 years, as required to assess criterion A1. Despite the lack of formal data, multiple independent experts are unanimous in their opinion, based on anecdotal observations, that at least 50% of the ecosystem distribution has been converted to pasture in the past 50 years. The high degree of certainty about the rate of decline should inform a Red List assessment – the status of the forest ecosystem is **likely** (Table 4) to be at least Endangered and is **very unlikely** to be Least Concern.

Qualitative expert knowledge may also add value to quantitative measurements. For example, data from repeat surveys of fish in a marine reef ecosystem may indicate a 32% decline in abundance over the past 50 years, but experts are unanimous that surveys are limited to the most exploited reefs and, based on anecdotal observations, they infer that fish abundance is likely to have remained 'approximately stable' on many unexploited reefs. If fish abundance was assumed to decline by 0–20% on these unexploited reefs (a worst-case interpretation of 'approximately stable'), the overall average decline across all reefs is estimated as 15–25%. In this case, a status of Least Concern or Near Threatened may be **more likely than not** (i.e. more likely than Vulnerable and other categories), despite the estimate based on formal data.

Both examples above show how expert knowledge can improve inferences about Red List status compared to assessments based exclusively on measurements. However, expert opinion is notoriously unreliable, subject to various social biases, influenced by a range of experiential and behavioural factors, and expert performance is very difficult to predict (Burgman, 2015). Use of expert opinion to estimate quantities required for Red List assessments must therefore be subject to standards and procedures that reduce the risks of errors and bias. The recommended standards for Red List assessments and their rationale are given in Table 5. These are formalised in structured elicitation protocols such as IDEA ('Investigate', 'Discuss', 'Estimate' and 'Aggregate'; Hemming et al., 2018 – see Table 5 for key features). While the application of structured elicitation protocols is strongly recommended, it is important that sufficient time is devoted to training and calibration of experts in the method, as well as allowing sufficient time for experts to consult relevant sources (publications, datasets, etc.) to inform their individual estimates. This can be very time-consuming, but expert elicitations that aim to elicit estimates of indicators for dozens of Red List assessments or more are unlikely to produce reliable outcomes due to the effects of expert fatigue (Burgman, 2015).

Table 5. Recommended standards and procedures for expert elicitation and handling uncertainty. Source: Based on Burgman, 2015, Hemming et al., 2018.

Step	*Recommended approach*	*Rationale*
Selecting experts	People who are: (i) reasonably familiar with the ecosystem type, the area in which it occurs, and the processes that affect it; and (ii) frequently seek feedback and consider uncertainty in their advice. Seek diversity and avoid homogeneity in selecting expert groups.	Expertise declines dramatically outside an individual's specialisation or experience. Basic familiarity is relevant but expert performance appears independent of experience and standing. Experts who seek frequent feedback on their judgements, subdue overconfidence and consider uncertainty perform well.
Number of experts	A minimum of three. More is better.	Estimated values averaged across multiple experts outperform individual estimates, including those by the most experienced experts.
Information provided to experts	Available data and qualitative observations relevant to the quantity being estimated, including sources, contextual information including definitions of terms and details of sampling design and methods. Inform experts of the elicitation process (steps 1–3) and the qualities associated with high performance (see Burgman, 2015).	Provides a common base of information on which to base an estimate. Raises awareness of cognitive factors associated with accurate expert estimates and reduces linguistic uncertainties.

Step	*Recommended approach*	*Rationale*
Elicitation step 1	Each expert is asked to estimate a required quantity (e.g. decline in distribution over past 50 years) independently of (i.e. without conferring with) others. Four values are required for each estimate in the following order: - a plausible upper bound - a plausible lower bound - a best estimate - the probability that the true value lies between the upper and lower bound	Independent estimates for each expert avoid social elicitation biases associated with dominant personalities, seniority, perceptions of peers, etc.
Elicitation step 2	Experts are provided with all estimates without names of those who made them. In plenary, they are given an opportunity to discuss the reasons considered in coming to an estimate.	Exchange of ideas and factors relevant for consideration, additional data and observations, supports more informed estimates.
Elicitation step 3	Each expert is given the opportunity to revise their estimates from step 1 independently of other experts, in the light of discussion in step 2.	Reduces social biases, while incorporating additional information.
Synthesis	The best estimates are averaged across all experts. Upper and lower bounds are converted to 90% confidence interval, assuming a probability distribution and transformation that are appropriate to the quantity estimated (Speirs-Bridge et al., 2010), and averaged across assessors.	Central tendency of multiple independent estimates is more likely to be close to the true values than any other expert estimate. Upper and lower bounds based on means exclude extreme outlying values.
Assessment against Red List criterion	The Red List status is calculated for the best estimate, upper and lower bounds, producing a bounded estimate of the threat category for that criterion.	Uncertainty (represented by upper and lower bounds) is propagated transparently through the assessment, allowing reporting of the best estimate of threat category, as well as plausibly optimistic and pessimistic categories, given the available information.

4. Assessment process

Assessing an ecosystem type against the *IUCN Red List of Ecosystems Categories and Criteria* is a sequential process. All components must be completed before submission of the assessment (Figure 7).

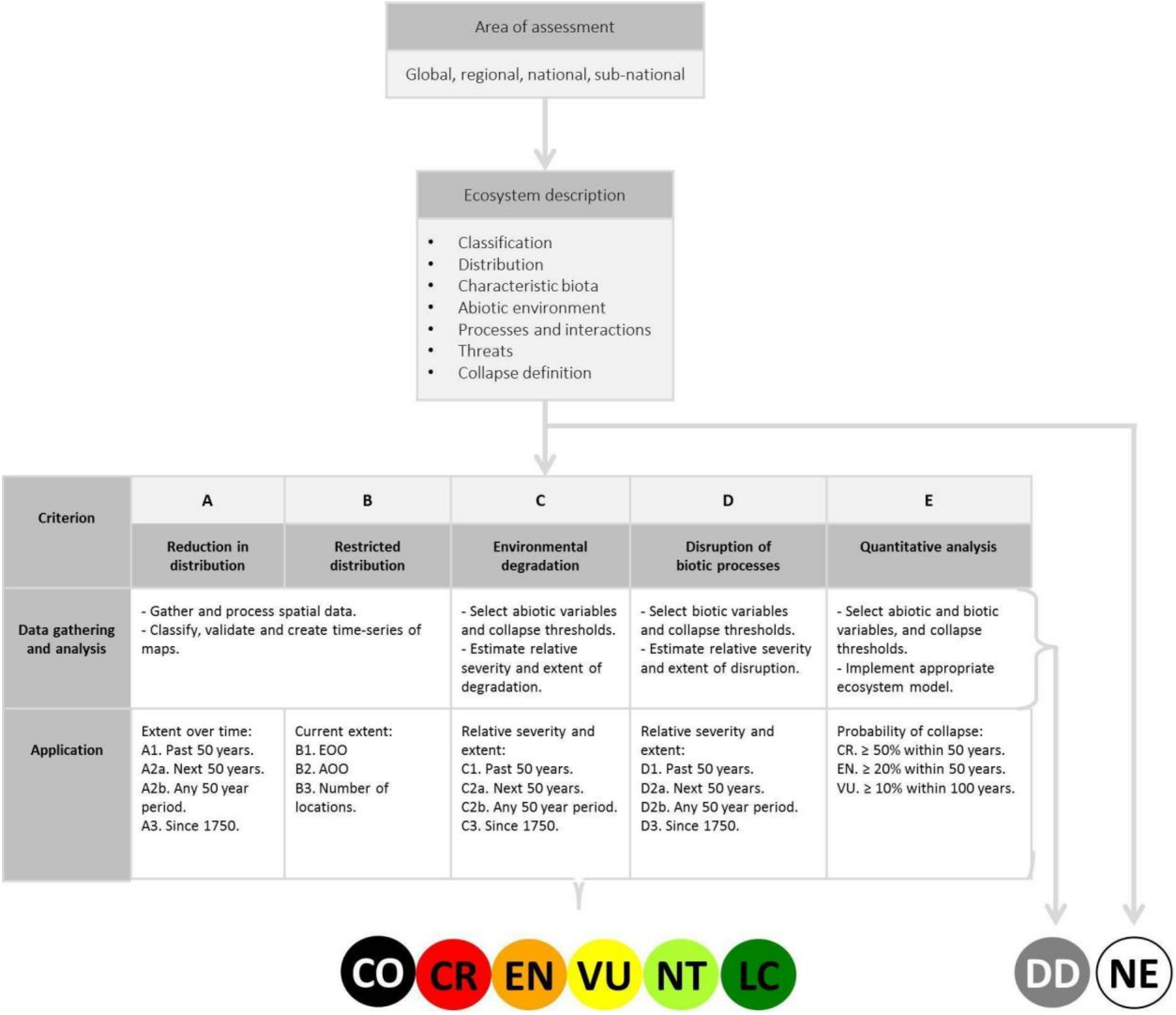


Figure 7. Process for assessing the risk of collapse of an ecosystem type. Source: Authors.

4.1. Area of assessment

Red List of Ecosystems (RLE) assessments may be undertaken within different geographic areas. Global assessments consider all occurrences of an ecosystem type throughout the world and are based on units of assessment that are referrable to levels 4 or 5 of the IUCN Global Ecosystem Typology (Keith et al., 2022). This is essential for the set of broadly defined ecosystem types that will form the global RLE, and for informing international biodiversity targets and conservation strategies. For example, the assessment of the coral reefs of the Western Indian Ocean can be considered an assessment of global coral reef ecosystem types in that region (Obura et al., 2021).

4.1.1. Sub-global Red List assessments

Sub-global RLEs assess the status of ecosystem types within only a part of the world, often based on units of assessment that are finer than those required for global assessment (i.e. within level 6 of the Global Ecosystem Typology). They are typically undertaken to support ecosystem management within regional, national or local jurisdictions. Thus, their area of assessment is typically defined by political (continental, national or state) or ecoregional boundaries (ocean basins or catchments). Examples of sub-global RLEs include those for Germany (Riecken et al. 2009), Western Australia (DEC, 2007), Finland (Kontula & Raunio, 2009, 2019), Venezuela (Rodríguez et al., 2010), Austria (Essl & Egger, 2010), Norway (Lindgaard & Henriksen, 2011; NBIC, 2018), South Africa (Skowno & Monyeki, 2021), New Zealand (Holdaway et al., 2012), El Salvador (Crespin & Simonetti, 2015), Colombia (Etter et al., 2015; 2017), Chile (Pliscoff, 2015), Myanmar (Murray et al., 2019) and Italy (Blasi et al. 2023). Sub-global assessments that are confined to national borders are termed 'national Red Lists.'

For sub-global assessments, it will usually be appropriate to assess ecosystem types of finer thematic resolution (i.e. at level 6 of the IUCN Global Ecosystem Typology) than those for global assessments (at level 4 or 5), as sub-global assessments will usually require finer detail to support ecosystem management decisions. Thus, a national RLE may have a larger number of more finely divided ecosystem types for a given area, compared to a global-level RLE assessment (Section 3.1.3 The influence of scale on assessment outcomes). When the sub-global area of assessment is similar to, or smaller than the extent of occurrence (EOO) or area of occupancy (AOO) thresholds for the Vulnerable category, listing of ecosystem types under criterion B will depend solely on meeting the subcriteria (see Section 3.1.3 The influence of scale on assessment outcomes, Section 6 Criterion B. Restricted geographic distribution and Appendix 4).

The same ecosystem type may be assigned to different risk categories in sub-global and global assessments. Differences in status depend on whether the area of sub-global assessment includes the entire global distribution of the ecosystem type in historic, present and projected future time frames (Section 3.3.1 Time frames), as well as any differences in data used in the assessments and the time at which the assessments were carried out. Where a unit of assessment is: (i) endemic to a single country, and (ii) referrable to level 5 or 6 of the IUCN Global Ecosystem Typology, the national Red List assessment may also meet the requirements of a global assessment and its national and global status should be the same.

General guidance for sub-global assessments includes:

1. The area of assessment (e.g. political boundaries) must be clearly defined and supported with maps or other spatial data.
2. Comprehensive description of the assessment unit (ecosystem type) is required. Each unit should be attributed at least to Ecosystem Functional Groups (EFGs, level 3) in the IUCN Global Ecosystem Typology. Where attributions to two or more EFGs are plausible (e.g. in intermediate cases), all should be documented and the most likely attribution identified.
3. No modifications of the categories or criteria A, B, C, D or E are required when making sub-global assessments of ecosystems. Therefore, all thresholds, time frames, definitions and data requirements remain unchanged for sub-global applications of the RLE.
4. For sub-global assessments, each ecosystem type assessed must be clearly tagged with the area of assessment (point 1, above). For example, outcomes of national RLE assessments that consider only the national extent of the types should be tagged as national Red List status. Nationally endemic ecosystem types (i.e. known to occur entirely within the borders of one country) may be eligible for derivation of global status from national status. A process is under development to determine which assessments of endemic national types are suitable for adoption as global assessments. It is not always possible for teams focussed on a particular country to assess the portion of the type that lies in a neighbouring country (i.e., there may not be data or expertise to do this). Where possible, neighbouring countries should work together on a single RLE assessment for shared types. Ecosystem types judged to be endemic to a single country would have RLE assessments that are automatically assumed to be global in scope.

4.2. Describing the unit of assessment

To ensure repeatable application of the *IUCN Red List of Ecosystems Categories and Criteria*, detailed description and definition of the assessment units is an essential component of the assessment process. The description and assessment are based on a comprehensive review of available information about the ecosystem type under consideration. The description of an ecosystem type must provide contextual information on its classification. It must clearly describe four elements that define the ecosystem type: the spatial distribution (Section 4.2.2 Spatial distribution); characteristic native biota (Section 4.2.3 Characteristic native biota); abiotic environment (Section 4.2.4 Abiotic environment); and processes and interactions (Section 4.2.5 Conceptual models). It must further describe the threats (Section 4.2.6 Threats) and collapsed states (Section 4.2.7 Describing collapsed states).

Assessors should address all key elements required in the description for ecosystem types (Table 6). They should also justify why the unit selected for assessment is recognised as a separate ecosystem type, by identifying the key features that distinguish it from adjacent or similar ecosystem types. Information supporting the description of the ecosystem type should be included in the assessment documentation, and will be assessed by peer review. It is expected that all submissions to the global RLE will include relevant supporting information including a list of key references, maps, geographic coordinates, exemplar photographs and any other information that will facilitate repeatability of the assessment. These submissions will be openly accessible on the IUCN Red List of Ecosystems website (www.iucnrle.org).

Table 6. Key elements requiring description for ecosystem types.

Elements	*Description*
Classification	Cross-references to relevant ecological classifications: - IUCN Global Ecosystem Typology (especially level 3). - The source classification from which the assessment unit was derived. - Other relevant ecological classifications (e.g. other systems in wide local use, classifications systems for neighbouring areas).
Spatial distribution	Describe distribution and extent: - Accurate spatial distribution data. - Estimates of area for each time. - Time series, projections (past, present, future).
Characteristic native biota	Identify defining biotic features: - Diagnostic native taxa and their relative abundance in comparison to other ecosystem types. - Functional components of characteristic biota and their roles in the focal system compared to others. - Limits of spatial and temporal variability in the ecosystem biota. - Exemplar photographs.
Abiotic environment	Identify defining abiotic features: - Text descriptions and citations for characteristic states or values of abiotic variables. - Graphical descriptions of abiotic variables. - Exemplar photographs.
Processes and interactions: - among biota - between biota and environment	Describe key ecosystem drivers: - Text descriptions and citations. - Conceptual model. - Exemplar photographs.
Threats	Describe major threats and impacts on ecosystem functioning: - Text descriptions of mechanisms, severity, extent and trends with relevant citations. - Diagnosis based on IUCN Threats Classification Scheme. - Exemplar photographs.
Collapse definition	Describe ecosystem-specific collapsed state(s) and threshold(s).

4.2.1. Classification

All assessment units must be cross-referenced to the IUCN Global Ecosystem Typology (Keith et al., 2020, 2022; https://global-ecosystems.org/). This is IUCN's global standard for ecosystem classification and is included within the United Nations Family of Statistical Classifications for international reporting. It is also the reference classification for ecosystem assets in the United Nations System for Environmental Economic Accounting – Ecosystem Accounting (UNCEEA, 2021).

The IUCN Global Ecosystem Typology is a hierarchical classification with six levels. The top three levels distinguish major groups of ecosystems by their functional properties and the three lower levels distinguish different compositional expressions of ecosystem types within each functional group (Keith et al., 2022; Table 7). RLE assessments may be carried out on ecosystem units defined at a range of thematic scales, but should only be carried out on units classified in levels 4–6 of the typology. Ecosystem classification units defined at levels 1–3 of the typology are used for contextualisation and comparison, but are too broad for use as assessment units and therefore the Red List criteria should not be applied to them.

Table 7. Definitions of hierarchical levels within the IUCN Global Ecosystem Typology.
Source: After Table S3.1 in Keith et al., 2022.

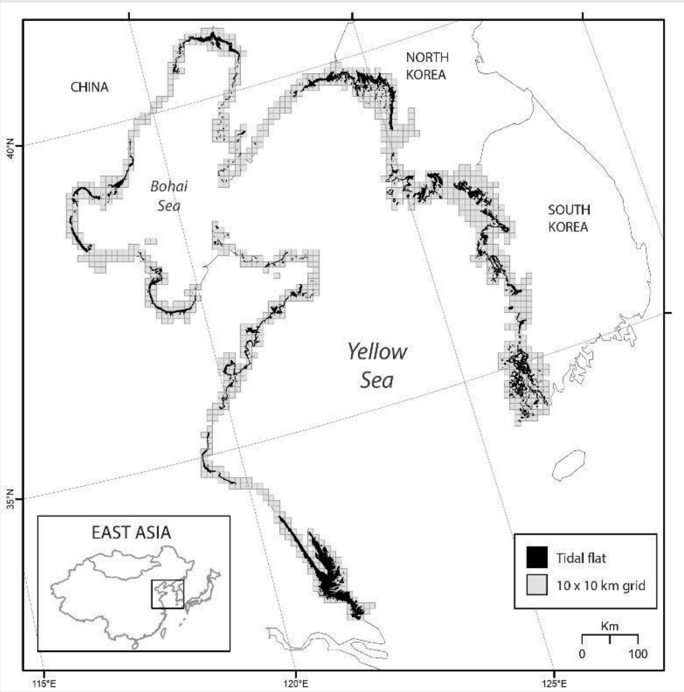
Level	*Definition*
1 Realm	One of five major components of the biosphere that differ fundamentally in ecosystem organisation and function: terrestrial, freshwater, marine, subterranean, atmospheric. See https://global-ecosystems.org/explore.
2 Functional biome	A component of a realm united by one or a few common major ecological drivers that regulate major ecosystem functions and ecological processes, derived from the top-down by subdivision of realms (level 1), e.g. https://global-ecosystems.org/explore/biomes/T4.
3 Ecosystem Functional Group	A group of related ecosystems within a biome that share common ecological drivers promoting convergence of ecosystem properties that characterise the group. Derived from the top-down by subdivision of biomes, e.g. https://global-ecosystems.org/explore/groups/T4.2.
4 Regional ecosystem subgroups	An ecoregional expression of an Ecosystem Functional Group derived from the top-down by subdivision of Ecosystem Functional Groups (level 3). They are proxies for compositionally distinctive geographic variants that occupy different areas within the distribution of a functional group.
5 Global ecosystem type	A complex of organisms and their associated physical environment within an area occupied by an Ecosystem Functional Group. Global ecosystem types grouped into the same Ecosystem Functional Group share similar ecological processes, but exhibit substantial difference in biotic composition. They are derived from the bottom-up, either directly from ground observations or by aggregation of sub-global types (level 6).
6 Sub-global ecosystem type	A subunit or nested group of subunits within a global ecosystem type, which therefore exhibit a greater degree of compositional homogeneity and resemblance to one another than to those in other global ecosystem types (level 5). These represent units of established classifications (e.g. at national level), in some cases arranged in a sub-hierarchy of multiple levels, derived directly from ground observations.

Ecosystem Functional Groups (level 3 of the Global Ecosystem Typology) enable generalisations and predictions about ecosystems with similar functional properties because they are groups of related ecosystem types within a biome characterised by common ecological drivers that promote convergence of ecosystem properties. Attribution of individual ecosystem types to functional groups therefore supports the diagnostic process in Red Listing by providing access to general information about ecosystem responses to environmental change (Keith et al., 2022). For this reason, and for reporting general patterns in risks to different ecosystem groups, assessors should attribute each individual assessment unit (ecosystem type) to an Ecosystem Functional Group based on the available descriptive information (https://global-ecosystems.org/). Global Red List assessments must be based on units at level 4 or 5. Sub-global assessments may be based on units at levels 4–6.

4.2.2. Spatial distribution

Information on the spatial distribution of an ecosystem type is best represented by maps or inventories of localities. They can be derived from remote sensing, biophysical distribution models, field observations or a combination of all three (Box 3). The spatial features of some ecosystems (such as pelagic environments) are inherently dynamic over relatively short time frames, so spatial distributions can only be described at very coarse levels of resolution. Given the diversity of methods and maps available, an important aspect of the description is to justify why a particular spatial dataset is an adequate representation of the ecosystem distribution. Further information on clearly describing the spatial distribution of an ecosystem type is provided in Section 5 Criterion A. Reduction in geographic distribution and Section 6 Criterion B. Restricted geographic distribution. Assessors are encouraged to deposit the ecosystem map in a suitable online repository.

Box 3. Distribution map of the Yellow Sea tidal flat ecosystem.



The distribution of the **Yellow Sea tidal flat** ecosystem was mapped by applying a peer-reviewed remote sensing classification method to Landsat Archive satellite imagery (Murray et al., 2012, 2014; Murray & Fuller, 2015). The classified map has an overall accuracy of > 94% when assessed using a confusion matrix, a widely implemented method for assessing the accuracy of classified maps (Murray et al., 2014; Congalton & Green, 2008). Estimates of the area of the ecosystem type for criterion A are derived from the distribution of the ecosystem (black), whereas the area of occupancy (AOO) is determined by counting the number of 10 x 10 km cells in which > 1 km^2 of the ecosystem type occurs (Murray et al., 2015).

4.2.3. Characteristic native biota

The concept of characteristic native biota is central to ecosystem risk assessment and is therefore an important component of the description of the ecosystem type (Box 4). The characteristic native biota include the genes, populations, species, assemblages of species and their key interactions that: (i) compositionally distinguish an ecosystem type from others (diagnostic components); and (ii) are centrally relevant to ecosystem dynamics and function, such as ecosystem engineers, trophic or structural dominants, or functionally unique elements (functional components). The diagnostic components of characteristic native biota should demonstrate a level of compositional uniqueness and identify functionally important elements. In general, the description need not include exhaustive species inventories.

Characteristic native biota are crucial in the diagnosis of ecosystem collapse because they define part of the 'identity' of the ecosystem type. Thus, the loss of characteristic native biota or processes in which they play a functional role signals a transformation of identity, collapse of the ecosystem type and replacement by a novel system.

Characteristic native biota may be defined in terms of taxonomy or functional traits (e.g. guild composition, trait spectra, structural features such as architecture of trees or corals) and excludes alien species and uncommon or vagrant species that contribute little to ecosystem function. Examples of characteristic native biota include species that are endemic or near-endemic to the ecosystem type, predators that structure the animal communities, tree species that create microclimates in their canopies or at ground level, reef-building corals and oysters that promote niche diversity for cohabiting fish and macro-invertebrates, nurse plants and those that provide sites for predator avoidance, burrowing animals, guilds of nitrogen fixers, key dispersal agents responsible for movement of biota or resources, peat-forming plants, detritivore guilds, and flammable plants that promote recurring fires.

Box 4. Describing characteristic native biota.
Source: Adapted from Appendix S2 in Keith et al., 2013.

Raised Bogs, Germany

This ecosystem type is characterised by vegetation dominated by peat mosses (e.g. *Sphagnum magellanicum*, *Sphagnum fuscum*) and insectivorous plants like sundew (*Drosera* sp.). The dominance by peat mosses together with geomorphic and hydrological processes distinguishes raised bogs from other ecosystem types. Other typical species for raised bogs in Germany are the vascular plants bog-rosemary (*Andromeda polifolia*) and cranberry (*Vaccinium oxycoccos*), the butterfly species *Boloria aquilonaris* (Cranberry Fritillary), the moth *Carsia sororiata* (Manchester Treble-Bar) and the ground beetle *Agonum ericeti* (Blab et al., 1995).

Great Lakes Alvar, North America

This ecosystem type is characterised by a variable physiognomy, from open perennial (rarely annual) grassland or shrubland and nonvascular pavement (5–25% herb and or shrub cover) to dense grassland or shrubland (> 25%) with scattered evergreen needleleaf (more rarely broad-leaf deciduous) trees (Reschke et al., 1999; Catling & Brownell, 1995). Species composition contains a mix of tallgrass prairie graminoids and forbs and sub-boreal to boreal shrubs and trees. Key dominants and differentials include the perennials *Schizachyrium scoparium*, *Sporobolus heterolepis*, *Danthonia spicata* and *Deschampsia caespitosa*; less commonly with *Sporobolus neglectus*, *Sporobolus vaginiflorus*, and *Panicum philadelphicum*. Key shrubs, when present, are *Juniperus communis*, *J. horizontalis*, *Dasiphora fruticosa ssp. floribunda* and *Rhus aromatica*. Trees, when present, include *Thuja occidentalis*, *Picea glauca*, *Pinus banksiana*, and *Abies balsamea* (in more northern sites) and *Juniperus virginiana*, *Quercus macrocarpa* or *Quercus muehlenbergii* (more southern sites).

Giant Kelp Forests, Alaska

Alaskan kelp forests are structurally and functionally diverse assemblages. They are characterised by species of brown algae in the Order Laminariales including *Nereocystis luetkeana, Laminaria groenlandica*, *Alaria fistulosa*, *Agarum fimbriatum* and *Thalassiophyllum* sp. (Steneck et al., 2002). These create a complex and dynamic layered forest architecture up to 15 m tall that provides substrate, shelter and foraging resource for a diverse fauna assemblage of epibenthic invertebrate herbivores and pelagic vertebrate predators (Steneck & Watling, 1982; Estes et al., 2009). Characteristic invertebrates include urchins, *Strongylocentrotus franciscanus*, *S. purpuratus* and *S. droebachiensis*, limpets, and starfish, *Solaster* spp. Fish, including the Pacific cod (*Gadus macrocephalus*) and rock greenling (*Hexagrammos lagocephalus*), are important predators that depend directly or indirectly on the ecosystem (Reisewitz et al., 2006). Characteristic mesopredators include sea otters (*Enhydra lutris*), harbour seals (*Phoca vitulina*), Steller sea lions (*Eumetopias jubatus*) and northern fur seals (*Callhorinus ursinus*). Steller's sea cow (*Hydrodamalis gigas*), now extinct, was a functionally unique herbivorous member of the vertebrate assemblage (Domning, 1972). Large pelagic predators are also important components of the ecosystem, including killer whales

(*Orcinus orca*) and over 15 species of great whales including sperm (*Physeter macrocephalus*) and fin whales (*Balaenoptera physalus*). Kelp forests are generally separated geographically by continental land masses or deep sea. The Alaskan kelp forests are continuous with those of California, but differ compositionally in their more diverse assemblage of macroalgae, including *Macrocystis pyrifera*.

Shallow under-ice benthic invertebrate communities, Antarctica. Source: Clark et al., 2015.

Under-ice communities are typically composed of a mix of sessile suspension feeders and mobile macro-invertebrates, elements of which are reminiscent of deep-sea fauna but occur at depths as shallow as a few metres. Sessile fauna include Porifera (Demospongia, Hexactinellida, Calcaria), Gorgonaria, Pennatularia, Alcyonaria, Stolonifera, Hydrozoa, Actiniaria, Bryozoa, Brachiopoda, Polychaeta, and both solitary and colonial Ascidiacea (Dayton, 1990; Gili et al., 2006). Dominance of some sessile taxa is known to occur at local scales, such as by sponges (Dayton, 1979, McClintock et al., 2005) and ascidians (pers. obs). Fauna with fragile skeletons are distinctly abundant, which is thought to be due to the lack of durophagous (skeleton crushing) predators (Aronson & Blake, 2001) but may also relate to low wave energy in ice-protected coasts. Mobile invertebrates occur with these sessile fauna or can dominate in some areas. Commonly occurring taxa include Echinodermata (Echinoidea, Asteroidea, Ophiuroidea, Holothurioidea) and Peracarida (Amphipoda, Isopoda, Tanaidacea, Mysidacea, Cumacea) both of which are very successful in Antarctica and can exhibit high abundances or dominance of particular species. Other common mobile epifauna include Pycnogonida, Ostracoda, Caridea, Teleostei, Prosobranchia, Opisthobranchia, Polyplacophora, Bivalvia and Nemertinea (Dayton, 1990; Gili et al., 2006). Many of these are symbionts and use sessile invertebrates as habitat, including specialised predators such as nudibranches, asteroids and gastropods. Some fauna, such as the pycnogonids display gigantism, where individuals grow to much larger sizes than related taxa in non-polar regions (Chapelle & Peck, 1999).

4.2.4. Abiotic environment

Descriptions should identify salient abiotic features that influence the distribution or function of an ecosystem type, define its natural range of variability, sustain its characteristic native biota, and differentiate it from other systems. For terrestrial ecosystems, salient abiotic features may include substrates, soils and landforms, as well as ranges of key climatic variables, while those of freshwater and marine ecosystems may include key aspects of water regimes, light regimes, tides, currents, climatic factors, and physical and chemical properties of the water column (Box 5).

Box 5. Describing the abiotic environment.

Gnarled Mossy Cloud Forest, Lord Howe Island, Australia. Source: Auld & Leishman, 2015.

The Lord Howe Island Gnarled Mossy Cloud Forest occurs on the summit plateau and ridgetops of two mountains on Lord Howe Island. The climate is temperate, and sea level parts of the island have a mean annual temperature of 19.2°C, ranging from 17–25°C in summer to 14–18°C in winter (Mueller-Dombois & Fosberg, 1998). At sea level, average annual rainfall is 1,717 mm, with a maximum of 2,886 mm and a minimum of 998 mm (Mueller-Dombois & Fosberg, 1998). Temperature decreases with altitude in the southern mountains (0.9°C for every 100 m rise in altitude; Simmons et al., 2012). Cloud forests on Pacific islands typically occur between 800 and 900 m a.s.l. (Meyer, 2011), and on Lord Howe Island, the Gnarled Mossy Cloud Forest ecosystem occurs from 750 to 875 m a.s.l. The annual rainfall in Gnarled Mossy Cloud Forest is thought to be much higher than at sea level (although this has not been quantified) and spread throughout the year (DECC, 2007). The two southern mountains (Mounts Gower and Lidgbird) obtain significant moisture from both rainfall and direct canopy interception of cloud water (horizontal precipitation or cloud stripping), and their peaks are often shrouded in cloud (Auld & Hutton, 2004). Cloud forests are characterised by increased rainfall and cooler temperatures than forest with no cloud (Jarvis & Mulligan, 2011), and this is thought to also apply to the Gnarled Mossy Cloud Forest ecosystem (Auld & Leishman, 2015).

Yellow Sea Tidal Flats, East Asia. Source: Murray et al., 2015.

The Yellow Sea is a shallow (mean depth c. 45 m), semi-enclosed sea with surrounding geography varying from mountain ranges in South Korea to low-elevation coastal plains across much of the northern and western regions (Healy et al., 2002; MacKinnon et al., 2012). As such, tidal flats in the Yellow Sea are among the largest on Earth. In areas with high tidal amplitude (macrotidal, >4 m) they may attain a width of nearly 20 km when exposed at low tide (Healy et al., 2002). A key feature of the Yellow Sea tidal flats is the seasonal switching from an erosion- to accretion-dominated system in some areas, depending on the occurrence of the monsoon season (Wang & Zhu, 1994). The ecosystem is dependent on the continuing operation of a suite of coastal processes that are focused on sediment transport and dynamics. Sediments are transported to tidal flats by coastal and tidal currents, where the deposition process is influenced by factors such as sediment texture and size, occurrence of vegetation, wave dynamics, rainfall and the composition of the benthic community, which facilitates local bioturbation, biodeposition and biotransportation (Wang et al., 2012). Storms, wind and wave action cause seaward erosion of tidal flats, and compaction and subsidence reduce their elevation, so sediment trapping and replenishment are required to offset these processes and maintain tidal flat extent. However, a feature that distinguishes tidal flats in the Yellow Sea from adjacent regions is that the tidal flat ecosystem is largely erosion-dominated, requiring ongoing sediment replenishment and transport to persist (Healy et al., 2002). Therefore, disruption of sediment provision via reduced supply from sources such as rivers, and interruption of sediment transport and deposition mechanisms, are considered the primary processes that lead to degradation of the ecosystem (Wang et al., 2012).

4.2.5. Conceptual models

A qualitative understanding of ecosystem dynamics is essential for assessing risks related to functional declines. Generic mechanisms of ecosystem dynamics can often be inferred from related systems if the ecosystem type under assessment lacks direct studies. For example, pelagic marine systems are typically dominated by trophic interactions in which elements of the main trophic levels are known, even if particular predator-prey relationships are not (Estes et al., 2009). Tree and grass dynamics in savannahs across the world are influenced by fire regimes, herbivores and rainfall, although their relative roles may vary among savannah types (Lehmann et al., 2014). All descriptions of ecosystem types should include a narrative account of ecosystem dynamics that addresses key ecological processes defining the identity and behaviour of the ecosystem type and the threats that may cause their loss or disruption.

A conceptual model of key ecosystem dynamics is required for each ecosystem type as part of its assessment. A conceptual model is a diagram of key ecosystem processes and threats, and serves four purposes. First, the creation of a conceptual model compels assessors to think through and clarify their assumptions and understanding of ecosystem processes. Second, the conceptual model provides a basis for conducting the risk assessment, by informing selection of relevant variables for assessing criteria C and D (Section 7.3 Applying criteria C and D). Third, the conceptual model is a communication tool that effectively summarises key features of an ecosystem type for risk managers, conservation practitioners, peer reviewers and the wider community. Finally, the conceptual model is useful for underpinning the development of a quantitative model for criterion E.

Two types of conceptual models are particularly useful for RLE assessments: cause-effect models and state-and-transition models (Box 6). Cause-effect models depict the interaction and dependencies among model components, such as characteristic biota, the abiotic environment and threats (Box 6a). State-and-transition models depict switches between ecosystem states due to changes in the abiotic environment or ecosystem processes (Box 6b). For example, changes in the average water level determine transitions between the degraded hypersaline and unhealthy hypersaline states in the Coorong lagoon (Appendix S2.19 in Keith et al., 2013; Lester & Fairweather, 2011, 2009).

A standard visual repertoire can help develop consistent cause-effect models (Figure 8). Characteristic biota are represented by green hexagons, the elements of the abiotic environment by blue hexagons, biotic processes by green ovals, abiotic processes by blue ovals, and threats by red rectangles. Positive, negative and hypothesised relationships can be represented by appropriate symbols. The use of arrows accompanied by plus and minus signs is discouraged. Distinct ecosystem components functioning together should form part of a compartment. For example, the Gonakier forest in Senegal (Appendix S2.7 in Keith et al., 2013) can be described by two faunal and floral compartments, driven by abiotic processes that are influenced by threats (Box 6c).

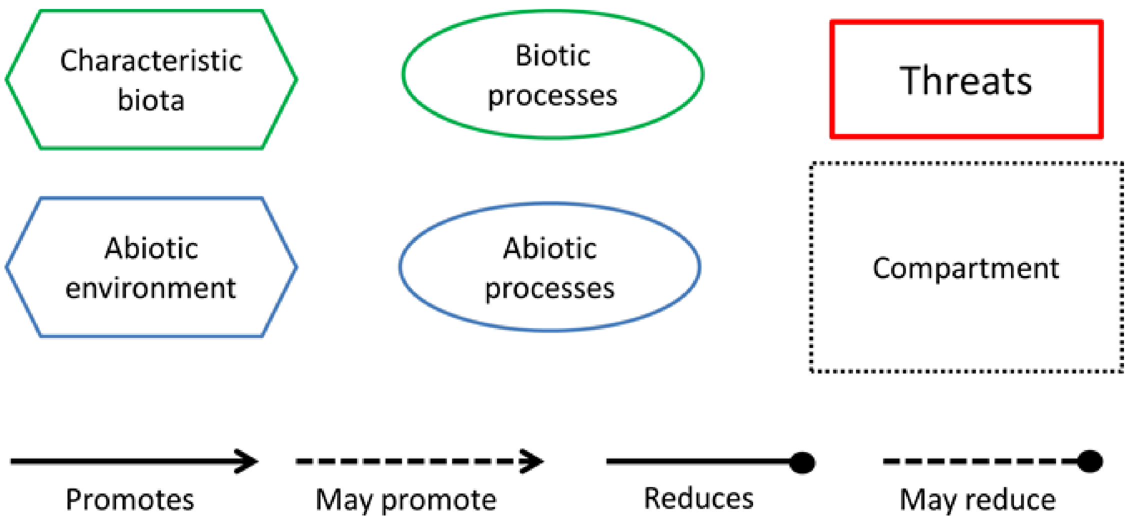


Figure 8. A common visual repertoire for cause-effect models. Source: Authors.

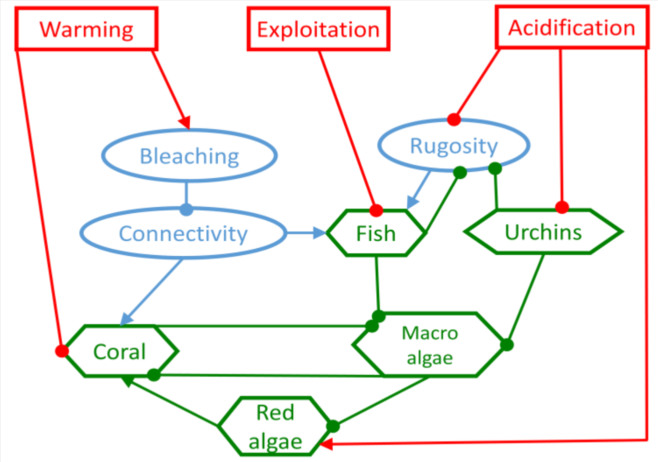
General guidelines for developing conceptual models for RLE assessments include:

1. Conceptual models of ecosystem types should be complete, unambiguous and easy to understand. They should be consistent with the narrative description of ecosystem processes and functions, and should not introduce elements which have not been described in the narrative. They should focus on processes especially relevant to the application of criteria C and D, and to the definition of the collapsed state of the ecosystem type.
2. Overly complex conceptual models should be avoided, so models will typically include fewer details than the narrative text. Assessors are encouraged to think carefully about the level of complexity and hierarchical organisation of the conceptual model, revisiting the purpose of developing a conceptual model (described above) if necessary. Overall, the least complex model covering all ecosystem processes will be the most appropriate (typically fewer than 12 elements).
3. The inclusion of processes relevant to other ecosystem types (but not to the ecosystem type of interest) is discouraged.
4. Repetition of components and relationships should be avoided.
5. Assessors may refer to the IUCN Threats Classification Scheme to review the range of potential threats. Although developed for application to individual species, aspects of this classification are also relevant to ecosystems. The inclusion of generic drivers such as human population growth or economic factors is not recommended.

Development of the conceptual model may reveal uncertainties in the understanding of ecosystem processes. It may be necessary to draft two or more alternative conceptual models to represent this uncertainty. Refining the model multiple times may help to explore and refine ecosystem processes and clarify the layout of the model. An effort should be made to reach a consensus conceptual model for the ecosystem type, using the narrative text to highlight the greatest sources of uncertainty. When assessing criterion E, it may be useful to include a second, more complex model to describe selected indicators and modelled relationships among components. Tools to assist in construction of conceptual models are in development, including a computer programme to support the development of internally consistent conceptual models. The programme will allow users to save and retrieve conceptual models for a range of ecosystems, use a common visual repertoire and evaluate the effects of threats on ecosystem processes.

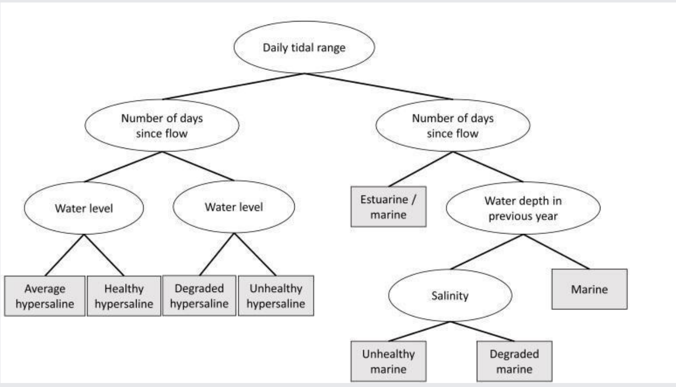
Box 6. Conceptual models representing processes and interactions.

a) Cause-effect model



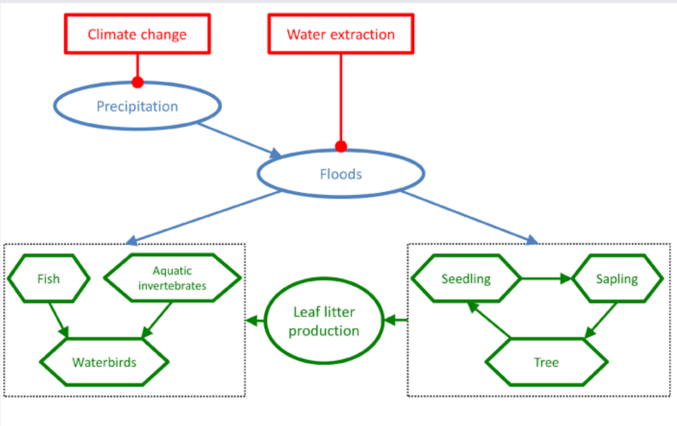
Cause-effect model of a **Caribbean coral reef** (Appendix S2.17 in Keith et al., 2013). Warming, pollution, exploitation and acidification are direct threats. Bleaching, rugosity and connectivity are key ecosystem processes. The system alters between coral and algae-dominated patches.

b) State-and-transition model



State-and-transition model of the **Coorong lagoon** in Australia (adapted from Appendix S2.19 in Keith et al., 2013). Rectangles represent alternative states of the ecosystem, ovals represent drivers of transitions between states. Average salinity determines shifts between the unhealthy marine and degraded marine states.

c) Cause-effect model with compartments



Cause-effect model of the **Gonakier forest** in Senegal (Appendix S2.7 in Keith et al., 2013). The model is composed of two compartments. Forest regeneration depends on floods, and contributes to leaf litter production. Leaf litter production in turn provides nutrients for the aquatic fauna.

4.2.6. Threats

Descriptions of ecosystem types should be accompanied by a review of threatening processes that cause ecosystem change. Describing the threats to an ecosystem type requires two elements: (i) a brief description and explanation of the primary threats causing ecosystem change; (ii) identification of threats with reference to the IUCN Threats Classification Scheme in The IUCN Red List of Threatened Species (IUCN, 2012), currently also the framework applied to ecosystems. When combined, the description of threatening processes and stresses, the threat classification under the IUCN Threats Classification Scheme, and the conceptual model for an ecosystem type will assist in identifying collapsed states and key variables for assessing change in abiotic and biotic function. The framework in Table 8 (drivers, threats and stresses) outlines how threats affect ecosystems.

Consistent terms for drivers, threats and stresses are needed for ecosystem assessment (Table 8). A direct threat for one ecosystem type or organism can be an indirect threat for another or pose no threat to other organisms. For example, unsustainable fishing will directly threaten target and by-catch species and may also have indirect effects (negative or positive) on species that prey upon, compete with or are preyed upon by targeted species. This complexity of effects requires careful consideration and definition of threats for each ecosystem type.

Table 8. Definitions of threats, drivers and stresses. Source: Salafsky et al., 2008.

Term	*Definition*	*Synonyms*
Driver	The ultimate factors, usually social, economic, political, institutional, or cultural that enable or otherwise add to the occurrence or persistence of proximate direct threats. There is typically a chain of drivers behind any given direct threat.	Contributing factors, underlying factors, root causes, indirect threats, pressures
Threat	Direct threats are the proximate activities or processes that have impacted, are impacting, or may impact the status of the ecosystem being assessed (e.g., unsustainable fishing or logging). Threats can be past (historical), ongoing, and/or likely to occur in the future. Natural phenomena are also regarded as direct threats in some situations.	Direct threats, sources of stress, pressures, proximate pressures, stressors
Stress	Stresses are the effects on ecosystem features that are impaired directly by threats (e.g. reduced abundance of keystone species, fragmentation of habitat). A stress is not a threat in and of itself, but rather a degraded condition or symptom of the target that results from a direct threat. The RLE risk protocol aims to quantify these symptoms to assess declines towards collapsed states.	Symptoms, key degraded attributes

Description of threats

A summary of the main threats currently affecting or likely to affect the ecosystem type is required supporting information for all ecosystem types. The description provides a brief explanation of the major threats (past, present and future), the drivers of those threats, and the resultant stresses or symptoms of the ecosystem. Identifying stresses is highly informative for defining collapsed states and assessing criteria C and D. The geographic extent, severity and trends of threats should also be described with relevant citations. National threats classification schemes may be used to describe threats, but assessors should report both the national designation and identify threats according to the IUCN Threats Classification Scheme. Graphs, figures and exemplary photographs are encouraged to illustrate the impact of threats on the characteristic native biota, physical environment and interactions among them. An example of a threats description is provided in Box 7.

Threats Classification Scheme

The IUCN Threats Classification Scheme (www.iucnredlist.org/technical-documents/classification-schemes/threats-classification-scheme) is currently the recommended reference classification. The scheme was developed to support The IUCN Red List of Threatened Species. It is hierarchical, consisting of three levels with increasing detail, and contains 12 main threat categories. For RLE assessment, the description of threats to an ecosystem type must correspond with threats from the IUCN Threats Classification Scheme. Attribution of the major threats affecting an ecosystem type is required as supporting information for all ecosystem types except where there are no known threats to those assigned to the Data Deficient or Least Concern categories. Assessors should diagnose and record threats to the lowest possible level in the IUCN Threats Classification Scheme.

Coding of timing, scope and severity for each major threat is not required but can be provided. If assessors decide to also record minor threats (threats affecting only a very small proportion of the distribution), then it is essential that the timing, scope and severity be described for all of the threats recorded. This will allow major and minor threats to be clearly identified for the ecosystem type and assist higher level analyses of the RLE.

Box 7. Describing threats. Source: Appendix S2.9 in Keith et al., 2013.

The **Coolibah-Black Box Woodlands of south-eastern Australia** is a flood-dependent woodland ecosystem type affected by five main threats (Appendix S2.9 in Keith et al., 2013; NSW Scientific Committee, 2004). Expansion and intensification of agricultural land use has replaced large areas of woodland with crops and pastures in recent decades (Keith et al., 2009). Furthermore, extraction of water from rivers for irrigation has altered flood regimes and their spatial extent, reducing opportunities for reproduction and dispersal of characteristic flora and fauna (Thoms & Sheldon, 2000; Thoms, 2003; Kingsford & Thomas, 1995; Kingsford & Johnson, 1998; Kingsford & Auld, 2005). Future climate change may also affect the spatial and temporal availability of water in the system. Invasive plants have spread with agricultural intensification and are reducing the diversity and abundance of native biota. Additionally, invasion of the mat-forming forb *Phyla canescens* reduces the diversity of native ground layer plants (Taylor & Ganf, 2005). This species has spread rapidly in response to altered water regimes and persistent heavy livestock grazing (Earl, 2003). Finally, overgrazing by feral goats, rabbits and domestic livestock has altered the composition and structure of the woodland vegetation, through selective consumption of palatable native ground layer plants and seedlings of trees and shrubs (Reid et al., 2011; Robertson & Rowling, 2000). These effects are most marked beneath trees and around watering points where livestock concentrate their activities.

The threats affecting this ecosystem type correspond with five threats (underlined) and their hierarchical categories in the IUCN Threats Classification Scheme:

- 2. Agriculture & aquaculture
  - 2.1 Annual & perennial non-timber crops
    - <u>2.1.3 Agro-industry farming</u>
  - 2.3 Livestock farming & ranching
    - <u>2.3.3 Agro-industry grazing, ranching or farming</u>
- 7. Natural system modifications
  - 7.2 Dams & water management/use
    - <u>7.2.3 Abstraction of surface water (agricultural use)</u>
- 8. Invasive & other problematic species, genes & diseases
  - 8.1 Invasive non-native/alien species/diseases
    - <u>8.1.2 Named species – *Phyla canescens*</u>
- 11. Climate change & severe weather
  - <u>11.2 Droughts</u>

The description of threats and stresses underpinned the selection of variables for assessing criteria C and D and clarified their link to collapse of this ecosystem type. Under criteria A and B, the ecosystem type was "assumed to have collapsed when its mapped distribution has declined to zero as a consequence of clearing for agriculture". Because flood regimes are fundamental to ecosystem dynamics and water extraction for irrigation is a major threat, median daily river flow was identified as a suitable variable for assessing environmental degradation under criterion C.

4.2.7. Describing collapsed states

Ecosystem collapse is a key concept in the RLE (Section 3.2 Ecosystem collapse) and underpins the application of the *IUCN Red List of Ecosystems Categories and Criteria*. Assessors should describe the collapsed state(s) of an ecosystem, based on the information summarised in the description of the ecosystem type and the conceptual model. If multiple states of collapse are possible (e.g. due to different threats), all of these should be described with similar levels of detail. Descriptions should focus on the key defining features of the ecosystem type. Collapse thresholds for the application of criteria A and B are typically defined as 100% loss of spatial distribution of the ecosystem type (i.e. 100% decline under criterion A; EOO = 0 km^2 and/or AOO = no 10 x 10 km grid cells occupied under criterion B). Use of alternative thresholds of collapse for criterion A or B must be thoroughly justified. Collapse thresholds for the application of criteria C, D and E should be identified as part of the assessment of those criteria (Section 7.3 Applying criteria C and D). Assessors are encouraged to provide examples of locally collapsed occurrences of the ecosystem type to support their descriptions of collapsed states (Box 8).

Box 8. Defining ecosystem collapse.

The **Mountain Ash Forest of south-eastern Australia** is a unique ecosystem dominated by the world's tallest flowering plant species (*Eucalyptus regnans*). Mountain ash supports a wide range of plant species and a rich array of native mammals and birds, including the Endangered Leadbeater's possum and the Vulnerable yellow-bellied glider (Lindenmayer, 2009). The availability of old-growth (primary) forest and natural tree hollows is a critical factor in the survival of cavity-dwelling animals, which are absent or rare when the density of hollow-bearing trees falls below one per hectare (Keith et al., 2013; Burns et al., 2015).

Ecosystem collapse is considered to occur under any of the following (Burns et al., 2015):

- 100% of the area where the ecosystem currently occurs is no longer bioclimatically suitable (criterion C).
- The abundance of hollow-bearing trees drops below one per hectare averaged across the entire ecosystem distribution (subcriterion D2 and criterion E).
- Less than 1% of old-growth forest remains in the ecosystem, i.e. < 1% of the forest is in a primary state (subcriteria D1 and D3).

4.3. Evaluating the criteria

A key principle of Red List assessments is that each ecosystem type must be assessed against all of the RLE criteria so far as the available data permit. A similar principle applies to The IUCN Red List of Threatened Species criteria (IUCN SPC, 2024). Sections 5 to 8 provide detailed information on how to gather data, perform an assessment, consider data quality and uncertainty, and document an assessment outcome for each of the criteria – Section 5 Criterion A. Reduction in geographic distribution, Section 6 Criterion B. Restricted geographic distribution, Section 7 Criterion C and D. Environmental degradation and disruption of biotic processes and Section 8 Criterion E. Quantitative risk analysis.

At the outset of an assessment, all ecosystem types are considered Not Evaluated (NE) for all criteria (Figure 9). The next step is to determine whether adequate data exist for application of each of the criteria, which requires data searches of the scientific literature, data repositories, unpublished reports, expert opinion (see Section 3.3.4 Quantitative data and expert knowledge), historical accounts, past and present maps, satellite imagery or any other source of relevant data. If no adequate data exist to assess any of the criteria, the overall assessment outcome is Data Deficient (DD; Figure 9).

Following this initial assessment of data, assessors must systematically evaluate each of the RLE criteria. If an assessor is unable to apply a criterion (noting the key principle above), the risk assessment outcome for this criterion is Not Evaluated. If a reasonable search effort indicates that adequate data are not available to assess under a criterion, the risk assessment outcome for this criterion is Data Deficient (DD). The difference between Not Evaluated and Data Deficient is important for reporting purposes. The search effort for appropriate data should be briefly described in documentation.

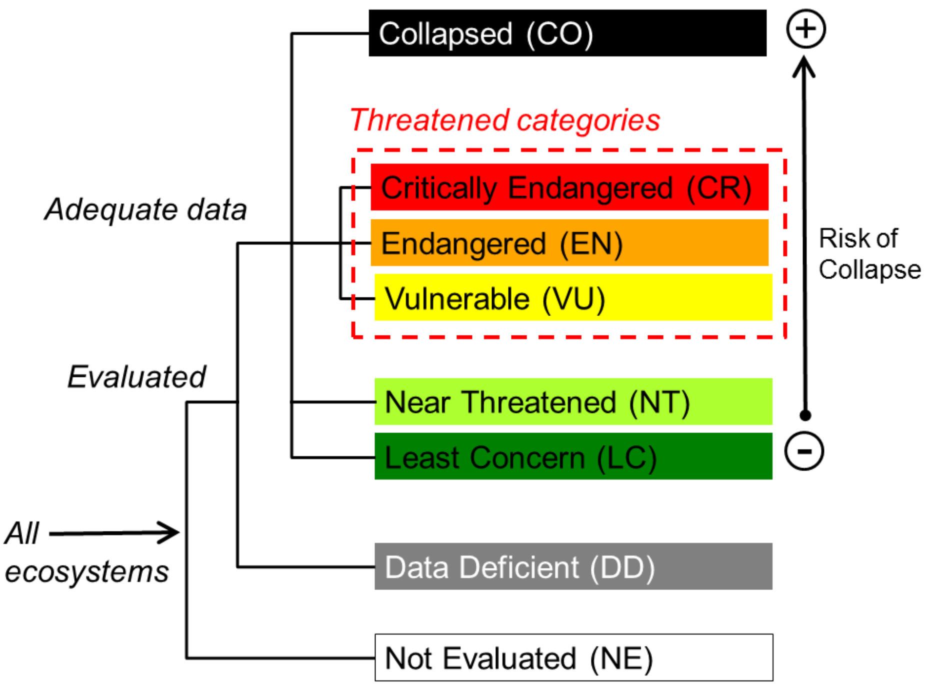


Figure 9. Process of evaluating the IUCN Red List of Ecosystems criteria. Source: Authors. Ecosystem types that are yet to be evaluated are assigned to Not Evaluated (NE). The first step in evaluation is to determine whether data are sufficient to support an assessment of any criteria. Ecosystem types are assigned to Data Deficient (DD) when such data are lacking. Ecosystem types with sufficient data are assigned to categories from Least Concern (LC) to Critically Endangered (CR) in increasing risk of collapse (- to +), unless they have already Collapsed (CO). The three threatened categories (dashed line) are determined by assessing the quantitative criteria.

Although the overall outcomes of the assessment are more certain if multiple criteria are assessed, it is possible to estimate the status of an ecosystem type based on a single subcriterion if all other criteria and subcriteria are Data Deficient. However, the fewer the number of criteria and subcriteria assessed, the greater the risk of underestimating the overall threat category because the most severe symptoms of decline or degradation may not have been assessed. The following procedure may be applied to reduce the occurrence of Type II errors in which ecosystem types at risk of collapse are erroneously assessed as Least Concern where data are limited.

If there are sufficient data to assess only one subcriterion of one criterion (e.g. B1) and the outcome of that assessment is Least Concern (i.e. all other subcriteria are Data Deficient), then assessors have discretion to assign either Least Concern or Data Deficient as the overall status of the ecosystem type. In this case, Data Deficient may only be assigned as the overall status if:

1. The only subcriterion that was assessed does not address the major threat(s) to the ecosystem type based on all available evidence (e.g. as summarised in the conceptual model, see Section 4.2.5 Conceptual models).
2. It is likely that the ecosystem type could be eligible for a threatened category (CR, EN or VU) if sufficient data were available to assess another subcriterion.
3. The reasoning for inferences drawn on points 1 and 2 are justified in the assessment documentation.

For example, if a forest ecosystem was assessed as Least Concern (LC) under subcriterion A1 because of a negligible rate of reduction in its geographic distribution over the past 50 years, degradation by recurring fires is the major threat identified in its conceptual model, and it could plausibly be assessed as Vulnerable (VU) if data were available for fire-related indicators under criterion C, then the overall status may be assigned as Data Deficient (overriding the Least Concern outcome from subcriterion A1).

If at least two of the 20 subcriteria are assessed (i.e. sufficient data are available) and all produce Least Concern outcomes, then the overall status of the ecosystem type must be Least Concern (i.e. there is no discretion to assign the overall status as Data Deficient).

4.4. Assessment outcome

A summary table for each ecosystem type reports the assessment outcome for all criteria (and subcriteria) as well as the overall status (Box 9). There are a total of 20 subcriteria within the five criteria in the *IUCN Red List of Ecosystems Categories and Criteria*, each of which can be assigned one of the eight risk categories (Figure 1). The results for all subcriteria under criteria A, B, C and D, and the methods used to assess the subcriteria (i, ii or iii), must be reported during the assessment process.

If some ecosystem types are Data Deficient or Not Evaluated for some of the subcriteria; this must be included in the summary table (Box 9). If all subcriteria are Data Deficient, the overall outcome of the assessment is Data Deficient. If all subcriteria are Not Evaluated, the overall outcome of the assessment is Not Evaluated. If all subcriteria are either Not Evaluated or Data Deficient, the overall outcome of the assessment is Data Deficient. If sufficient data exist for only one subcriterion, and its assessment produces a Least Concern outcome, assessors may apply the procedure outlined in Section 4.3 Evaluating the criteria.

Following the precautionary principle and to ensure that the most severe symptoms of risk determine the assessment outcome, the highest risk category obtained from any of the assessed criteria will be the overall risk status of the ecosystem. The main method currently used for representing uncertainty in ecosystem assessment is to use bounded estimates (Section 4.4.1 Dealing with uncertainty).

The following rules should be applied to determine bounded estimates of the overall status where multiple criteria have been assessed:

1. Determine the overall status by identifying the highest best estimate of the risk category returned across all five criteria.
2. The lower bound of the overall status is the highest lower bound across any of the subcriteria that return the same category as the overall status.
3. The upper bound of the overall status is the highest upper bound across any of the subcriteria that return the same category as the overall status.

Box 9 provides examples for interpreting these rules.

Box 9. Assessment outcome. Source: Adapted from Keith et al., 2013.

Caribbean Coral Reefs. Source: Adapted from Appendix S2.17 in Keith et al., 2013.

Caribbean coral reefs are primarily fringing reefs and bank barrier reefs separated from island and mainland shorelines by reef flats, shallow waters or slightly deeper lagoons (Alevizon, 2010). Due to the difficulties of remotely measuring the distribution of live coral and mosaic marine ecosystems, the ecosystem is listed as DD under all subcriteria of criterion A. The ecosystem is assessed as LC under all subcriteria of criterion B due to its large extent of occurrence, area of occupancy and number of threat-defined locations. The data for criterion C are currently under review. At the time of assessment, the ecosystem is assessed as NE under all subcriteria of criterion C. Data on coral cover and reef rugosity lead to similar estimates for subcriterion D1: EN (plausible range VU–CR). No projections are available for future disruptions to biotic interactions, so the ecosystem is listed as DD under D2. The ecosystem type is listed as EN under subcriterion D3 based on historical data. No quantitative analysis has been carried out to assess criterion E, so the status is NE under criterion E.

Applying rule 1 from Section 4.4 Assessment outcome, the best estimate of overall risk is EN based on D1 and D3. The highest lower bound across those subcriteria (rule 2) is EN based on D3, which is therefore the lower bound of the overall status. Note that the lower bound based on D1 is VU, which cannot be the lower bound of the overall status because the overall status must be at least EN based on the lower bound for D3. Finally, the upper bound for the overall status is CR (applying rules 1-3), based on D1, which returns the highest upper bound for the two subcriteria that return the highest best estimate. The overall risk status of Caribbean coral reefs is therefore reported as EN (plausible range EN–CR).

Criterion	*A*	*B*	*C*	*D*	*E*	*Overall*
Subcriterion 1	**DD** A1	**LC** B1a,b,c	**NE** C1	**EN** (VU-CR) D1	**NE** E	**EN** (EN-CR) D1,D3*
Subcriterion 2	**DD** A2a,b	**LC** B2a,b,c	**NE** C2a,b	**DD** D2a,b		
Subcriterion 3	**DD** A3	**LC** B3	**NE** C3	**EN** (EN-EN) D3		

** Overall status should specify best estimate, plausible lower and upper bounds and all criteria and full subcriteria that support the overall status (other examples: VU (VU–CR) B1ai, iii, B3, D2a; CR(CR–CR) A2a, B2bii, C1b)*

Coastal Sandstone Upland Swamps of south-eastern Australia. Source: Adapted from Appendix S2.1 in Keith et al., 2013.

The Coastal Sandstone Upland Swamps of south-eastern Australia are treeless bogs that form relatively abrupt boundaries with surrounding eucalypt-dominated forests and woodlands that occupy more freely draining soils (Keith & Myerscough, 1993). They are strongly associated with high rainfall and moisture. Interactions between hydrological processes and fire regimes are crucial to the development of upland swamps and maintenance of their diverse and characteristic biota. To assess potential future decline due to climate change, Keith et al. (2013) used a range of plausible bioclimatic distribution models to predict the distribution of the ecosystem type under future climate scenarios. Based on these models and scenarios, the distribution of the ecosystem was projected to decline by 58–90% (median 74%) over the next 50 years. The status of the ecosystem under subcriterion A2 was therefore determined to be EN (plausible range EN–CR).

The same distribution models used to assess future change in distribution were also used to assess trends in climatic suitability under criterion C. From 1983 to 2009, the summed abundance of woody resprouters declined by a mean of 37% at 72% of sampled sites. These are just below the severity and extent thresholds, respectively, for VU under criterion D1, assuming that zero abundance of resprouters marks the point of ecosystem collapse. No data were available prior to 1983, but if current declines were initiated prior to that time, they may exceed the threshold for VU status. The status of the ecosystem type is therefore likely to be NT (plausible range NT–VU) under subcriterion D1.

Applying rule 1 from Section 4.4 Assessment outcome, the best estimate of overall status of the ecosystem is EN based on subcriteria A2a, B1b,c, B2b,c and C2a. The overall lower bound is EN based on the same four subcriteria (rule 2). The overall upper bound is CR based on subcriteria A2a and C2a (rule 3). Therefore, the overall risk status of the Coastal Sandstone Upland Swamps of South-Eastern Australia is EN (plausible range EN–CR).

Criterion	*A*	*B*	*C*	*D*	*E*	*Overall*
Subcriterion 1	**LC** A1	**EN** B1b,c	LC C1	**NT** (NT-VU) D1	**DD** E	**EN** (EN-CR) A2a,B1, B2 C2a
Subcriterion 2	**EN**(EN-CR) A2a	**EN** B2b,c	**EN**(EN-CR)C2a	**DD** D2a,b		
Subcriterion 3	LC A3	**LC** B3	DD C3	DD D3		

4.4.1. Dealing with uncertainty

Uncertainty in any information used to evaluate the criteria should be propagated through the assessment and reported as part of the outcome. Reporting both the most likely risk category and other plausible categories, given the uncertainties in the data, is more useful than simply reporting the most likely category. The simplest means of characterising uncertainty is through bounded estimates. Bounded estimates represent a range of plausible alternative values for a measure. They can take into account uncertainty in thresholds describing collapsed states (Figure 4 and Box 2), mapped estimates of change in distribution (Box 10), and estimates of variables for measuring relative severity in criteria C and D (Section 7.3 Applying criteria C and D).

The upper and lower bounds of an estimate may be propagated through an assessment by repeating the same analysis for the best estimate, and the lower and upper bounds. For example, if the decline in the geographic distribution of an ecosystem type is estimated to be between 75–85% in the last 50 years, it could plausibly be either Endangered (decline between 50–80% based on the lower bound) or Critically Endangered (≥80% based on the best estimate and upper bound) under subcriterion A1. In general, the best estimate should be based on weight of evidence. For example, a bounded range of 70–82% decline suggests Endangered (EN) status is more likely than Critically Endangered (CR), though both are plausible and should be reported as the bounded status as follows: EN (EN–CR). Dealing with uncertainty in ecosystem risk assessment draws largely on the experiences of The IUCN Red List of Threatened Species (Newton, 2010; Regan & Colyvan, 2000; Akcakaya et al., 2000).

4.5. Documentation

All assessments must be accompanied by documentation and supporting information, which should undergo peer review by appropriate experts (Section 10 Databasing, peer review and publication), and must be readily available when the assessment is completed (see the RLE website for examples: www.iucnrle.org). All required fields in the online RLE database should also be completed (see the RLE website). The documentation must include the following sections:

1. **Summary.** A brief abstract (~200 words) that describes the complete assessment in summarised form, including the area of assessment, the focal ecosystem type and its defining features, threatening processes and the assessment outcome.
2. **Ecosystem description.** A complete description of the ecosystem type, including the elements listed in Table 6.
3. **Risk assessment.** Specific information on the application and outcome of each criterion e.g. inferences, statistical analyses and spatial analyses. The section should also include a discussion of assumptions, limitations or further data required. Further guidance is available in Section 5 Criterion A. Reduction in geographic distribution, Section 6 Criterion B. Restricted geographic distribution, Section Criterion C and D. Environmental degradation and disruption of biotic processes and Section 8 Criterion E. Quantitative risk analysis.
4. **References.** A complete reference list showing the sources of information used for the assessment must be provided.

4.5.1. Documenting change in threat status through successive Red List assessments

The RLE is used to monitor the status of ecosystems over time (Bland et al., 2019; Nicholson et al., 2024). Several countries have already established a time series of Red List assessments, for example, South Africa (Botts et al., 2020), Finland (Kontula & Raunio, 2019) and Norway (NBIC, 2018). These temporal applications and repeat assessments are expected to become more widespread in the near future as countries report progress on global ecosystem conservation and restoration targets using the RLE as the agreed Headline Indicator for ecosystems under the Kunming-Montreal Global Biodiversity Framework (Nicholson et al., 2024).

To interpret progress on these targets, it is essential to distinguish genuine changes in Red List status from non-genuine changes and to identify the reasons for change. Genuine changes in status are due to change in the risk of ecosystem collapse in response to intensification or abatement of threatening processes, or in response to ecosystem management or restoration action. Non-genuine changes in status (analogous to Type I errors) may result from a change in knowledge or methods (e.g. improved data, improved analysis). Conversely, changes in knowledge or methods between successive Red List assessments may also mask genuine changes in status for some ecosystem types, i.e. when the Red List status does not change due to a change in data type or methods between assessment, even though substantial changes in risk have occurred (analogous to Type II errors, Taylor & Gerodette, 1993). In these latter cases, successive assessments would show a change in Red List status if the latest data types and methods had been applied consistently across all assessments. Overall, the trends in status of a given assessment unit over successive Red List assessments may result from a combination of both genuine and non-genuine factors.

To ensure transparent and accurate reporting, genuine changes and genuine non-changes in Red List status must be distinguished and the reasons for non-genuine change or potential masking effects should be documented. This enables:

1. Calibration of successive assessments by hindcasting current methods and data types to previous assessments, wherever possible, to ensure that changes and non-changes in Red List status are genuine.
2. Exclusion of non-genuine assessments from interpretation and reporting of ecosystem trends where calibration is not possible.

Documentation of Red List time series should refer to standard reasons for interpreting change and non-change in status (Table 9). These categories are based on analogous categories used to estimate the Red List Index for Threatened Species (Butchart et al., 2004, 2007).

Table 9. Reasons for change or non-change in Red List status between successive assessments.

Reason for change/non-change in Red List status	*Explanation*
1. Genuine	A change in the underlying risk of ecosystem collapse due either to changing pressures from threatening processes, or conservation action through ecosystem protection, management or restoration. Genuine changes may be attributed when none of the other categories apply or where there is evidence that the change/non-change in status is genuine, despite other factors (below) affecting assessments.
2. Increased knowledge	Change in status due to improvements in data quality or quantity for the same indicators and analyses applied in the preceding assessment.
3. Change in method	Change or non-change in status due to use of different or additional indicators in the assessment of listing criteria, or in methods of time series or spatial analysis.
4. New ecosystem type	Change in status due to an ecosystem type assessed in a later assessment that was either Not Evaluated (NE) or Data Deficient (DD) in the preceding assessment (cf. 5).
5. Change in ecosystem classification	Change or non-change in status due to a change in circumscription of an ecosystem type or splitting or lumping of two or more assessment units in a previous assessment (cf. 4).
6. Error in previous assessment	Change or non-change in status due to an error discovered in a previous assessment and corrected in a subsequent assessment, e.g. misinterpretation of Guidelines, use of incorrect data.
7. Change in RLE version	Change or non-change in status may occur when the same ecosystem types are assessed under v2.0 after an earlier assessment based on v1.0 of the RLE criteria.

One or more of the reasons in Table 9 should be attributed to each change in status for each ecosystem type between successive Red List assessments. The reasons should be documented (Table 10).

Table 10: Recommended layout for documenting change and non-change in Red List status through successive assessments.

Ecosystem type	*Status in Assessment 1*	*Status in Assessment 2*	*Change in status*	*Reason*	*Interpretation notes*
Ecosystem type 1	**EN** (EN–CR)	**VU** (VU–VU)	genuine change	1	-
Ecosystem type 2	**LC** (LC–LC)	**LC** (LC–LC)	genuine non-change	1	-
Ecosystem type 3	**VU** (NT–VU)	**CR** (EN–CR)	non-genuine change	5	Assessment unit split into two ecosystem types between successive assessments.
Ecosystem type 4	**VU** (NT–EN)	**VU** (NT–EN)	non-genuine non-change	3; 6	New regression model used to assess trend and erroneous data points removed.
Ecosystem type 5	**EN** (EN–CR)	**EN** (EN–CR)	genuine non-change	1(2)	Genuine non-change despite use of improved map data in Assessment 2.

Information on genuine and non-genuine change is essential for estimating the Red List of Ecosystems Index and related indicators (Rowland et al., 2020). Only genuine changes in Red List status (category 1, Table 9) and genuine non-changes in status are included in the calculation of the Red List of Ecosystems Index. Although non-genuine changes and non-changes should be excluded, every effort should be made to minimise these by calibrating previous assessments to the current one. With the aid of efficient workflows, and depending on the reasons for non-genuine changes or non-changes (Table 9), it should be possible to correct errors and hindcast newly adopted methods and data types to maximise compatibility of assessments within the time series.

Calibration of a Red List time series should address the following issues:

1. Error correction (reason 6, Table 9):
    a. Error in workflow or computation (e.g. incorrect implementation of criteria).
    b. Error in the input data (e.g. miscoded data, labelling issues).
2. Workflow changes (reasons 3 & 7, Table 9):
    a. Version change in RLE standard criteria and categories.
    b. Change in methods for applying a criterion (e.g. different regression model for estimating trends).
3. Ecosystem classification changes (reasons 4 & 5, Table 9):
    a. Altered circumscriptions, spitting or lumping of units of assessment.
4. Data changes (reason 2, Table 9):
    a. Changes to ecosystem mapping (i.e. adjustment of spatial boundaries).
    b. Changes to the available pressures data (e.g. improved resolution land cover change, new data on invasive species).
    c. New data on pressures, threats or functional decline acquired enabling assessment of criteria that were previously Data Deficient.

The following recommendations should be considered when undertaking repeat Red List assessments and calculating the Red List of Ecosystems Index:

1. Undertake calibration wherever possible to ensure that successive Red Lists report genuine changes.
2. Apply reproducible methods and workflows whenever possible to enable methodological differences to be resolved retrospectively (e.g. by applying improved methods to old data).
3. Review previous assessments prior to each new assessment, identify inconsistencies with methods and data types planned in the forthcoming assessment, and integrate calibration into the workflow.

5. Criterion A. Reduction in geographic distribution

5.1. Theory

A decline in geographic distribution – defined as all spatial occurrences of an ecosystem type – influences its risk of collapse by: (i) reducing the ability of an ecosystem to sustain its characteristic native biota; and (ii) predisposing it to additional threats (Keith et al., 2013). The loss of characteristic native biota due to a declining distribution typically occurs through a combination of reduced carrying capacity, niche diversity, spatial partitioning of resources, and increased susceptibility to competition, predation and threats (MacArthur & Wilson, 1967; Shi et al., 2010; Harpole & Tilman, 2007; Hanski, 1998; McKnight et al., 2007). The rate of decline in an ecosystem distribution indicates its trajectory towards collapse, with ecosystem collapse typically occurring when no spatial occurrences of the ecosystem type remain (extent of distribution collapses to zero).

5.2. Thresholds and subcriteria

An ecosystem may be listed under criterion A if it meets the thresholds for any of four subcriteria (A1, A2a, A2b or A3), quantified as a reduction in geographic distribution over the following time frames:

Subcriterion	*Time frame*	*CR*	*EN*	*VU*
A1	Past (over the past 50 years)	≥ 80%	≥ 50%	≥ 30%
A2a	Future (over the next 50 years)	≥ 80%	≥ 50%	≥ 30%
A2b	Any 50-year period (including past, present and future)	≥ 80%	≥ 50%	≥ 30%
A3	Historical (since approximately 1750)	≥ 90%	≥ 70%	≥ 50%

5.3. Applying criterion A

5.3.1. Data requirements

The rate of decline in distribution is typically estimated from time-series data appropriate for the focal ecosystem type. Ecosystem maps – such as those derived from remote sensing classifications, distribution models, field observations or historical data – are a principal data source for assessing criterion A. Remote sensing is a commonly applied approach that contributes to mapping distributions of many terrestrial and marine ecosystems that have interpretable signatures from different remote sensing platforms (e.g. Figure 10). In most cases, it is necessary to apply remote sensing in combination with on-ground observations and environmental data to achieve the most accurate mapping outcomes. Where regional or local datasets are lacking, global datasets, such as those available for forests (Hansen et al., 2013), mangroves (Giri et al., 2011), surface water cover (Pekel et al., 2016) and tidal mudflats (Murray et al., 2019), may be suitable templates for superimposing appropriate classifications of ecosystem types.

When more than one source of data is available, such as different vegetation maps or estimates produced with different methods, assessors should first critically evaluate the efficacy of the alternatives as representations of the distribution of the ecosystem type. Decisions about data sources should be based on a comparison of map properties including (i) map accuracy and ground-truthing effort, (ii) methodological rigour, (iii) time since map development, (iv) spatial resolution, and (v) the availability of a consistent time-series. In some cases, it may be appropriate to develop a 'consensus' or composite map, or use multiple distribution maps to represent uncertainty in the extent of the ecosystem. If more than one data source is suitable, assessors can calculate estimates of area from each data source, and explore the sensitivity of ecosystem status to this data uncertainty (Section 4.4.1 Dealing with uncertainty). The net reduction in geographic distribution will then form an interval of estimates generated from each data source.

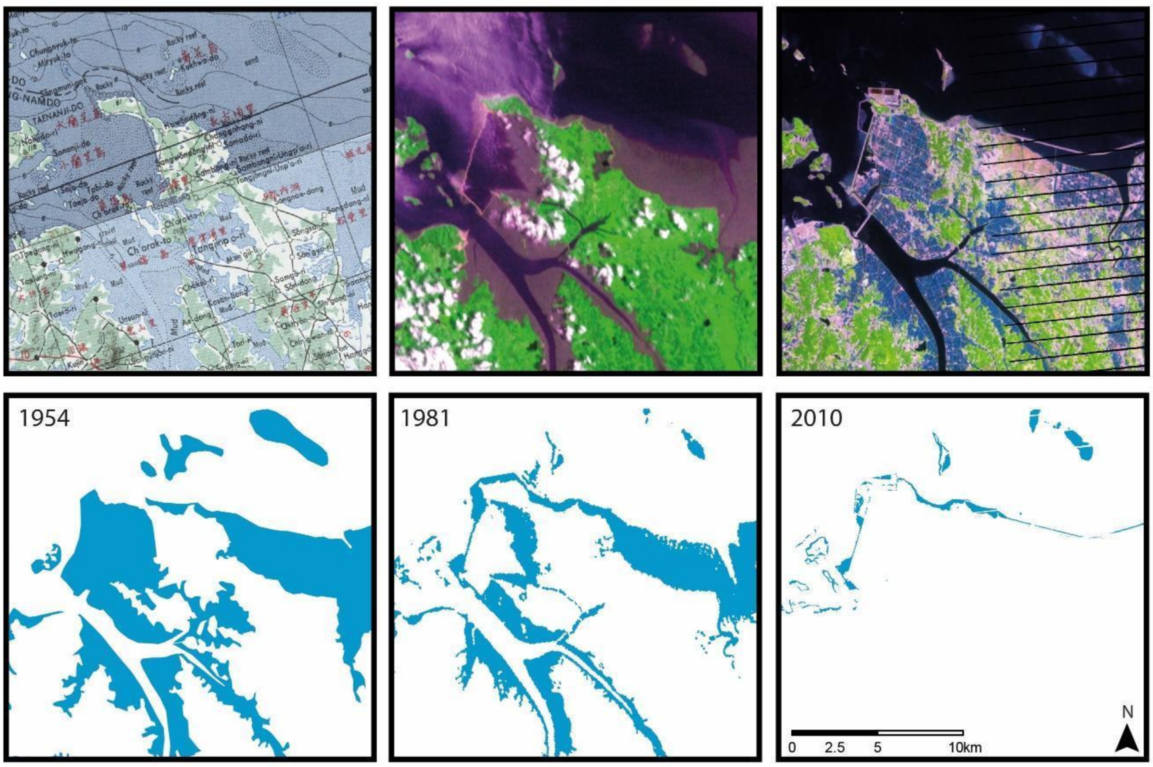


Figure 10. Time series maps of an ecosystem distribution inform the risk of ecosystem collapse. Source: Murray et al., 2012, 2014, 2015, 2019.
Here, historical topographic maps (1954) and Landsat Archive satellite imagery (1981, 2010) allowed a standardised time series of the area of the Yellow Sea tidal flat ecosystem to be developed for assessment under criterion A.

Where justified, spatial proxies for ecosystem distributions may be used, such as field observations of organism assemblages, keystone species, climate, substrate, topography, bathymetry, ocean currents, flood regimes, water cover, aquifers or some synthesis of these that can be justified as valid representations of the distribution of ecosystem biota or its niche space. For example, maps of physical factors such as sea floor characteristics, ocean currents, water temperatures and water chemistry may be appropriate for marine ecosystems. In some subterranean, freshwater and marine ecosystems, trends in the depth dimension may be appropriate proxies of declines in distribution, so long as they reflect trends in carrying capacity and niche diversity for characteristic biota (Keith et al., 2013).

Spatial distribution models offer an additional opportunity to formally select and combine the most suitable set of spatial proxies to predict ecosystem distributions. For example, Clark et al. (2015) used bathymetric spatial data and remote sensing data on sea ice concentration to model the distribution of suitable light conditions for under-ice marine benthic invertebrate communities in Antarctica. Models are especially useful for projecting time series of ecosystem distributions into the future for assessing criterion A2. Keith et al. (2014) modelled the distribution of a mire ecosystem under future climate scenarios using a map of present-day mires developed from satellite imagery, in combination with hydrologically-based climate, substrate and terrain predictor variables. In both studies, a mechanistic understanding of the relationship between occurrence of the ecosystem and limiting environmental factors was central to developing an adequate ecosystem map.

Climate change may lead to a change in the geographic distribution of an ecosystem type via changes in environmental conditions that directly affect ecosystem occurrence. Subcriterion A2 can be used to assess projected declines in the distribution, however, projected changes in actual distribution should be distinguished from projected changes in climatic suitability, which are assessed under criterion C (Section 9.1 Climate change). The rate of distribution change may be affected by several factors, such as the trajectory (speed, direction, magnitude) of climate change across the distribution, the ability of the characteristic native species to adapt to changing environmental conditions and/or track suitable conditions (e.g. Bairos-Novak et al., 2021), the availability of suitable environmental conditions, and species interactions such as competition and dependencies. For example, the distribution of mangroves in Moreton Bay has shifted landward due to sea level rise, encroaching and causing distribution declines in adjacent saltmarsh (Sievers et al., 2020).

It is beyond the scope of these Guidelines to provide detailed information on the acquisition, classification and accuracy assessment of spatial data. Nevertheless, it is assumed that spatial data used for assessments under criterion A are fit for purpose in being (i) consistent and comparable across time periods (unbiased), (ii) sufficiently accurate (Congalton & Green, 2008), and (iii) of a suitable grain size/scale for the ecosystem type being assessed (Murray et al., 2017).

5.3.2. Methods

To apply criterion A, at least two comparable estimates of the geographic distribution of the ecosystem type at different points in time are required. Although assessments can be completed with just two data points, efforts should be made to ensure appropriate power in a suitable statistical model of ecosystem change and that all model assumptions are addressed in the analysis.

In general, where data are available, estimates based on a statistical model fitted to a time series of estimated extent are preferred to a simple comparison of estimates at the beginning and end of an assessment time frame (see Section 5.3.1 Data requirements). This enables the shape of the trend to be taken into account and to distinguish overall trends from fluctuations. It also enables projection and hindcasting, subject to appropriate assumptions. The choice of a suitable statistical model must be justified and documented with appropriate ecological and statistical reasons. Where justified, and with appropriate treatment of uncertainty, assessors may interpolate or extrapolate estimates of change over the assessment time frame based on a statistical model fitted to available data. If data are available beyond the assessment time frame as well as within it, assessors should consider the benefits and limitations of fitting the model to a longer time series in order to calculate change within the assessment time frame. Good practices in data processing and analysis (Olofsson et al., 2013, 2014; Fuller et al., 2003), including model evaluation, should be employed to minimise bias in estimates of areal change over a time-series of spatial data. Guidelines for assessing change in species populations under criterion A of The Red List of Threatened Species provides further detail on methods relevant here (IUCN SPC, 2024).

Subcriterion A1 may be directly assessed if data are available for 50 years ago and the present. However, it is rare for the raw data to be available for precisely the time frames required by an assessment of criterion A. More typically, assessors must use methods of interpolation, extrapolation, or prediction to calculate estimates of distribution change over the last 50 years (A1), the next 50 years (A2), and/or since 1750 (A3). This will involve assumptions about the nature or pattern of change, as well as the quality of the data (Alaniz et al., 2016), which must be explained and justified in the documentation.

To assist calculations, a spreadsheet tool is available on the IUCN Red List of Ecosystems website (www.iucnrle.org). Several tools for assisting in this step are in development and will become available on the website in the future.

Future change

Subcriterion A2 requires projection of changes in ecosystem distribution over the next 50 years or any 50-year period including the present and future (Section 3.3.1 Time frames). For ecosystems likely to be affected by climate change, it may be possible to estimate future declines in distribution (subcriterion A2) using modelling approaches based on multiple environmental variables (described in Section 9.1 Climate change). Box 10 provides an example model projection applied to a cryogenic ecosystem type in Norway. In many other kinds of ecosystems, however, there may be substantial and unpredictable ecological lags as ecosystems adjust to new climatic conditions and model projections are best interpreted as projections of environmental suitability under subcriterion C2 than projections of distribution change under subcriterion A2.

Box 10. Assessing uncertain change in distribution due to climate change.
Source: Adapted from Aarrestad et al., 2018.

Snow bed in Norway is characterised by vegetation above or near the treeline and late-lying snow that limits the growing season and is mainly found in depressions in the landscape. Increasing average annual temperatures could have three different effects in these mountain ecosystems. First, an upwards shift in the treeline altitude could promote forest growth and reduction of area for snow beds (Bakkestuen et al. 2008). Second, increasing snow melt higher up in the mountains creates new areas for colonisation by characteristic specialist snow bed species. At the same time, increasing temperatures also affect the duration of the snow cover and the water content in the soil, and are expected to favour colonisation of snow bed areas by shrubs, graminoids and herbs and change species composition in the long term. These three effects of rising temperature might be weakened or enhanced by additional complex interactions between biotic and abiotic factors such as competition, grazing, wind, and changeable ice and snow cover. Changes in soil-nitrogen content also affect the dynamic balance between snow beds and other ecosystem types (Steinbauer et al., 2018; Austnes et al., 2018).

The predicted changes in the treeline altitude and subsequent reduction in suitable areas, combined with new areas with higher snow melt under climate warming, were used to estimate potential change in distribution of snow bed under subcriterion A2. Although treeline inertia was not quantified, decadal-scale shifts in treeline observed nearby in Sweden (Kullman & Öberg, 2009), suggest that modelled shifts over the next 50 years are likely to represent actual changes in distribution, hence modelled projections were assessed under subcriterion A2, rather than C2.

The current distribution map of the ecosystem type was based on a satellite-based map of vegetation types for Norway (30 x 30 m resolution from Landsat; Johansen, 2009), and the current treeline is based on the lower limit for the subalpine zone (Bakkestuen et al., 2008). Under the RCP4.5 scenario, annual average temperature is predicted to rise by 2.257°C over the next 50 years. Under these conditions, the treeline is predicted to rise by up to 375 m (not accounting for inertia in the treeline), with a lapse rate of 0.6°C per 100 m elevation (Wieser & Tausz, 2007). However, the relationship between rises in treeline and warming climate conditions is inconsistent among studies across Northern Europe. For example, the forest line rose by 25 m in 50 years in the Kola Peninsula, Russia with no change in temperature (Mathisen et al. 2014). The treeline increased by 75 m over 30 years in mountainous areas of Sweden, corresponding to a 1°C rise in temperature (Kullman & Öberg, 2009). How high the forest line will rise in the next 50 years, if the average temperature increases by 2.257°C, is therefore highly uncertain. To estimate the corresponding area loss to increases in the treeline at 50 m intervals from 50–375 m, a GIS analysis was undertaken using the Digital Terrain Model 50 dataset (50 x 50 m cells, 4–6 m accuracy) from the National Mapping Authority (2007). The calculations show that with a treeline elevation rise of 375 m, a maximum of 83% of the current mountain area will experience temperatures high enough for forests to form.

Assuming the proportion of snow bed above the treeline does not change, the reduction in distribution will be 26% if the treeline rises by 50 m, and 40% if it rises by 100 m in elevation. Factoring in potential changes in soil-nitrogen content, the distribution of snow bed is predicted to decline by 30% if the treeline rises by 50 m as a conservative estimate. However, there are major uncertainties related to the current distribution from the vegetation map (Erikstad et al., 2009), the inertia (i.e. lagged ecological response to warming) of the ecosystem type when the vegetation responds to changes in climate, and the possibility of new snow beds forming at higher elevations in areas that were previously inhospitable but will experience more frequent snow-free periods under climate warming. Taking this uncertainty into account, particularly in relation to inertia and expansion of snow bed at the leading edge, it is assumed that the loss of distribution will be just less than 30%, resulting in the ecosystem type being listed as Near Threatened under subcriterion A2a.

Historical change

Subcriterion A3 requires estimation of changes in ecosystem distribution over historical time since onset of the industrial era, notionally in 1750 (see Section 3.3.1 Time frames for interpretation). Changes in ecosystem properties over historical time frames may be estimated with the aid of models based on environmental relationships or inferences that draw evidence from relictual present-day distributions and/or historical information, including that available in historical documents, surveys, maps, oral histories and expert elicitation (e.g. Mladenoff et al., 2002; Bickford & Mackey, 2004). Qualitative or quantitative models that link the occurrence of an ecosystem type to environmental conditions can be used to predict areas that were suitable for occurrence of the ecosystem type just prior to industrialisation (i.e. 1750). If the models are based on present-day observations of occurrence and contemporary environmental conditions, the outputs will show suitability for occurrence of the ecosystem type at the present day, had it not been transformed by intensive land use. This may provide a useful approximation for retrospective estimation of historical distribution prior to transformation, assuming environmental conditions (as defined by model predictors) were similar to those at the present day.

The recommended protocol involves the following steps:

1. Develop a spatial model of environmental suitability using appropriate environmental predictors (ideally variable selection should be informed by a conceptual model and relate to characteristic ecosystem properties and processes, rather than correlative occurrence relationships alone).
2. Critique and revise the model with available information on contemporary distribution (e.g. formal cross validations, expert knowledge) and on historical distribution (e.g. historical journal accounts and sketch maps, specimen collections of characteristic biota, place-based photographic images, drawings, occurrences of relictual biota such

as trees in transformed landscapes, etc.). The modelled distribution should exclude areas occupied by other ecosystem types.

3. Use the validated model or qualitative inference to project the distribution of environmentally suitable conditions for occurrence of the ecosystem type throughout its potential range (i.e. including areas currently occupied by intensive land use).
4. If intensive land use commenced after the onset of industrialisation (c. 1750) and environmental conditions have not changed appreciably since then, the retrospective projection may be assumed to represent the distribution at the onset of industrialisation.
5. If intensive land use commenced after the onset of industrialisation (c. 1750) and environmental conditions have changed appreciably since then, the model must be hindcast using the appropriate environmental settings that approximate prevailing conditions at the onset of industrialisation. The model predictors will define how conditions are described (e.g. mean temperatures, seasonal rainfall, etc.) and estimates should be informed by available data, projections and other information on conditions as at c. 1750.
6. If intensive land use commenced before the onset of industrialisation (c. 1750), the estimated baseline extent (at c. 1750) should be adjusted to exclude the proportion of depletion estimated to have occurred before c. 1750. This proportion may be estimated from a validated temporal model of intensive land-use change, available historical information or using expert opinion with appropriate structured elicitation methods (see Section 3.3.4 Quantitative data and expert knowledge). If environmental conditions may have changed appreciably since then, the model must be hindcast using the appropriate environmental settings that approximate prevailing conditions at c. 1750.

Modelled or interpolated historic reference states (see Glossary) permit declines to be estimated based on the difference between the current state of an ecosystem and its expected state in the absence of industrial-scale anthropogenic effects. Steps 4–6 describe how models and inferences need to be applied differently, depending on the history of intensive land use or resource use and whether it commenced before or after the onset of industrialisation (c. 1750). Similar approaches may be applied to hindcast ecosystem indicators for assessment of subcriteria C3 and D3.

In most parts of the world, industrial-scale exploitation of ecosystems commenced after 1750 (i.e. almost all of the Americas and Oceania, large portions of other terrestrial regions, as well as the oceans), and step 4 will be appropriate for most terrestrial ecosystem types characterised by slow turnover and long-lived functional or structural dominants, such as forests, deserts, etc. More labile ecosystem types with faster turnover rates (e.g. some wetland or coastal ecosystem types such as mangroves) may be more responsive to environmental change and may require hindcasting described in step 5.

Regions with a long history of intensive land use that extends well before the industrial era include eastern China, parts of southeast Asia, south Asia, much of Europe, long-settled river valleys such as the lower Nile and Tigris-Euphrates and other localised areas in Africa and the Americas. In most of these regions, intensive land use continues, albeit with changing character. Approaches described in step 6 will be appropriate in these cases.

In some localised areas, intensive land use declined and further ecosystem transformations occurred after pre-industrial land use intensification. For example, on the lowlands of the Yucatan Peninsula (Mesoamerica), a new forest developed after the ninth century where an earlier one had been replaced by crop fields and settlement (Turner & Sabloff, 2012). Changes to ecosystems that occurred over multiple centuries or millennial time scales are less relevant to contemporary ecosystem management and therefore should be excluded from Red List assessments. For example, the relevant ecosystem type for contemporary Red List assessment on the Yucatan lowlands is the forest that developed after the ninth century and occupied the area at the onset of industrialisation. The degree to which this forest ecosystem resembles the earlier one displaced by Mayan settlement is not directly relevant to contemporary Red List assessment.

It is very challenging to estimate ecosystem extent and state variables at c. 1750 in regions where intensive land use is ongoing, and limited evidence is available on prior depletion and degradation of ecosystems at the onset of the industrial era. Every effort should be made to make (at least coarse) bounded estimates of ecosystem extent at this time based on historical sources such as surveys, narratives, journals, transformational events and sketch maps etc. (e.g. Mladenoff et al., 2002; Bickford & Mackey, 2004). Where insufficient or inaccessible information preclude estimates of a reference state (Glossary), the Red List subcriteria that address declines over historical time frames must be assessed as Data Deficient.

5.3.3. Assumptions

Whether inferences are made from time series of satellite images or from other data sources, two important aspects will fundamentally influence assessments: (i) assumptions about the rate of decline; and (ii) the number of points in the time series. When the rate of decline is estimated from two observations (e.g. maps) over a specified time frame, assessors should use information about the causes and context of the decline to deduce the likely trajectory of decline (Figure 11).

Although criterion A can be applied acceptably with only two data points, more data enables a more certain diagnosis of the shape of the trajectory, allows the fitting of alternative models, and hence will result in more accurate interpolation, extrapolation or prediction to the full time frames required by criterion A. Selection of candidate models should always be informed by the causes and context of the decline and assessors should ensure that the assumptions of the model are adequately met. Where the drivers of ecosystem change undergo a fundamental change during the assessment period (e.g. cessation of a threat), segmented regressions may offer an appropriate method for estimating the magnitude of change over the full assessment period. At least two plausible alternative scenarios should be explored and all sources of uncertainty in spatial data and decisions about assessment time frames should be clearly described and justified (e.g. Alaniz et al., 2016).

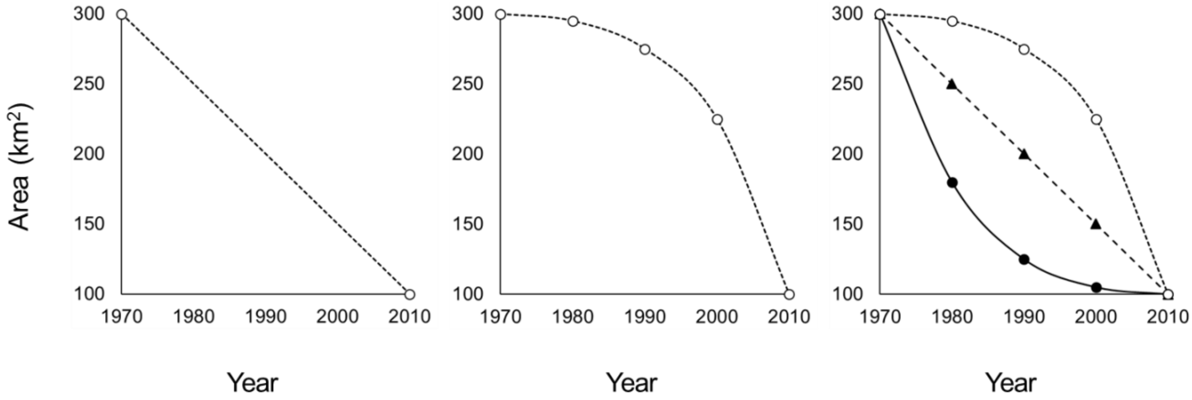


Figure 11. Different potential trajectories of decline for an ecosystem. Source: Authors. All trajectories in this figure have the same endpoints: 300 km^2 in 1970 and 100 km^2 in 2010. A simple interpolation between the two extremes assumes linear decline (left panel). Addition of estimates for intermediate times could reveal that the decline is not linear (middle panel). Different ecosystem types could also exhibit contrasting trajectories with identical endpoints. Future projections of distribution based on these trajectories would clearly differ (right panel).

Two common scenarios (Figure 12) may be modelled using an exponential function (proportional rate of decline), in which a constant fraction of the remaining distribution is lost each year, and a linear function (absolute rate of decline) which assumes a constant area is lost each year (Keith et al., 2009):

Proportional rate of decline: $$PRD = 100 \times \left(1 - \left(\frac{Area_{t2}}{Area_{t1}}\right)^{\frac{1}{(Year_{t2}-Year_{t1})}}\right)$$

Absolute rate of decline: $$ARD = \frac{Area_{t2}-Area_{t1}}{Year_{t2}-Year_{t1}}$$

The predicted changes of these alternative models become more different the further they are extrapolated into the future, as illustrated when each is fitted to a time series of spatial data on the extent of Coolibah-Black Box Woodland, an ecosystem on a semi-arid floodplain in eastern Australia (Figure 12; Keith et al., 2009). In the absence of any other information, proportional (PRD) or absolute (ARD) rates of decline may represent plausible optimistic and pessimistic scenarios, respectively, when a time series of observations does not span the full assessment time frame required by the Red List criteria (Box 11). However, a longer time series of observations – together with an understanding of the drivers of change, the regulatory context, regional variability in land suitability, and the extent of protected tenures across the distribution of the ecosystem – can help to select more realistic models (Keith et al., 2009). More realistic models will produce narrower bounds of uncertainty on the estimated change in distribution. For example, ecosystems in the early stages of large-scale exploitation may be more likely to exhibit linear patterns of decline (ARD) than those in an advanced stage of decline, where the area lost over time will eventually reduce to zero with diminishing area (Puyravaud, 2003).

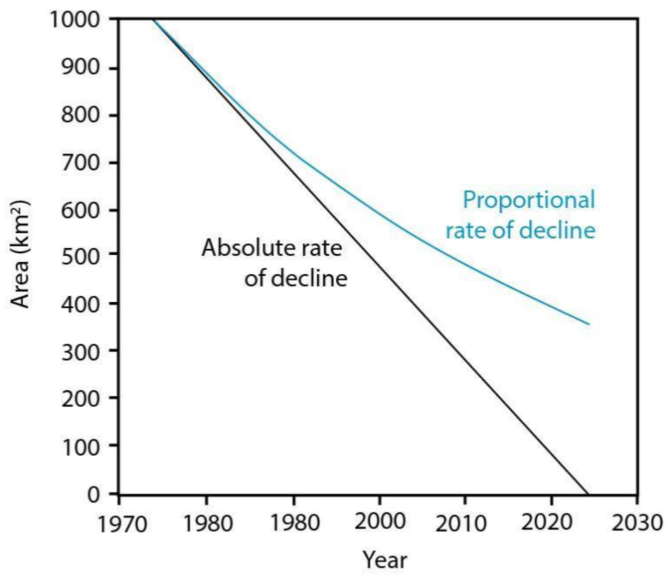


Figure 12. Alternative scenarios for decline in distribution of a model ecosystem.
Source: Keith et al., 2009, 2013.
The figure shows an ecosystem with an initial area (1974) of 1,000 km^2. It declined at a rate of 2% per year during the surveyed years, but the outcome was substantially different if the decline was proportional (PRD) or absolute (ARD). In a PRD, the decline is a fraction of the previous year's remaining area (0.02 × last year's area), whereas in an ARD the area subtracted each year is a constant fraction of the area of the ecosystem at the beginning of the decline (0.02 × 1,000 = 20 km^2/year). Under a PRD scenario, this ecosystem would be considered Endangered under A2b (50% decline over any 50-year period including the present and future), while under an ARD scenario if would have disappeared by 2024, and be assessed as Collapsed.

Box 11. Proportional and absolute rate of decline (criterion A). Source: Portillo, 2014.

Sierra de Perijá is the mountain range that separates north-western Venezuela from north-eastern Colombia. The humid forests in the Venezuelan side of Perijá are threatened by the expansion of large-scale commercial agriculture, primarily of a tuber, the arrowleaf elephant ear (*Xanthosoma saggittifolium*). Using Landsat satellite images, it was estimated that in 1986 the humid forests of the watersheds of the Guasare, Socuy and Cachirí rivers occupied 328 km^2, while in 2001 they had decreased to 198 km^2. These two estimates allow assessment of ecosystem status under subcriterion A2b, using 1986–2001 to first estimate an observed rate of change over 15 years, and then extrapolating projected losses to 2036 (Portillo, 2014).

The forests in 2001 occupied 198 km^2 or 60.4% of their former area in 1986, thus declining at a mean proportional rate of 3.3% per year. The next step is considering how this rate may change over time to project losses at 2036. Assuming a proportional rate of decline (PRD) between 2001 and 2036 results in a total decline of 81.5% between 1986 and 2036. Assuming an absolute rate of decline (ARD) it is predicted to decline by 100% by 2024. Therefore, under criterion A2b PRD leads to a classification of Critically Endangered (≥80% decline over any 50-year period including the present and future), while ARD leads to a classification of at least Critically Endangered (≥80% decline over any 50-year period including the present and future), although it seems unlikely to collapse entirely if fragments of forest remain in less accessible mountain terrain. In conclusion, the ecosystem is considered Critically Endangered (CR) under subcriterion A2b (Portillo, 2014). Information on the most likely shape of decline can help determine which of these two plausible categories should be reported as the best estimate.

5.3.4. Documentation

Assessors should (i) cite data repositories for time-series maps of ecosystem distributions used in the assessment (see the RLE website for a list of preferred spatial data repositories: www.iucnrle.org); (ii) provide full bibliographic references; (iii) justify why the spatial data used are an adequate representation of the distribution of the focal ecosystem type; (iv) justify assumptions and alternative scenarios used to interpolate, extrapolate or predict changes in distribution from the available data; and (v) explain the methods of calculation including the assumed threshold of collapse. In addition, assessors are encouraged to describe the source of the spatial data (such as satellite sensor type) and its spatial resolution (grain size), and comment on the accuracy of all classified maps.

6. Criterion B. Restricted geographic distribution

6.1. Theory

The size of the geographic distribution of an ecosystem influences its risk of collapse when confronted with a spatially explicit threat or catastrophe (Keith et al., 2013, 2018). In general, ecosystems that are widely distributed or exist across multiple independent patches are at lower risk from catastrophes, disturbance events or any other threats that exhibit a degree of spatial contagion (e.g. invasions, pollution, fire, forestry operations, and hydrological or regional climate change). The primary role of criterion B is to identify ecosystems whose distribution is so restricted that they are at risk of collapse from the chance occurrence of a single or few threatening events (Rodríguez et al., 2015). It uses simplified metrics of risk spreading or insurance effects – the probability that some portion of the distribution of an ecosystem type will be outside the spatial footprint of, and thus unaffected by any single threatening event (Figure 13). Criterion B also includes an approximation for an estimate of occupied habitat for component biota, which is positively related to population viability irrespective of exposure to catastrophic events.

Two measures of ecosystem distribution serve as standardised proxies of insurance effects that represent conceptually different aspects of geographic range size for both species (Gaston, 1994; Gaston & Fuller, 2009) and ecosystems (Keith et al., 2013; Murray et al., 2017). Extent of occurrence (EOO) (subcriterion B1) measures the spread of risk over a contiguous area that encloses all known spatial occurrences using a minimum convex polygon. In contrast, area of occupancy (AOO) (subcriterion B2) measures the spread of risk among occupied natural patches through a count of occupied grid cells (Keith et al., 2013).

AOO and EOO have been shown to perform better than other spatial distribution metrics (such as mean patch area, core area) for predicting the risk of ecosystem collapse in landscapes subject to stochastic threats (Murray et al., 2017). These measurement protocols are appropriate for all assessment units, including ecosystem types with depth dimensions or particular distribution patterns (Keith et al., 2018), such as linearly occurring ecosystem types (e.g. rivers and streams, gallery forest, etc.).

6.2. Thresholds and subcriteria

An ecosystem may be listed under criterion B if it meets the thresholds for either of three subcriteria (B1, B2 and B3), which indicate restricted geographic distribution as follows:

Subcriterion	*Measure of geographic distribution*	*CR*	*EN*	*VU*
B1	Area of a minimum convex polygon (km^2) enclosing all occurrences (extent of occurrence, EOO) is: **AND** at least one of the following (a–c): (a) An observed or inferred continuing decline in **ANY OF**: i. spatial extent of the ecosystem; **OR** ii. environmental quality appropriate to the characteristic biota of the ecosystem; **OR** iii. biotic interactions* appropriate to the characteristic biota of the ecosystem. (b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. (c) Ecosystem exists at:	≤ 2,000 1 threat-defined location	≤ 20,000 ≤ 5 threat-defined locations	≤50,000 ≤ 10 threat-defined locations
B2	The number of 10×10 km^2 grid cells occupied (area of occupancy, AOO) are: **AND** at least one of a–c above (same as for B1).	≤ 2	≤ 20	≤ 50
B3	A very small number of threat-defined locations (generally fewer than 5) **AND** prone to the effects of human activities or stochastic events within a very short time period, and thus capable of Collapse or becoming Critically Endangered (CR) within a very short time period (B3 can only lead to a listing as Vulnerable, VU).			VU

** Note: The full text of Clause B1(a)iii of the criteria, is expressed as "a measure of disruption to biotic interactions appropriate to…" (IUCN, 2016). Alternative phrasing given in this summary table avoids potential misinterpretation associated with double-negative expression when that phrase is read in context with clause B1(a), "continuing decline" in "... [disruption to] biotic interactions". An identical interpretation applies to the corresponding clause, B2(a)iii.*

6.3. Applying criterion B

6.3.1. Data requirements

The geographic distribution of an ecosystem type is assessed under criterion B with two standardised spatial metrics: the extent of occurrence (EOO) and the area of occupancy (AOO) (Gaston & Fuller, 2009; Keith et al., 2013); and a third qualitatively interpreted metric, the number of threat-defined locations. In addition, assessment of criterion B requires an evaluation (qualitative or quantitative) of whether continuing declines in spatial extent, environmental quality, or increasing disruption of biotic interactions are occurring or likely to occur as a result of threats. Accurate maps of the current distribution of an ecosystem type, information about the direction of current trends, and an understanding of the threats influencing the ecosystem are needed (Keith et al., 2013). For further information on data sources and the requirements of distribution maps for application in the RLE, refer to Section 5.3 Applying criterion A.

In some cases, spatial data may be insufficient to estimate EOO or AOO, but there is evidence that a small number of plausible threatening events may cause an ecosystem to become Critically Endangered or Collapsed within the near future. Such ecosystem types may be eligible for listing as Vulnerable under criterion B3 if they occupy few threat-defined locations relative to the extent of threatening events. Distribution maps, locality records or expert knowledge are required to determine the number of threat-defined locations in which an ecosystem occurs.

6.3.2. Methods for assessing criteria B1 and B2

Ensuring standardised methods are applied to the diverse sources of spatial data available for Red List assessments is essential to ensure accurate and consistent assessment outcomes and avoid artefacts of data and methods. Therefore, EOO and AOO must always be measured in ways that comply with the methods specified below:

1. **Extent of occurrence (EOO).** The EOO of an ecosystem is the area (km^2) of a minimum convex polygon – the smallest polygon in which no internal angle exceeds 180° that encompasses all known current spatial occurrences of the ecosystem type. The minimum convex polygon (also known as a convex hull) must not exclude any areas, discontinuities or disjunctions, regardless of whether the ecosystem can occur in those areas or not. Regions such as oceans (for terrestrial ecosystems), land (for coastal or marine ecosystems), or areas outside the study area (such as in a different country) must remain included within the minimum convex polygon to ensure that this standardised method is comparable across ecosystem types. In addition, these features contribute to spreading risks across the distribution of the ecosystem by making different parts of its distribution more spatially independent.

2. **Area of occupancy (AOO).** Estimates of ecosystem extent are highly sensitive to the grain size (spatial resolution) of source maps (Nicholson et al., 2009), so all measures of AOO of an ecosystem type must be standardised to a common spatial grain (Keith et al., 2018). The AOO of an ecosystem defined in the RLE is determined by counting the number of 10 × 10 km^2 grid cells that contain the ecosystem type. This relatively large grain size is applied for four reasons: (i) ecosystem boundaries are inherently vague (Regan et al., 2002), so it is easier to determine that an ecosystem occurrence falls within a larger grid cell than a smaller one; (ii) larger cells may be required to diagnose the presence of ecosystems characterised by processes that operate over large spatial scales, or possess diagnostic features that are sparse, cryptic, clustered or mobile (e.g. pelagic or artesian systems); (iii) larger cells allow AOO estimation even when high resolution distribution data are limited; and (iv) simulation studies have indicated that larger cells better predict risk in the face of real-world threat events than finer scale cells (Keith et al., 2018). A global 10 × 10 km gridded dataset suitable for this purpose is available via a public data repository in raster and vector formats (Murray, 2017).

   Some ecosystem distributions comprise a highly skewed distribution of patch sizes. In these cases, large numbers of small patches contribute a negligible risk-spreading effect compared to that of larger patches. Assessors should therefore apply a correction by excluding from the AOO those grid cells that, collectively, contain patches of the ecosystem type that account for less than 1% of the total mapped area of the ecosystem type, thus always including 99% of the ecosystem extent.

   The protocol for this adjustment includes the following steps:
   - Intersect AOO grid with the ecosystem's distribution map.
   - Calculate extent of the ecosystem type in each grid cell ('area') and sum these areas to obtain the total ecosystem area ('total area').
   - Arrange grid cells in ascending order based on their area (smaller first).
   - Calculate accumulated sum of area per cell ('cumulative area').

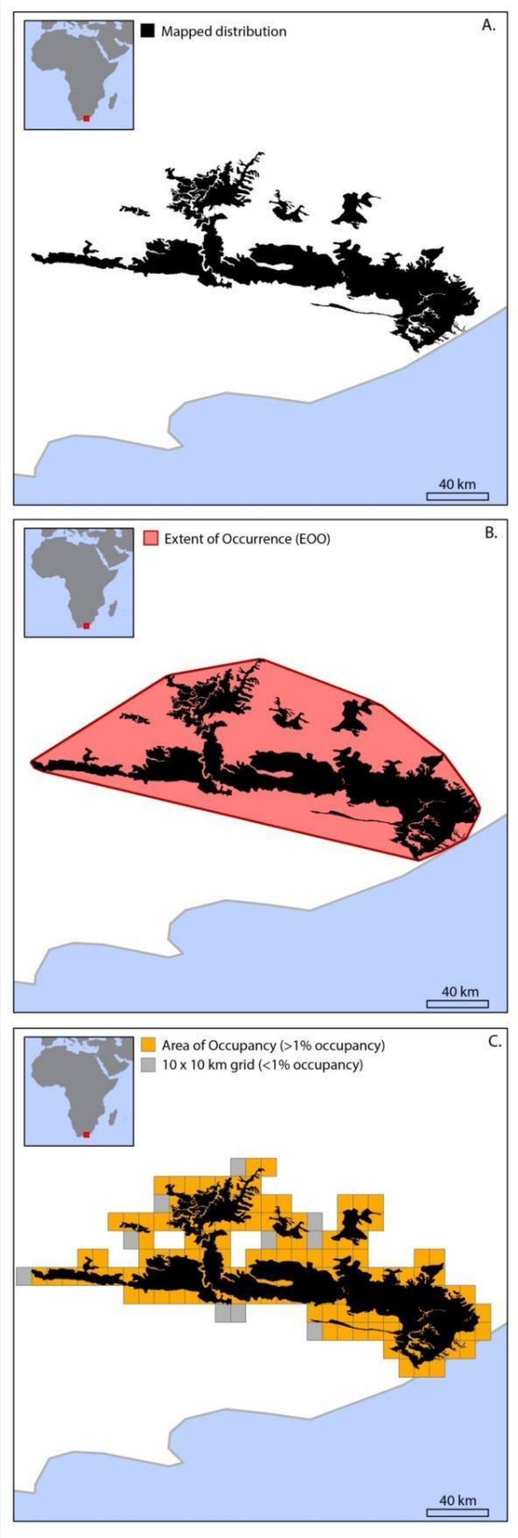
- Calculate ‘cumulative proportion’ by dividing ‘cumulative area’ by ‘total area’ (cumulative proportion takes values between 0 and 1).
- Calculate AOO by counting the number of cells with a ‘cumulative proportion’ greater than 0.01 (i.e. exclude cells that in combination account for up to 1% of the total mapped extent of the ecosystem type).

An earlier correction protocol proposed in Guidelines version 1.1 (Bland et al., 2017) recommended exclusion of cells in which the ecosystem type accounts for less than 1% of the cell area (i.e. <1 km^2). However, such a correction results in exclusion of all AOO grid cells in extreme cases if mapped occurrences of an ecosystem type are small and widely separated and thus always occupy less than 1% of cell area. To avoid the resulting underestimation of insurance effects, assessors were required to determine which ecosystem types the AOO correction should be applied to. This new protocol, which excludes 1% of the total mapped area provides an independent statistical rule for cell exclusions and therefore requires no subjective judgement about when it should be applied. Comparative trials of the two correction protocols indicate that the new formulation limits underestimation of insurance by removing the tail of skewed patch size distributions from the AOO count, but results in smaller and more consistent corrections across ecosystem types with different distribution patterns.

Several spatial tools have been developed to assist in measuring the EOO and AOO of an ecosystem type. These will become available on the IUCN Red List of Ecosystems website: www.iucnrle.org/rle-material-and-tools.

To be eligible for listing under subcriteria B1 or B2, an ecosystem must first meet the EOO or AOO thresholds that delineate threat categories, as well as at least one of three conditions that address various forms of decline. These conditions distinguish restricted ecosystems at appreciable risk of collapse from those that persist over long time scales within small stable ranges (Keith et al., 2013). Only qualitative evidence of continuing decline is required to invoke the subcriteria, but relatively high standards of evidence should be applied.

Box 12. The extent of occurrence (EOO) and area of occupancy (AOO) of an ecosystem (criterion B).



The distribution of the **Great Fish Thicket**, South Africa (Mucina & Rutherford, 2006), is depicted by a raster dataset with a spatial resolution of 30 x 30 m (A). As mapped, the area of the Great Fish Thicket ecosystem type is 6,763.4 km^2.

A minimum convex polygon – the smallest polygon that encompasses all known occurrences of the ecosystem type in which no internal angle exceeds 180° – is applied to estimate the extent of occurrence (EOO) for assessment under criterion B1 (B). The area of the minimum convex polygon is 18,359.2 km^2, meeting the initial requirements for an Endangered classification under criterion B1.

To estimate the area of occupancy (AOO) for assessment under criterion B2, the number of cells covered by the ecosystem type is required (C). The standardised measurement of AOO ensures that distribution data mapped at varying resolutions is generalised to a common 10 x 10 km grid, allowing consistent comparisons across ecosystem types and neutralising differences in the granularity of available spatial data for different ecosystem types. First, a 10 x 10 km grid is applied to the ecosystem type, indicating that 155 10 x 10 km grid cells intersect the distribution map (shown in orange and grey). Second, when the number of cells that contain very small patches (<1% of total mapped ecosystem area) that negligibly contribute to risk spreading are excluded (shown in grey), the AOO is measured as 145 grid cells (shown in orange). This AOO is greater than the thresholds for classification in a threatened category under B2.

Finally, to be eligible for listing in a threatened category under criterion B, qualitative evidence of continuing decline is also required. In this case, the Great Fish Thicket ecosystem type does not meet any of the additional subcriteria, and is thus assigned an overall classification of Least Concern.

Continuing decline

Condition B1a and B2a address continuing declines in (i) ecosystem distribution (i.e. spatial extent), (ii) abiotic environment or (iii) biotic processes. To invoke this condition, the declines must (i) reduce the ability of an ecosystem to sustain its characteristic native biota; (ii) be non-trivial in magnitude; and (iii), be more likely than not to continue into the future (Table 4). Episodic or intermittent declines qualify as continuing, so long as they are recurring and uncompensated by increases of comparable magnitude. Downward phases of cyclical changes or fluctuations do not qualify as continuing declines. Evidence of compensatory increases should generally be detectable between successive episodes of decline in order to infer fluctuations, rather than continuing decline, in extent or the biotic or abiotic processes of an ecosystem type. These requirements imply an understanding of the causes of decline to support a correct inference. Even with an understanding of mechanisms, inferences about continuing declines may be uncertain, especially where fluctuations are known or likely to occur, for example in boom-bust ecosystems with large multi-year fluctuations (Dickman et al., 2014). In cases where continuing declines are equally likely to be occurring or not occurring, upper and lower bounds of the status under criterion B should be estimated by propagating both scenarios through the criteria.

Assessing threatening processes

To invoke an observed or inferred threatening process under conditions B1b and B2b, assessors must first identify one or more specific threatening processes, and also, present convincing and generally agreed evidence that such threats are very likely (Table 4) to cause continuing declines within the next two decades. These requirements imply an understanding of how the threats affect the defining features of the ecosystem and the timing of their effects. Speculation about generic threats with uncertain impacts or onset, of itself, does not meet the required standard of evidence for invoking threatening processes under conditions B1a or B2b and is discouraged. Relevant evidence includes observations of similar threats in the past or on similar ecosystem types or settings, as well as accumulated knowledge about the behaviour and nature of the threat itself.

Evidence of past or current declines is not essential for inferring a threatening process under conditions B1b and B2b if there is plausible evidence inferred from serious and imminent threats likely to cause future declines in ecosystem distribution or function within the next two decades. For example, climate change may over time cause the distribution of the ecosystem type to shift, contract, and/or fragment or cause certain functions to decline. To infer such threats, the mechanisms by which climate change drives decline need to be identified and a non-trivial ecosystem response needs to be plausible within the next 20 years (e.g. through increased incidence or impact of events such as heat waves).

Threat-defined locations

Conditions B1c, B2c and B3 require an estimate of the number of threat-defined locations that are occupied relative to the extent of serious plausible threats. A threat-defined location is defined as a geographically or ecologically distinct area in which a single threatening event can rapidly affect all occurrences of an ecosystem type (Glossary). Note that in the context of RLE assessment, a threat-defined location is not necessarily the same as a locality or site of occurrence; rather, a threat-defined location is defined entirely by the spatial extent of the most serious plausible threats (this is consistent with the definition of locations for The IUCN Red List of Threatened Species). The size of the threat-defined location depends on the maximum plausible area covered by a threat event and may include part of one, or many separate patches/occurrences of an ecosystem type (Figure 13).

A threat event is an incidence of a threatening process, capable of causing decline or degradation of an ecosystem's properties that occurs independently of other such events in time and space (Glossary). A threat event or series of events at different times that affect two or more spatially separate occurrences of an ecosystem may be interpreted as a single threat event if they are driven by a common causal factor(s) or occur under similar conditions, such that cumulatively, they generate contagion in the spatial footprint of the threat over time. For example, in a commercially exploited forest, individual logged patches do not represent independent threat events if they are driven by the same market demand, are harvested under the same regulatory regime or occur within the same logging concession. Rather, the collective suite of patches affected or potentially affected by the logging process should be interpreted as a single independent threat event (see table in Box 13).

Where an ecosystem type is affected by more than one threatening event, threat-defined locations should be defined by considering the most serious plausible threat (IUCN, 2012). Where an ecosystem type is not affected by any threatening events, the number of threat-defined locations cannot be estimated and the conditions or subcriterion that refer to the number of locations will not be met. Box 13 contains further guidance and examples to support the interpretation of the threat-defined location concept, with a more detailed example in Box 14.

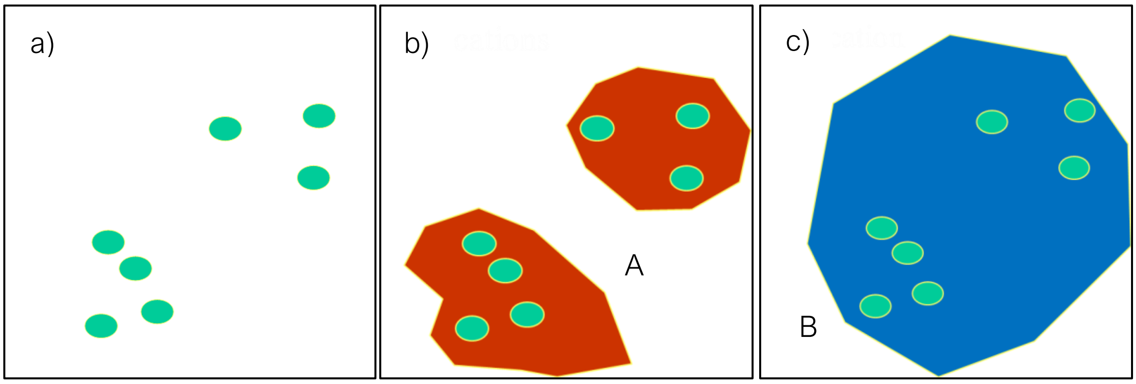


Figure 13. Threat-defined locations. Source: Authors.
Stylised distribution of an ecosystem type (a) that occurs in seven discrete patches (green ovals). Expected extent of spatial footprints of threat A (b) and threat B (c) in relation to the distribution of the ecosystem type. The expected extent of spatial footprints may be drawn from inferences based on historical events and relevant precedents, development or land use plans, knowledge on regional climate processes or contagion (e.g. spread of invasive species), market and supply properties and other drivers of resource use, or predictive models or similar processes. In this example, if threat A is the most serious plausible threat, the ecosystem type occurs in two threat-defined locations. If threat B is the most serious plausible threat, then the ecosystem type occurs in one threat-defined location. If threats A and B are equally serious, then Threat B poses greater risks to the ecosystem type because of its greater spatial extent and thus the distribution should be interpreted as one threat-defined location.

Where climate change is the most serious plausible threat, assessors must examine if these changes are likely to occur uniformly (i.e., one threat-defined location) or unequally across the whole distribution (i.e., multiple threat-defined locations). For example, tropical glaciers in Ecuador are located at different elevations, glaciers at lower elevations (below 4,500 m) are exposed to higher increases in mean annual temperature and are likely to lose ice mass more quickly than higher elevation glaciers (Ferrer-Paris & Keith, 2024). In most cases, climate change will be a secondary threat that enhances one or more primary threats to the persistence of the ecosystem type, such as increased frequency or severity of fires, sea-level rise, storm regimes, hydrology or spread of disease. In such cases, assessors should use these proximal threats to determine the number of threat-defined locations. See Box 13 for guidance on estimating the number of locations.

Box 13. Estimating the number of threat-defined locations.

The steps required to estimate the number of threat-defined locations include:

1. Identify all plausible threatening processes (see Glossary) that may affect the distribution of the ecosystem type or its characteristic biotic or abiotic processes. In the first instance,this step should draw information from the ecosystem description (Section 4.2 Describing the unit of assessment).
2. Rank the seriousness of plausible threats in terms of their severity (i.e. maximum potential impact on ecosystem properties) and the maximum potential spatial extent of independent threat events (see Glossary). Note that the most severe threat may not be the most extensive, and vice versa. Where uncertainty exists about which threat is the most serious, two or more threats may be designated as similarly 'serious' and considered in step 3.
3. Interpret the geography of the ecosystem distribution and the number of independent threat events (see Glossary), based on the maximum potential spatial footprint, that would be required to affect the entire distribution of the ecosystem type (Figure 13). This requires consideration of disjunctions and barriers that may prevent segments of the ecosystem distribution from being exposed to the same threat event.
4. The most serious plausible threat may not extend across the full distribution of the ecosystem type (e.g. protected areas may be effective in preventing habitat conversion). In such cases, the next most serious threat(s) should be used to estimate the number of threat-defined locations in the remaining part(s) of the distribution. The total number of threat-defined locations for the ecosystem type will be the sum of locations in each part of its distribution.
5. Where there are no plausible threats to the ecosystem type, subcriteria B1(c), B2(c) and B3 are not met. This should be distinguished from cases in which there is insufficient information to assess the number of threat-defined locations (i.e. a Data Deficient outcome).
6. If no threats can be identified in part of the distribution of the ecosystem type, the following options will be appropriate under different circumstances: (i) if most of the distribution has no threat, the subcriteria that refer to the number of threat-defined locations (B1(c), B2(c), B3) are not met; (ii) the number of threat-defined locations is based on the smallest size of locations in the areas with identified threats; (iii) the number of threat-defined locations is based on the most likely threat that could affect the currently-unaffected areas in the future; (iv) if insufficient information is available to apply options (ii) or (iii), and less than 30% of the distribution has no threat, Near Threatened status may be invoked for subcriteria B1(c) and B2(c) if the number of threat-defined locations in the remainder of the distribution is nine or less, and for B3 if the number is four or less.

7. Where two or more plausible threatening processes are similarly serious, the number of threat-defined locations should be based on the threat that produces the smallest estimate.
8. Where there is uncertainty in the number of threat-defined locations based on steps 1–5 (i.e. most cases), assessors are encouraged to make bounded estimates (see Section 3.3.3 Standards of evidence and dealing with uncertainty).

Assessor's reasoning on the interpretation of steps 1–8 (especially in step 6) should be documented transparently to justify the estimated number of threat-defined locations. The following examples may assist estimation of threat-defined locations.

Threat type	*Considerations*
Habitat conversion (urban/peri-urban)	In urban and peri-urban landscapes, conversion of natural or semi-natural ecosystems are typically driven by socio-economic factors and engineering or service factors, which both generate contagion. These should be the primary consideration in defining independent threat events, i.e. the relevant spatial unit is the jurisdiction in which population pressures, social needs and planning regulation occur (e.g. a planning district, an entire city or conurbation and its peri-urban halo, a district encompassing related villages influenced by similar socio-economic events). Footprints of individual buildings or service infrastructure, neighbourhood or subdivision precincts are not threat-defined locations.
Habitat conversion (rural)	In rural landscapes, habitat conversion is driven by land-use changes and the growth of cities and towns. These factors should be the primary considerations for defining threat locations. The threat-defined location should be an entire area or village that contributes to population needs and planning regulations.
Habitat conversion (mining/industrial)	Mining and industrial infrastructure are driven by the location and extent of the resource being mined or processed, as well as the market for the resource. The resource supply source (e.g. cane field district for sugar mills, mineral reserve for mining) should be a primary consideration for threat-defined locations. Similar considerations apply to land surface, underground and sea-floor mining. In some cases, widely separated areas may be developing simultaneously in response to a common market demand. This may warrant their interpretation as a single threat-defined location.
Forest logging	Threat-defined locations will generally be determined by concessions (owned by the same or different operators) that fall under the same regulatory regime governed by similar types of practices and market pressures. Where illegal or unregulated harvest occurs, socio-economic and markets should be considered in delineating threat-defined locations.
Fishing	For freshwater fishing, waterbodies and their catchments should be considered in delineating locations, noting that spatial autocorrelation may occur across multiple adjacent catchments or clustered lakes depending on social factors and proximity. For freshwater, coastal and marine fisheries, regulatory jurisdictions and illegal or unregulated fishing are relevant considerations as for forest logging.
Hunting, control and persecution	Spatial patterns of threats posed by regulated hunting is determined by licencing and compliance jurisdictions, which should be an important consideration for threat-defined locations. However, illegal and unregulated hunting are often prevalent and require socio-economic and market factors, particularly proximity, attitudes and livelihood dependencies of local communities likely to be involved.

Invasive species and disease outbreaks	Threat-defined locations will depend on the dispersal and establishment biology of the invasive agent and its vectors of spread. Insights may be inferred from historical spread of the species of disease. In general, existing infestations alone will not be suitable as threat-defined locations.
Disturbance regimes: fires, floods and storms	Estimates of threat-defined locations for disturbance regimes, floods and storms are driven by both climatic and social factors. For example, social factors (and to some degree, lightning) determine the incidence of fire ignitions. Fires, floods and storms are also influenced by topography (catchments, terrain). Incidence of extreme weather events influence the spatial footprint of fires (e.g. extended droughts, periods of high wind, high temperature and low humidity), floods (e.g. intense rainfall events) and other ecosystem disturbances. Historical information on the largest recorded spatial footprints within respective regions provide some guidance for interpreting step 3. The nature of the threat needs to be understood (e.g. fire *per se* may not be a threat, but certain combinations of frequency, season or intensity of fire may be). Regional climates define the conditions, but should also consider temporal autocorrelation associated with regional interannual weather cycles, landscape heterogeneity, and for fire, social norms of burning (whereby large fires may burn complementary parts of a landscape in successive years, creating a larger footprint).
Freshwater extraction, diversion, flow barriers and impoundments	Threat-defined locations for water extraction depend on connectivity of flows. For example, disconnected lakes will be separate locations. Stream catchments or reaches could be locations depending on existing and potential localities of extraction points. Adjacent catchments could be within the same threat-defined location if within the same irrigation scheme or if the same water consumer source extracts from otherwise independent streams. Threat-defined locations for groundwater extraction will depend on connectivity of aquifers in relation to existing and potential extraction points.
Tectonic, volcanic, mass movement and tsunami events	Threat-defined locations for these geophysical events will depend on the point localities of sources and spread dynamics. Judgements will be guided largely by historical events and landscape and seafloor morphology.
Climate change – related threats	Climate change is a global threat, but it is important to understand how global changes will affect the proximal threats to the ecosystem. The spatial expression of climatic changes that can drive changes in abiotic and biotic processes of the ecosystems is likely to be very different from case-to-case, and interact with different landscape and geographical features. Ecosystems that are spread along latitudinal or altitudinal gradients or with occurrences isolated by orographic barriers (terrestrial ecosystems), in landscapes with complex geomorphology (wetlands and coastal ecosystems) or exposed to different oceanic currents (marine ecosystems) are likely to experience different rates of change. Understanding this geographical context must be considered in estimating the number of threat-defined locations.

6.3.3. Methods for assessing subcriterion B3

Subcriterion B3 requires only qualitative information on the distribution of an ecosystem and threats to its persistence. To compensate for this type of evidence (compared to quantitative estimates in other criteria), a higher standard of qualitative evidence is required. The highest category that can be invoked by subcriterion B3 is Vulnerable (i.e. ecosystem types are not eligible for Critically Endangered or Endangered status under criterion B3).

Subcriterion B3 comprises two parts which must both be met for an ecosystem type to qualify for Vulnerable status. First, the ecosystem type must have a very restricted distribution, generally with fewer than five threat-defined locations (Box 13). Second, the ecosystem type must be facing severe threats (human activities or stochastic events) within a very short time period in an uncertain future, and thus capable of collapse or becoming Critically Endangered within a very short time period. In other words, the impact of the threat is very likely (Table 4) to occur in the near future and its consequences are severe. Assessors have some flexibility to interpret the 'very short time period', but this generally means within the next two decades.

Box 14. Determining the number of threat-defined locations (criterion B).
Source: Adapted from Appendix S2.9 in Keith et al., 2013.

Coolibah-Black Box Woodland of south-eastern Australia

In its mature state, Coolibah-Black Box Woodland has an open structure with widely scattered trees, a variable cover of shrubs and grassy ground layer. The characteristic vertebrate fauna includes diverse assemblages of woodland and wetland bird species, many of which depend on tree hollows, other features of large trees or standing water for breeding and foraging (NSW Scientific Committee, 2004).

The most serious plausible threats are land clearing and changes to water regimes. Spatial patterns of land clearing show a high degree of contagion, with the best predictor of future clearing being the proximity of a patch to land parcels already cleared of native vegetation. A broad interpretation of threat-defined locations under subcriterion B3 identifies three jurisdictional zones with different regulatory controls on land clearing: the leasehold Western Division of New South Wales, the freehold Central Division of New South Wales, and Queensland. This results in an estimate of three threat-defined locations as defined by land clearing. A narrower interpretation of threat-defined locations based on neighbourhoods of contagion would produce an estimate of more than five. Small protected areas are excluded from these threat-defined locations, as they are not threatened by land clearing. These areas were assessed by considering the next most serious plausible threat: changes to water regimes. As protected areas are located in at least two different sub-catchments with different water management infrastructure, there are at least two further threat-defined locations. Hence the most precautionary interpretation produces an estimate of five threat-defined locations, although it is likely that there are more.

Based on current rates of depletion due to land clearing (subcriterion A1) and current rates of environmental degradation due to changes in water regime (subcriterion C1), the ecosystem is unlikely to collapse or become Critically Endangered within the near future (c. 20 years). The ecosystem type therefore does not meet subcriterion B3, so the status of the ecosystem type is Least Concern under this subcriterion.

Cape Flats Sand Fynbos of South Africa

Cape Flats Sand Fynbos is a species-rich, dense, moderately tall shrubland with scattered emergent shrubs (Rebelo et al., 2006). The ecosystem type is an edaphically determined species assemblage restricted to tertiary acid, deep grey regic sands at low elevations (20–200 m) on flat to undulating terrain. Cape Flats Sand Fynbos is restricted to the Western Cape province of South Africa, almost entirely within the limits of the City of Cape Town. The most severe threat to the ecosystem type is habitat destruction associated with urban development (Rebelo et al., 2006; Wood et al., 1994).

Occurrences that are currently within proclaimed reserves are protected from this threat, although these stands are threatened by invasion of exotic plants (Rebelo et al., 2006). As the entire distribution of the ecosystem type is within the City of Cape Town, the unproclaimed remnant vegetation is subject to the same development pressures, regulatory regimes and planning authority. The distribution is therefore interpreted as two semi-independent threat-defined locations; one outside protected areas (threatened by habitat destruction and invasive plants) and one within protected areas (threatened by invasive plants, but not habitat destruction).

Given the severe and immediate nature of the threats, the ecosystem type is prone to the effects of human activity or stochastic events such that it is capable of collapse or becoming Critically Endangered within a very short time period. The status of the ecosystem type is thus Vulnerable under subcriterion B3.

6.3.4. Documentation

For each assessment of an ecosystem type, assessors should: (i) provide the current maps of ecosystem distributions (similar to those in Box 12) that were used to estimate the EOO and AOO and to determine the number of threat-defined locations; (ii) provide full bibliographic references; (iii) justify why the spatial data used is an adequate representation of the distribution of the focal ecosystem type (if not already done so for criterion A); (iv) explain why a correction to AOO was justified if one was applied; (v) justify inferences about continuing declines, and threats that may lead to continuing declines within the next 20 years; and (vi) justify estimates of the number of threat-defined locations through reference to the most serious plausible threats and their spatial characteristics (Box 13). As with assessments under criterion A, description of the source (such as satellite sensor type), accuracy and the spatial resolution (grain size) of all spatial data used in an assessment is strongly encouraged. Deposition of spatial data used for AOO and EOO into an appropriate data repository is encouraged and should be referenced in the documentation supporting the assessment.

7. Criterion C and D. Environmental degradation and disruption of biotic processes

7.1. Theory

The IUCN Red List of Ecosystems (RLE) risk model aligns with community assembly theory, which defines abiotic and biotic selection filters that influence the establishment and persistence of species within communities (HilleRisLambers et al., 2012). At the ecosystem level, these 'filters' represent physical and biological properties and processes that characterise an ecosystem type. Directional changes in these characteristic properties and processes can be symptomatic of ecosystem transformation and eventual collapse. The RLE risk model therefore defines two criteria for assessing declines in ecosystem functions or processes based on symptoms of abiotic (environmental) degradation (criterion C) and disruption of biotic processes, including declines of biota with key functional roles (criterion D). Separate criteria are necessary because the causes, effects and mechanisms of functional decline differ fundamentally between the degradation of abiotic and biotic components of ecosystems (Keith et al., 2013). The indicators for assessing criteria C and D must therefore be specific to an ecosystem type or a group of ecosystem types that share critical properties associated with functional change (e.g. an Ecosystem Functional Group in level 3 of the Global Ecosystem Typology, Section 3.1.1 Ecosystem typologies and Section 4.2.1 Classification; Keith et al., 2022).

Some threatening processes can affect both biotic and abiotic properties, but these should still be assessed independently so that the overall status is determined by the most severe symptom of risk. Criteria C and D must therefore be assessed separately using ecosystem-specific indicators (state variables) that represent defining features of the ecosystem type and hypothesised symptoms of decline in abiotic (criterion C) or biotic (criterion D) properties, while also considering any interactive effects in the analysis (e.g. Obura et al., 2022). For further discussion see principles for indicator selection (Table 11) and associated text in Section 7.3.1 Selection of indicator variables.

Assessment of Red List criteria C and D comprises three conceptual components: the severity, extent and time frame of ecosystem degradation. Different subcriteria address the standardised time frames for recent past (subcriteria C1 & D1), future (subcriteria C2 & D2) and historic past (subcriteria C3 & D3) defined in Section 3.3.1 Time frames. The severity and extent components of degradation are assessed in combination against standardised thresholds, allowing the criteria to accommodate different pathways of decline (Figure 14).

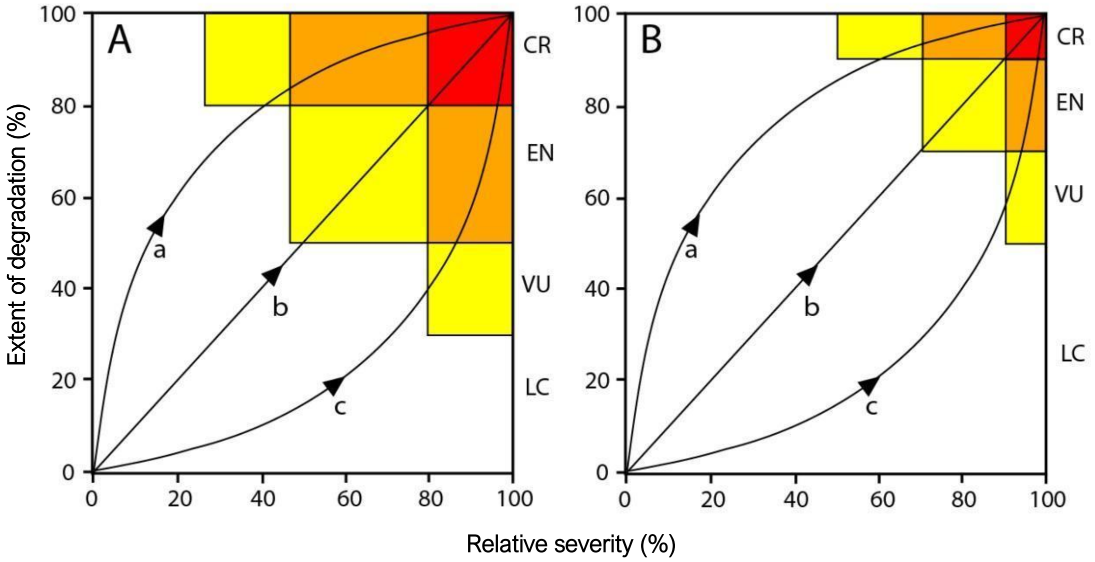


Figure 14. Contrasting pathways of environmental or biotic degradation and their corresponding risk classifications under criteria C1, C2, D1, D2 (A) or C3, D3 (B).
Source: Authors.
Arrows indicate direction of temporal changes: (a) initially widespread and relatively benign degradation, which increases in severity, (b) severity and extent of degradation increase at similar rates, (c) localised but severe degradation, later becoming more widespread.

Criterion C. Environmental (abiotic) degradation is the deterioration of the physical, non-living attributes that have a defining role in ecological processes and/or the distribution of an ecosystem type. Environmental degradation reduces the capacity of an ecosystem to sustain its characteristic biota. For example, declines in limiting resources (niche dimensions defined by availability of water, light, nutrients, etc.), changes in ambient conditions that affect the availability of resources or ability of organisms to acquire them (e.g. temperature, salinity), or changes in disturbance regimes (floods, fires, storms) may in turn transform niche diversity and/or ecological processes in a range of terrestrial, freshwater and marine ecosystems (Harpole & Tilman, 2007).

Criterion D. The persistence of biota within ecosystems depends on biotic processes and interactions. This includes competitive, predatory, facilitatory, mutualistic, trophic and pathogenic processes; mobile links (e.g. seasonal migration); and species invasions, all of which contribute to the biotic filters in ecosystem assembly. Biodiversity loss, especially the loss of key functional components, reduces the capacity of ecosystems to capture resources, produce biomass, decompose organic matter and recycle carbon, water and nutrients, and also reduces the stability of these functions through time (Cardinale et al., 2012). The identity of organisms within a system controls its functioning as key taxa make disproportionate contributions to ecosystem functions. The diversity of organisms is also important, because niche partitioning and positive species interactions promote complementary contributions to ecosystem functions.

Feedback interactions are crucial for an ecosystem type to absorb environmental change while maintaining characteristic biota and processes. Conversely, significant disruptions to biotic processes and interactions can cause collapse, regime shifts and re-organisation into novel ecosystems (Thébault & Loreau, 2005). Disruption of interactions through trophic cascades is one of five major threats to biodiversity (Diamond, 1989), although non-trophic interactions also play important roles (Fontaine et al., 2005; Goudard & Loreau, 2008). Certain ecosystem types may be especially sensitive to disruption of biotic processes and interactions, such as systems with strong top-down trophic regulation, with many mutualistic or facilitation interactions that are strongly dependent on mobile links, and where positive feedbacks operate between the biota and disturbance regimes.

7.2. Thresholds and subcriteria

The thresholds for severity and extent of degradation are assessed in a combined manner to determine the category of risk. Ecosystems are listed as Critically Endangered (CR) if degradation is both extremely severe (≥ 80% relative severity) and extensive (across ≥ 80% of the distribution). Ecosystems may be eligible for listing in lower threat categories if they are undergoing very severe but localised degradation or less severe degradation over extensive areas (Figure 14). Ecosystems that just fail to meet the thresholds for the Vulnerable category may be assigned to the Near Threatened category. For example, an ecosystem undergoing > 80% decline in environmental quality over 20–30% of its distribution, or 20–30% decline over 30–40% of its distribution could qualify as Near Threatened.

Criterion C

An ecosystem may be listed under criterion C if it meets the thresholds for any of four subcriteria (C1, C2a, C2b or C3), which express different levels of environmental degradation over the following time frames:

Subcriterion	*Time frame*		*Relative severity (%)*		
C1	The past 50 years based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
		≥ 80	CR	EN	VU
		≥ 50	EN	VU	
		≥ 30	VU		
C2	C2a. The next 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** C2b. Any 50-year period including the past, present and future, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
		≥ 80	CR	EN	VU
		≥ 50	EN	VU	
		≥ 30	VU		
C3	Since 1750 based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 90	≥ 70	≥ 50
		≥ 90	CR	EN	VU
		≥ 70	EN	VU	
		≥ 50	VU		

Criterion D

An ecosystem may be listed under criterion D if it meets the thresholds for any of four subcriteria (D1, D2a, D2b or D3), which express different levels of biotic disruption over the following time frames:

Subcriterion	*Time frame*	*Relative severity (%)*			
D1	The past 50 years based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
		≥ 80	CR	EN	VU
		≥ 50	EN	VU	
		≥ 30	VU		
D2	D2a. The next 50 years, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** D2b. Any 50-year period including the past, present and future, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
		≥ 80	CR	EN	VU
		≥ 50	EN	VU	
		≥ 30	VU		
D3	Since 1750 based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 90	≥ 70	≥ 50
		≥ 90	CR	EN	VU
		≥ 70	EN	VU	
		≥ 50	VU		

7.3. Applying criteria C and D

The first step to applying criteria C and D is to diagnose the key mechanisms and pathways of ecosystem degradation (see Section 4.2.5 Conceptual models for guidance on diagnosis with the aid of conceptual models). Once the expected symptoms of degradation are deduced from these mechanisms and pathways, assessors can identify candidate indicators for assessing the severity and extent of degradation.

7.3.1. Selection of indicator variables

In both strategic and systematic assessments, the state variables with direct and clear cause-effect relationships and the greatest sensitivity to declines in characteristic native biota or ecological processes will be the most suitable indicators. Table 11 identifies principles that guide the selection of appropriate indicators.

An understanding of the causes and symptoms of ecosystem degradation (Principle 1, Table 11) can come either from direct observation or inference based on comparable ecosystem types. A carefully developed conceptual model (Section 4.2.5 Conceptual models) can help to diagnose threatening processes (Section 4.2.6 Threats) and their effects on ecosystem properties that may ultimately drive transitions between healthy and collapsed states.

Diagrammatic assembly models for relevant Ecosystem Functional Groups (Keith et al., 2022) may be a useful starting point for the development of ecosystem-specific conceptual models. Understanding the pathways of degradation and their symptoms should inform the identification of data for measuring degradation of key abiotic and biotic properties of the ecosystem type under assessment (Principle 2, Table 11). Suitable indicators must enable assessment of both the severity and extent of degradation, through scalar and spatial components of the data (Principle 3, Table 11; see Section 7.3.4 Calculating relative severity).

Suitable indicators must also enable an inferred threshold of collapse (Principle 4, Table 11; Section 4.2.7 Describing collapsed states). The selection of indicators and the definition of collapse (Section 3.2 Ecosystem collapse) and associated thresholds need to be closely aligned and may require iteration. Suitable indicators should enable a threshold of collapse to be practically estimated from available information, with upper and lower bounds of the estimate reflecting uncertainty (see Box 15 for an example). The information used to infer thresholds of collapse may include:

1. Evidence on physiological tolerance of key biota.
2. Observations of local collapse under extreme conditions or events.
3. Distributional limits of system types along relevant environmental gradients.
4. Trait-based generalisations about the sensitivities of key biota.
5. Modelled dynamics or distributions that identify limits of ecosystem function or persistence.

Indicator selection must therefore consider the form of evidence available on ecosystem collapse. Section 9.1.3 Setting collapse thresholds offers further discussion on estimation of collapse thresholds and indicator selection to assess risks of ecosystem collapse related to climate change.

Indicator selection typically involves a trade-off between indicator specificity, data availability and the practicality of assessing multiple, functionally similar ecosystem types. Principle 5 (Table 11) should receive weighted consideration in such trade-offs, to the point where limited data availability precludes application. Irrespective of whether comprehensive data on direct indicators are available, assessors should not use highly abstract indicators for assessing criteria C or D. Where data on direct indicators are patchy, it may be possible to use them to calibrate less direct indicators for which more data exist. In some cases, the same indicators may be used to assess similar degradation processes that affect multiple ecosystem types, so long as these share similar mechanisms and pathways of degradation (e.g. Kontula & Raunio, 2019; Skowno & Monyeki, 2021). For example, an assessment of several wetland ecosystem types in which declines in water quality are associated with loss of wetland biota could use relevant water quality indicators to assess criterion C and richness or abundance of sensitive wetland biota to assess criterion D.

Table 11. Indicator selection principles for assessing criteria C and D.

Principle	*Example*
1. Indicator selection should be based on an understanding of the causes and symptoms of ecosystem degradation.	Intensive timber harvest in boreal forest ecosystems causes structural simplification and loss of biota dependent on primary forest habitat (Kontula & Raunio 2019). Exposure of photic coral reefs to increased frequency and severity of heat waves causes bleaching, reduced coral cover and niche diversity (Obura et al., 2021).
2. Indicators must be ecosystem-specific, i.e. relate to ecological properties of particular ecosystem types, or groups of functionally similar ecosystem types, and their threats. Applying the same generic indices across functionally contrasting ecosystem types is unlikely to assess degradation accurately if key processes differ among these ecosystems.	See Table 12 and Table 13.
3. A suitable indicator comprises a scalar component that measures the severity of decline, and a spatial component that indicates the extent of degradation, in characteristic ecosystem processes or properties.	Data on levels of water extraction and surface area for individual wetlands were combined to assess the relative severity of environmental degradation over the entire area of the swamps, marshes and lakes of the Murray-Darling Basin (Keith et al., 2013).
4. The choice of indicators must enable estimation of threshold values that mark the collapse of an ecosystem type and its transformation into another ecosystem type.	See Figure 3, Box 16 and Section 9.1.3 Setting collapse thresholds.
5. When multiple indicators are used to assess criterion C or D, the relationship between them should be made clear. The rationale for assessing them individually as independent indicators or within a combined indicator should be explicit and well justified. - Indicators should generally be based on the simplest, most informative and direct variable, with redundant correlated variables excluded. - Where a combination of two or more interdependent variables is ecologically and mathematically justified, it should be based on robust, transparent rules with documented rationale and justification. - Combining different pathways of degradation into a single scalar index (e.g. via an additive score) is discouraged because generic indices generally submerge complex relationships in ways that are not transparent, poorly understood and have unintended effects on index values.	See Section 7.3.2 Multiple indicator variables Icy substrates are key abiotic components of glacier ecosystems in decline due to global warming (Ferrer-Paris et al., 2023). Changes in ice volume or mass balance are direct indicators of change in the substrate, although detailed measurements are limited to relatively few glaciers in the world. Climatic conditions suitable for ice accumulation are a less direct indicator, but change can be calculated from global and regional datasets assuming they will correlate with mass balance. Change in ice mass is the most informative and direct indicator and should be used in preference to redundant indirect climate indicators for well sampled glacier types. However, the indirect climatic indicators may be used for glacier types without detailed ice mass data.

Principle	*Example*
6. Interactions between two or more pathways of degradation may be represented through robust, expert-based rules to define thresholds of ecosystem collapse based on a simple combination of ecosystem variables. Rule-based methods require assessors to explicitly state how and why the variables are combined.	See Section 7.3.2 Multiple indicator variables Collapse of an ecosystem type may be defined by a requirement for two variables to have crossed their respective thresholds. Alternative pathways of collapse may be defined as occurring when an indicator (e.g. water salinity) either exceeds an upper threshold or falls below a lower threshold value.
7. Estimates of the relative severity and extent of ecosystem degradation can be based on spatial data, inferences or expert-derived estimates.	See Section 3.3.3 Standards of evidence and dealing with uncertainty.

Abiotic indicators

Criterion C requires suitable indicators that represent abiotic properties (components or processes), which vary greatly between ecosystem types and their respective threatening processes. Examples for a range of terrestrial, freshwater and marine ecosystems are given in Table 12.

Criterion C offers many opportunities to assess risks to ecosystems that stem from anthropogenic climate change (see Section 9.1 Climate change). Direct measures of climate change include temperature, precipitation, humidity and wind (Table 12). These may be assessed individually, or together in models of climatic suitability of an area for a given ecosystem type (see Section 7.3.2 Multiple indicator variables). Direct measures of climate change, however, may be very indirect indicators of ecosystem degradation. In many cases, it may be more appropriate to measure abiotic variables that represent proximal measures of environmental responses to climate change (e.g. ice melt, cloud diminution, Table 12). Assessors must justify their choice of indicators by describing the pathways through which climate change affects them and the flow-on effects on ecosystem dynamics, with transparent evaluation of the limitations of the indicator, and the resulting uncertainty in the risk category via plausible bounds. See Section 9.1 Climate change for further information on using projections from climate models.

Table 12. Examples of variables potentially suitable for assessing the severity of environmental degradation under criterion C.

Environmental degradation	*Variables*
Geophysical	
Desertification of rangelands	Proportional cover of bare ground, soil density, soil compaction indices, remote sensing indices of change (Zhao et al., 2005; Ludwig et al., 2007).
Sedimentation of streams, coral reefs	Sediment accumulation rates, sediment load of streams, discharge, turbidity of water column, frequency and intensity of sediment plume spectral signatures (Rogers, 1990).
Structural simplification of stream beds and marine benthos	Microrelief, rugosity abundance of benthic debris, trawling frequency and geophysical spatial heterogeneity pattern (Watling & Norse, 1998). Diversity of micro-terrain features (Cabezas et al., 2009).

Chemical	
Eutrophication of soils, freshwater streams or lakes	Levels of dissolved or soil nitrogen, phosphorus, cations, oxygen, turbidity, bioassay (Carpenter, 2003) or atmospheric influx.
Acid rain	Rainwater chemistry indicators of acid rain induced deforestation (Likens, 1992).
Salinisation of soils or wetlands	Field monitoring of salinity of soils, groundwater or surface water (Micklin & Aladin, 2008). Remote sensing of ground surface albedo (Metternicht & Zinck, 2003).
Ocean acidification (related to CO_2 emissions)	Aragonite concentration reduces growth rates of carbonate exoskeletons, shells and reefs, with implications for reef structures and bioerosion (Bland et al., 2017).
Hydrological	
Changed water regime or hydroperiod	In situ (field-based) monitoring of stream flow volume, tidal inundation, or piezometric water table depth (Mason et al. 2021). Remote sensing of surface water extent, frequency and depth of inundation (Mac Nally et al., 2011). Spatial variance in inundation depth and duration (Cabezas et al., 2009).
Drying or terrestrialisation of wetlands	Surface water extent or soil moisture indicates transformation of wetlands to terrestrial systems with associated change in biotic properties (Ferrer-Paris et al., 2019; Keith et al., 2023b).
Climatic	
Ambient climate conditions (precipitation, temperature)	Mean trends or projections of suitable climate may be used, so long as a relationship with ecosystem degradation is justified (Burns et al., 2015; Ferrer-Paris et al., 2019).
Sea surface temperature	Higher water temperatures cause thermal stress and mass bleaching of coral (Bland et al., 2017; Obura et al., 2021).
Frequency or severity of extreme climate events (e.g. heat waves, droughts, storms)	Observed or projected values may be used as indicators, so long as relationship with ecosystem degradation is justified, e.g. bleaching events (Hughes et al., 2017), hurricanes (Bland et al., 2017).
Changes in cloud incidence or level	Observed or projected cloud cover, cloud altitude may be used as indicators where they affect identified ecosystem properties through changes to water budgets, thermal stress or insolation (e.g. Pounds et al., 1999; Auld & Leishman, 2015).
Sea-level rise	Acoustic monitoring of sea level, extent of tidal inundation in floodplains, estuarine systems (Hannah & Bell, 2012; Sievers et al., 2020) and intertidal rocky shores (Schaefer et al., 2020).
Ice or snow melt	Remote sensing of sea ice extent (Hong & Shin, 2010), glacial ice mass (Ferrer-Paris & Keith, 2024), sea ice breakout date (Clark et al., 2015), snowpack depth or extent (Williams et al., 2015).

Biotic indicators

A broad set of variables are potentially useful for quantifying biotic processes and associated functional declines. This includes changes in species richness, composition and dominance; relative abundance of species functional types, guilds or alien species; measures of interaction diversity; changes in identity and frequency of species movements; measures of niche diversity and structural complexity (Table 13).

Criterion D offers options for assessing degradation resulting from fragmentation via a range of direct indicators of response and indirect indicators of changing spatial configuration (Table 13). A key consideration is to assess the fragmentation process, rather than static patch statistics as symptoms of risk. This approach enables the RLE assessment to distinguish between ecosystems at risk of collapse due to fragmentation from those that exhibit long-term dynamic equilibrium across a relatively stable configuration of small and/or isolated patches, as may be expected, for example, in naturally isolated inselberg and lake systems that may have persisted over geological time scales. Section 9.2 Fragmentation provides guidance on options for indicators and analysis based on the expression of expected fragmentation symptoms.

Table 13. Examples of biotic variables potentially suitable for assessing the severity of disruption to biotic interactions under criterion D.

Indicator variable	*Role in ecosystem resilience and function*	*Example*
Species richness (the number of species within a taxonomic group per unit area).	Ecological processes decline at an accelerating rate with loss of species (Cardinale et al., 2011). Species richness is related indirectly to ecosystem function and resilience through its correlations with functional diversity, redundancy and complementarity.	Response of species diversity of grasses and relative abundance to varying levels of grazing in grassland (Walker et al., 1999).
Species composition and dominance.	Shifts in dominance and community structure are symptoms of change in ecosystem behaviour and identity.	Shift in diet of top predators (killer whales) due to overfishing effects on seals, caused decline of sea otters, reduced predation of kelp-feeding urchins, causing their populations to explode with consequent collapse of giant kelp, structural dominants of the benthos (Estes et al., 2009; Box 18).

Indicator variable	*Role in ecosystem resilience and function*	*Example*
Abundance of key species (ecosystem engineers, keystone predators and herbivores, dominant competitors, structural dominants, transformer invasive species).	Invasions of certain alien species may alter ecosystem behaviour and identity, and make habitat unsuitable for persistence of some native biota. Transformer alien species are distinguished from benign invasions that do not greatly influence ecosystem function and dynamics.	Invasion of crazy ants simplifies forest structure, reduces faunal diversity and native ecosystem engineers (Green et al., 2011). Invasion of arid Australian shrublands and grasslands by Buffel Grass makes them more fire prone and less favourable for persistence of native plant species (Clarke et al., 2005; Miller et al., 2010).
Functional diversity (number and evenness of types).	High diversity of species functional types (e.g. resource use types, disturbance response types) promotes coexistence through resource partitioning, niche diversification and mutualisms (Allen et al., 2005). Mechanisms similar to functional complementarity.	High diversity of plant-derived resources sustains composition, diversity and function of soil biota (Eisenhauer et al., 2011). Fire regimes promote coexistence of multiple plant functional types (Keith et al., 2007).
Functional redundancy (number of taxa per type; within- and cross-scale redundancy; see Allen et al., 2005).	Functionally equivalent minor species may substitute for loss or decline of dominants if many species perform similar functional roles (functional redundancy). Low species richness may be associated with low resilience and high risks to ecosystem function under environmental change (Allen et al., 2005; Walker et al., 1999).	Response of bird communities to varying levels of land use intensity (Fischer et al., 2007).
Functional complementarity (dissimilarity between types or species).	Functional complementarity between species (e.g. in resource use, body size, stature, trophic status, phenology) enhances coexistence through niche partitioning and maintenance of ecosystem processes (Cardinale et al., 2007).	High functional complementarity within both plant and pollinator assemblages promotes recruitment of more diverse plant communities (Fontaine et al., 2005).
Interaction diversity (interaction frequencies and dominance, properties of network matrices).	Interactions shape the organisation of ecosystems, mediate evolution and persistence of participating species and influence ecosystem-level functions, e.g. productivity (Thompson, 1997).	Overgrazing reduced diversity of pollination interactions (Vázquez & Simberloff, 2003). Pollinator network structure (e.g. nestedness, existence of a highly interdependent plant and pollinator core group) influences the likelihood of extinction cascades (Campbell et al., 2012; Lever et al., 2014).

Indicator variable	*Role in ecosystem resilience and function*	*Example*
Trophic diversity (number of trophic levels, interactions within levels, food web structure).	Compensatory effects of predation and resource competition maintain coexistence of inferior competitors and prey. Loss or reduction of some interactions (e.g. by overexploitation of top predators) may precipitate trophic cascades via competitive elimination or overabundance of generalist predators.	Diverse carnivore assemblages (i.e. varied behaviour traits and densities) promote coexistence of plant species (Calcagno et al., 2011), decline of primary prey precipitates diet shifts and phase shifts (Springer et al., 2003).
Spatial flux of organisms (rate, timing, frequency and duration of species movements between ecosystems).	Spatial exchanges among local systems in heterogeneous landscapes provide spatial insurance for ecosystem function (Loreau et al., 2003). Exchanges may involve resources, genes or involvement in processes (Lundberg & Moberg, 2003).	Herbivorous fish and invertebrates migrate into reefs from seagrass beds and mangroves, reducing algal abundance on reefs and maintaining suitable substrates for larval establishment of corals after disturbance (Moberg & Folke, 1999).
Structural complexity (e.g. complexity indices, number and cover of vertical strata in forests, reefs, remote sensing indices).	Simplified architecture reduces niche diversity, providing suitable habitats for fewer species, greater exposure to predators or greater competition for resources (due to reduced partitioning).	Structurally complex coral reefs support greater fish diversity (Arias-González et al., 2012). Structurally complex woodlands support greater bird diversity (Huth & Possingham, 2011).
Patch configuration.	Changes in various metrics of patch configuration may indicate symptoms of ecosystem fragmentation and associated edge effects and limitations on rescue and recolonisation processes.	Changes in patch configuration metrics indicate functional decline of Fynbos ecosystem types undergoing fragmentation from expansion of urban and agricultural land uses (Ntshanga et al., 2021).

7.3.2. Multiple indicator variables

General indices of ecosystem condition or integrity that may combine several abiotic and biotic proxy variables are generally not valid indicators for application in Red List assessments because such indices fail to address the ecosystem-specific mechanisms and processes of functional decline. A single informative, sensitive and direct indicator is usually the simplest option for assessment (Table 14). Where data exist for multiple correlated variables only the most direct, sensitive variable with the most complete coverage should be assessed (i.e. redundant variables excluded). Alternatively, where data are available for independent variables that represent **different** degradation processes or **different** symptoms of the same degradation process, these indicators may be assessed individually or in combination against the criteria, subject to justification of assumptions that underpin the method of combination (Principle 5, Table 11).

Aggregation of two or more variables into composite indices (e.g. additive scores) may be an option to assess combined effects. However, there are significant limitations of such indices. First, combined indices can introduce compensatory or averaging effects, or additional sources of noise, resulting in reduced sensitivity to detect declines, or worse, erroneous trends. Second, aggregation relies on ecological and statistical assumptions about interactions between different variables that are rarely tested and compound uncertainties, especially in data-poor ecosystems. Assessors should therefore avoid aggregating variables when they are uncertain about ecosystem dynamics and the assumptions underpinning the aggregation, and instead use the most important component variables as individual indicators (Principle 5, Table 11). The most important variables are those that directly represent ecosystem processes and have high sensitivity to change in ecosystem properties.

In some cases, however, it may be challenging to disentangle dependencies and the effects of different processes, and a single variable or several independent variables may not produce a sufficient indicator of degradation. If multiple suitable indicators are available for assessment of functional symptoms of collapse, and ecosystem processes are well described and understood, the assessors should follow the options in Table 14 to incorporate this information into the assessment. The approach depends on whether available data represent the same or different degradation processes and the same or different symptoms of ecosystem response. Thus, four scenarios of relationships between indicators should be considered (Table 14):

1. **Redundancy.** If it is certain that several variables represent similar symptoms of the same degradation process, the most direct and informative variable with the most complete dataset should be selected as the indicator and the others discarded, as they most likely provide redundant information about the transition towards ecosystem collapse. If there are concerns about the accuracy of the most direct indicator or incomplete spatial or temporal coverage, a less direct variable may be used. If redundancy is uncertain, multiple variables can be assessed independently. For an example see Principle 5 in Table 11.

2. **Complementarity.** If several variables measure different symptoms of the same degradation process, they may provide complementary information about the transition towards ecosystem collapse. Different analytical approaches apply, depending on the relative contributions of the symptoms and associated data.

3. **Independence.** If the variables represent different degradation processes and the threats are completely independent, they are likely to provide independent evidence about different pathways towards collapse, and should be assessed separately. Overall status is determined by the indicator that produces the most severe risk.

4. **Interactions (interdependence).** If the state variables represent different degradation processes that reinforce or mitigate each other, the outcome is likely to depend on combinations of their values. Where they are well understood, simple decision rules or correlative habitat suitability models may be implemented to account for these interactive effects on the risk of collapse. They may also be addressed in the design of simulation models of ecosystem dynamics (see Section 8 Criterion E. Quantitative risk analysis). However, simple indices are generally not capable of representing such interactions faithfully.

Ecosystems can exhibit multiple symptoms of degradation in response to one threatening process, or they may be exposed to multiple threatening processes that cause different symptoms. Independent assessment as separate indicators cannot account for interactions that involve synergistic or compensatory effects or other dependencies. For example, in alpine bogs, global warming reduces the water balance required for peat formation and also increases the risk of peat-consuming fires (Regan et al., 2020). In other cases, one threat can erode the resilience of an ecosystem and make it more sensitive to further degradation or collapse by other processes (Keith et al., 2023b).

Dependencies between two or more degradation variables, where they are reliably understood, may be addressed with rule-based approaches that faithfully and explicitly represent the dependencies. Bidirectional pathways of degradation in relation to a single scalar indicator may similarly be addressed with rule-based approaches. Where use of combined or bidirectional indices is justified, the assessment should be supported by a documented rationale for aggregation and a critical evaluation of ecological and mathematical assumptions underlying the method of aggregation (Principle 6, Table 11).

Combinatorial and sequential rules

Interactions among indicators can be assessed by designing and applying explicit rules that represent the relationships between indicators, where these are known with high confidence. For example, a combinatorial rule to assess ecosystem degradation via two interdependent threats may stipulate that ecosystem collapse occurs only when thresholds of two different indicators are both exceeded. A second example rule may require two bidirectional thresholds to be assessed in one indicator (i.e. when the ecosystem state exceeds either of maximum or minimum threshold values). Design of rule structures should reflect known inter-dependencies of ecosystem dynamics. To justify use of rule-based indicator combinations, assessors must explicitly describe the underlying interactions or dependencies, the evidence on which they are based, how the rules implement the relationships and evaluate the efficacy of any assumptions involved in the combined indicator. Diagrammatic state-transition models of ecosystem dynamics, or classification methods based on decision trees may be helpful tools to support this justification.

Sequential rules to combine multiple indicators may be more suitable than combinatorial rules to address other types of interactions or dependencies. For example, the most fundamental indicator may be assessed first, and the outcome adjusted based on assessments of other indicators (Box 15). This approach also requires explicit description of the rationale and rules used to ensure the process is transparent and reproducible.

While combinatorial or sequential rules offer an opportunity to design indicators to represent well-understood interactions between different degradation processes, they should be avoided where interactions are not known with reasonable certainty. This is because overly idiosyncratic or unnecessarily complex indicators may limit the comparability of Red List assessments across different ecosystem types.

Table 14. Summary of options for indicator design (see text for examples).

Number of candidate indicators available	*Relationship between candidate indicators*	*Options for indicator selection*	*Indicator combination*	*Relative severity (RS) and extent of degradation (ED) thresholds*	*Assess status*	*Examples*
None					Data Deficient	
One		Not necessary	All based on single indicator			
More than one for same threatening process	Redundancy (i.e. alternative variables measure same symptom)	Focus on single most informative variable	All based on single indicator			
	Complementary (i.e. alternative state variables measure different symptoms)	Consider each equally informative	Assess each indicator independently (do not combine)	RS and ED values for each variable	Use the indicator producing the greatest rate of decline, or all indicators contribute to a bounded estimate of decline.	Ghoraba et al. (2019)
		Ranked by known or expected contribution or importance	Assess each variable independently (do not combine)	RS and ED values for each variable	Rules to sequentially combine results from each variable into bounded estimate.	Obura et al. (2022), see example in Box 15
		Unranked, but known contributions	Calculate derived or combined index if assumptions justifiable	Calculate RS/ED values based on derived index	Based on derived index, consider uncertainty in index or thresholds.	
		Unranked, unknown contributions	Use statistical methods (correlative models, multivariate analysis)	Calculate RS/ED values based on derived index	Based on derived index, consider uncertainty in method, index or thresholds.	Environmental suitability (Ferrer-Paris et al., 2019)

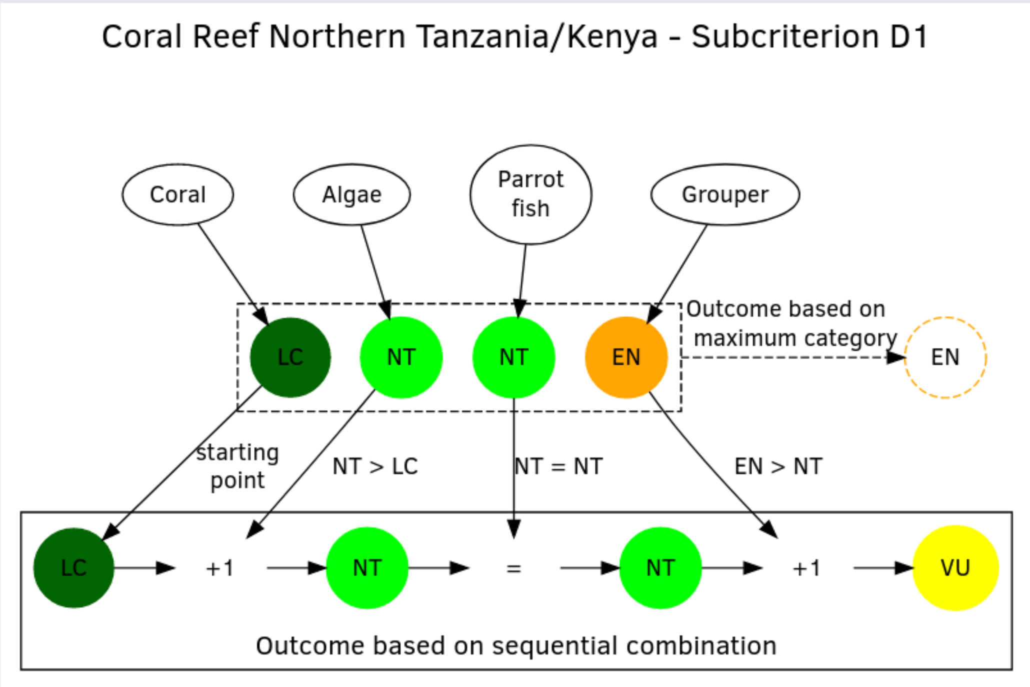
Number of candidate indicators available	*Relationship between candidate indicators*	*Options for indicator selection*	*Indicator combination*	*Relative severity (RS) and extent of degradation (ED) thresholds*	*Assess status*	*Examples*
More than one for multiple processes	Independent	Consider each equally informative	No, each variable considered independently	RS and ED values for each indicator	Use one indicator producing the greatest rate of decline, or all indicators contribute to a bounded estimate.	
	Interdependent	Consider interaction between variables	Consider indicators independently and additional variables representing combined effects	RS and ED values for each variable and their combined effect	Compare outcomes for each indicator independently, and for their interaction. Consider the plausibility of each outcome.	Regan et al. (2020): climate and fire effects on bogs

Box 15. Combinatorial rules for multiple indicators. Source: Obura et al., 2021.

An assessment of **coral reef ecosystems of the Western Indian Ocean** defined four interacting biotic compartments of the ecosystem (Obura et al., 2021): (i) hard corals as the engineers of coral reef ecosystems, (ii) fleshy algae, as the principal competitor of coral, (iii) parrotfish which have strong mediating effects and dependence on algae and corals, and (iv) groupers which as piscivores impose functional and trophic controls on multiple aspects of reef ecology. All are informative indicators of reef function responsive to different pressures, but also exhibiting different levels of spatial and temporal variability.

Assessing the indicators independently and assigning overall status of criterion D on the basis of the highest category does not consider the interactions and dependencies among the ecosystem compartments and may also make the outcome unduly sensitive to extreme values that result from inherent patchiness or temporal variability. To address this, Obura et al. (2021) designed and applied a simple sequential rule in which an initial risk category based on coral cover (the foundational compartment of the ecosystem) was adjusted upwards by one risk category if successive assessments of algal, herbivore and piscivore compartments produced a higher category than the preceding baseline. Alternatively, no adjustment was made if a compartmental assessment produced the same or lower status than the preceding baseline. For example, a baseline assessment of Least Concern (LC) (based on coral cover alone) could be adjusted to Near Threatened (NT) (with algal assessment), retained at Near Threatened (NT) based on assessment of herbivorous fish, and adjusted from Near Threatened (NT) to Vulnerable (VU) based on assessment of piscivorous fish, resulting in a final outcome of Vulnerable (VU) for criterion D1 (see figure).

Obura et al. (2021) compared three different methods to combine biotic indicators for different ecosystem compartments to understand the contribution of the sources of uncertainty to the final assessment outcome. They found that outcomes were sensitive to extremes when assessments were based on the maximum category, while the sequential rule reduced this bias and showed reasonable concordance in less extreme cases.



Correlative habitat suitability models

Correlative models of 'habitat' suitability offer an alternative to combine multiple input variables into an indirect indicator of environmental degradation for assessment under criterion C when there is less certainty about the mechanisms driving ecosystem collapse. These models fit occurrence records to environmental predictors to estimate habitat suitability in an area of interest, analogous to species distribution models (Elith et al., 2006). An advantage of such models, if they incorporate bioclimatic predictors, is that they may be used to project habitat suitability under future climates for assessment under subcriterion C2. However, their projections are highly sensitive to model structure, variable selection and niche occupancy.

When using habitat suitability models assessors should ensure that their predictor variables include those that represent degradation processes, as well as relevant covariables, and that the models address the following principles:

1. The input data is representative of environmental variation across the modelled area (Elith et al., 2006).
2. Selection of environmental predictors is based on sufficient knowledge to ensure that they represent ecological processes that mediate ecosystem occurrence and dynamics (Keith et al., 2014).
3. The model is constructed using modelling algorithms likely to produce high-performance outputs and the data meet their assumptions (Elith et al., 2006; Valavi et al., 2022).
4. Predictive performance is assessed and reported using appropriate measures that address different aspects of model performance appropriate to the model being applied (Valavi et al., 2022).
5. Responses to predictor variables are cross-checked between different algorithms (Elith et al., 2006).
6. Modelled responses are evaluated with regard to expected mechanistic responses (Keith et al., 2014).
7. Extrapolation risks are evaluated by quantifying the deviation between future and current environmental domains of the study area and by assessing the temporal constancy of correlations between variables (Keith et al., 2014).
8. Spatial consistency of the model is verified by a geographically stratified cross-validation or similar approach (Keith et al., 2014).
9. The robustness of predicted climate change impacts is incorporated into Red List assessments by quantifying uncertainty in both the climate and ecological models (Ferrer-Paris et al., 2019).

Importantly, correlative models assume that the presence and absence records of the ecosystem type sample its distribution at a time when it is in equilibrium with the environment, i.e. that it is not still expanding into its potential niche or persisting at sites that are no longer suitable. Therefore, in addition to requirements 1–9, assessors must justify use of correlative models with transparent ecological reasoning about the likelihood that some part of the suitable environmental niche space is currently unoccupied or that the ecosystem type currently occupies environmentally unsuitable sites due to lagged responses to environmental change. These assumptions and potential ecological lags in the equilibration of ecosystem distribution with habitat suitability (i.e. via movement of biota) limit the application of habitat suitability models in the assessment of criterion A, although they may be appropriate for such application in certain ecosystem types (e.g. some aquatic ecosystems) that are expected to adjust their distributions rapidly under changing environmental conditions.

Example applications of correlative models include future projections of suitable habitat of upland mire ecosystems in southeastern Australia (Keith et al., 2014), forests in the Americas (Ferrer-Paris et al., 2019), and tropical ecosystems in Myanmar (Murray et al., 2020). Further guidance on use of correlative models may be found in Section 9.1 Climate change and in the *Guidelines for Using the IUCN Red List Categories and Criteria* (IUCN SPC, 2024).

Aggregated indices of ecosystem integrity or condition

A third option for dealing with multiple indicator variables is to incorporate them into a single measure (often labelled an integrity index or condition index). Typically, these are derived either by constructing a mathematical index (e.g. additive or multiplicative), or by applying multivariate analyses (e.g. principal components analysis) to fit a new summary variable correlated with the inputs (e.g. an eigen vector). As noted above and in Table 11 (Principle 5), such aggregated indices are not recommended for use in Red List assessments due to multiple limitations, including difficulties in selecting and weighting input variables, the potential for unknown interactions such as compensatory effects among input variables, and reliance on assumptions that are difficult or impossible to evaluate across the environmental and spatial domain to which the index is applied. In many aggregated indices, relationships between the input variables are submerged, poorly understood, misrepresented by the method of aggregation, and thus have unintended effects on index values. In some cases, the input variables themselves are indirect indicators of pressure, limiting the sensitivity of the index to detect ecosystem response to the pressures. Even when no other data is available, assessors should avoid the use of aggregated indices that are not specific to the ecosystem under assessment. Applying generic indices across functionally contrasting ecosystems is unlikely to assess degradation accurately if key processes are not well understood or if they differ among ecosystem types (especially problematic in systematic assessments).

In some cases, where relationships among input variables and a degradation process are understood, it may be possible to design a viable index that represents a specific degradation process in a particular ecosystem type or group of functionally similar ecosystem types. For example, Ferrer-Paris et al. (2019) assessed forest degradation through defaunation via three spatial variables representing accessibility, demand and protection combined to estimate response ratios of the abundance of mammals in hunted and unhunted sites (Benítez-López et al., 2019). The output was in turn transformed to estimate potential declines across the distribution of ecosystem types. This kind of approach relies on multiple assumptions and simplifications, and might be uninformative for on-ground conservation efforts, where the estimated severity and extent of degradation are not sensitive to direct interventions.

7.3.3. Indicator analysis

For each indicator variable, assessors must:

1. Estimate the value of the initial state of the indicator variable (at the beginning of the assessment time frame).
2. Estimate the expected value in a collapsed state.
3. Measure or estimate the present or future value of the variable (i.e. at the end of the assessment time frame).
4. Calculate the relative severity for each observation or sample unit (where there is more than one).
5. Estimate the extent of the degradation by either direct inference, aggregation of observations, or by calculating proportional extent of degradation for a set of thresholds (see Section 7.3.5 Calculating extent of degradation).

Relative severity (step 4) is calculated using the unbounded or bounded formula, or other approaches relevant for the available data (see Section 7.3.4 Calculating relative severity). Note that the calculated relative severity can be negative if the ecosystem is recovering from degradation. Assessors may either estimate the extent of degradation that exceeds a threshold level of severity or estimate the average severity of degradation across the entire ecosystem distribution (100% of extent; Figure 14). Box 16 provides a simple example of this latter scenario.

The approach to estimating extent of degradation (step 5) will depend on the characteristics of the underlying indicator data (see Section 7.3.5 Calculating extent of degradation). If relative severity is calculated for a single unit or a small representative sample, inference can be based on expert opinion with appropriate structured elicitation methods and consideration of uncertainties (see Section 3.3.4 Quantitative data and expert knowledge and Section 4.4.1 Dealing with uncertainty). For values of relative severity calculated from a representative sample, assessment can be based on a single average relative severity value (relative severity formula in Section 7.3.4 Calculating relative severity). For spatially explicit or large heterogenous samples, the proportional extent of degradation can be calculated for each severity threshold (extent of degradation formula in Section 7.3.5 Calculating extent of degradation).

Assessors then determine the risk category by comparing their estimates of relative severity and extent of degradation to the respective thresholds specified in Section 7.2 Thresholds and subcriteria.

Box 16. Assessing environmental degradation (criterion C).
Source: Adapted from Appendix S2.7 in Keith et al., 2013.

Flooding is a key ecological process that sustains the **Gonakier Forests** for the Senegal River Floodplain in Senegal-Mauritania (Keith et al., 2013). As floods occur only during the wet season months, the maximum annual river height was assumed to be indicative of the river's capacity to flood each year. River height data were available for 100 years from 1904 to 2003. To assess criterion C, mean annual maximum river height across four gauging stations was used as a proxy for environmental degradation. River flows declined sharply, reaching a minimum during the late 1970s and 1980s. Floods of 2,500 m^3/s, which are needed for floodplain inundation, would be very unlikely to occur based on river flows observed during 1986–1989. Extreme rates of tree mortality were observed between the mid-1970s and the mid-1980s, corresponding to the lowest maximum river heights (473±27 cm) observed during the 100 years of records.

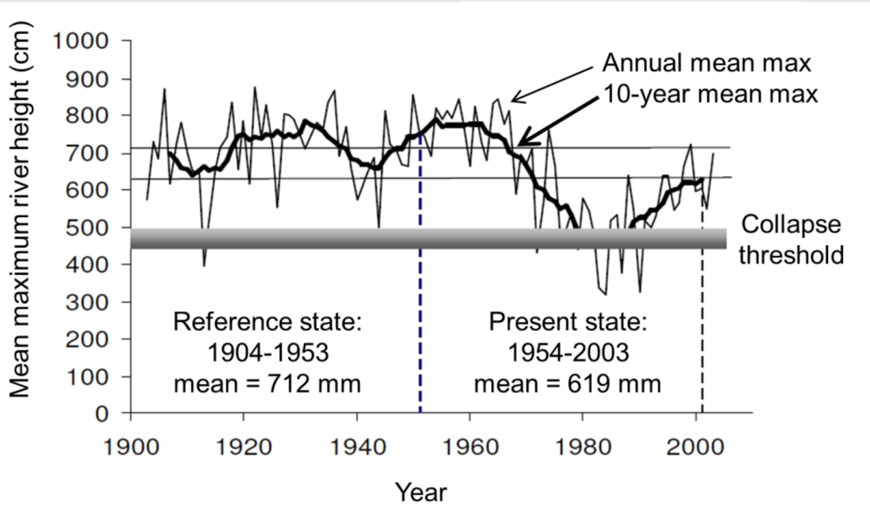
Based on these observations, the collapse threshold was defined as the mean maximum river height for a 50-year period falling below 450–500 cm, causing extensive tree mortality. To calculate the relative severity of hydrological decline, the time series was divided into the reference period (1904–1953) and the present period (1954–2003). Since the collapse threshold is an interval, relative severity was estimated for the lower and upper bounds of the interval.

For the lower bound (450 mm), relative severity is:

$$100 \times (\text{Observed decline}) / (\text{Maximum decline}) = (712 - 619) / (712 - 450) \times 100 = 35\%$$

For the upper bound (500 mm), relative severity is:

$$100 \times (\text{Observed decline} / \text{Maximum decline}) = (712 - 619) / (712 - 500) \times 100 = 44\%$$



River height in the Gonakier forest.

Since hydrological decline affects the entire ecosystem, it was assumed that the extent of the threat was > 80%, thus leading to the conclusion that the ecosystem is Vulnerable according to criterion C1 (degradation with relative severity ≥ 30% over an extent ≥ 80% in the last 50 years).

7.3.4. Calculating relative severity

Relative severity is an essential concept for comparing risks among ecosystems undergoing different types of degradation. Relative severity describes the proportional change observed in an environmental or biotic state variable scaled between two values: one describing the initial state of the system (0%), and one describing a collapsed state (100%). Thus, if an ecosystem type undergoes degradation with a relative severity of 50% over an assessment time frame, this implies that it has transformed halfway to a collapsed state.

Information on relative severity is combined with information on the proportion of the ecosystem affected (extent) to determine the risk category under criteria C and D.

Formulas for relative severity

There are several valid approaches to estimate, infer or project the values of relative severity, depending on the mechanism and pathway of degradation and the nature of available indicator data. In the simplest case, relative severity may be calculated by range-standardising the raw values of the state variable between its initial value and its collapse value (Keith et al., 2013), but a bounded formula is recommended when these values need to be aggregated spatially (Ferrer-Paris & Keith, 2024). If initial and final values come from two independent samples (i.e. temporal samples were not measured in the same sites), resampling can be used to estimate the expected value of relative severity and its variance (Obura et al., 2021). For highly fluctuating indicators, a frequency-based formula might be more appropriate (Appendix S2.18 in Keith et al., 2013). These options are described below.

Unbounded range standardisation formula. The following equations rescale an indicator variable to a proportional change towards collapse suitable for assessing functional criteria:

Relative severity (%) = (Observed or predicted decline / Maximum decline) × 100

The shape of observed, predicted and maximum decline will depend on the direction of decline. If the threshold of collapse indicates a minimum viable value (e.g. water levels, cover of a key vegetation component), use:

Observed or predicted decline = Initial value - Present or future value

and

Maximum decline = Initial value - Collapse value

If the threshold of collapse indicates a maximum tolerable value (e.g. concentration of pollutants or abundance of invasive species), use:

Observed or predicted decline = Present or future value - Initial value

and

Maximum decline = Collapse value - Initial value

Example calculation: The table shows estimates of mean standing stock of juvenile finfish in three regions of the Indian Sundarbans (Sievers et al., 2020).

	Site A	*Site B*	*Site C*
Initial value	219.5	667.9	790.3
Present value	149.8	320.9	338.2
Observed decline	69.7	347.0	452.1
Maximum decline	219.5	667.9	790.3
Relative severity	31.8 %	51.9%	57.2 %

If the threshold of collapse is zero, the observed decline for site A is calculated as 219.5 - 149.8 = 69.7 and maximum decline as 219.5 - 0 = 219.5. Thus, the relative severity for site A is:

$$RS = 100 * \frac{69.7}{219.5} = 31.8\%$$

Extreme values and bounded formula. The original formula for relative severity has a maximum value of 100% if it is applied to physical variables with absolute threshold values or relative variables with extreme thresholds (e.g. ice mass of 0 kg, vegetation cover reaches 0%), but it can have values larger than 100% if final values are more extreme than the chosen thresholds. The values of relative severity can become negative if final values are further away from collapse than the initial values (improved conditions).

These extreme cases might cause unexpected results when aggregating relative severity values over multiple units (mean of several samples, total of multiple occurrences, etc.).

Assessors can use a conditional formula (Ferrer-Paris & Keith, 2024) with three rules:

1. $RS = 0\%$, if $OD < 0$
2. $RS = 100\%$, if $OD \geq MD$
3. $RS = 100 * \frac{OD}{MD}$, otherwise.

Where *OD* is the observed or predicted decline, and *MD* is the maximum decline as described above.

Resampling estimate of relative severity. When initial and final values do not come from paired samples, it is possible to use bootstrap or jackknife procedures to estimate the sampling distribution of relative severity based on the previous formulas. Specific procedures for the relative severity formula have not been formally tested and optimised, but general principles can be applied (Varian, 2005).

For example, in the assessment of the coral reefs of the Western Indian Ocean, the current value of coral and algae cover and fish abundance for each site was available for the period 2013–2019, with 9 to 113 sites per ecoregion. The initial values for each ecoregion were calculated as mean estimates with a standard deviation based on different sources with variable sample size. A bootstrap approach was used to combine fixed current values per site with randomised initial values. For each variable and each ecoregion, a single iteration consisted of randomly extracting an initial value from a normal distribution defined by the ecoregion initial mean and standard deviation. The randomised initial value was used to calculate observed and maximum decline for each site, and then calculate relative severity as the ratio of observed and maximum decline. The calculations were repeated multiple times and a sensibility analysis suggested that results stabilised before 500 iterations, and a bootstrap sample size of 750 iterations was used for the assessment (Obura et al., 2021).

Formula based on frequency of collapse. The previous formulas are not adequate in some special circumstances, for example:

- If the data is categorical (occurrence of degraded vs. non-degraded states).
- If the data is a time series of highly fluctuating data (e.g. time series of concentration of pollutants) where measurements from a single observation or measurement site can be above or below a degradation threshold at different points in time.

To calculate relative severity based on frequency or prevalence of degradation, use the following formula:

$$RS = 100 * \frac{b}{n}$$

where *b* is the number of times that the observed state variable is classified as degraded, and *n* is the total number of observations (Appendix S2.18 in Keith et al., 2013). The degraded state can be recorded as a categorical variable or using a quantitative threshold (observed value > threshold value).

Example calculation: The table shows 17 measurements of the maximum phosphorus (PO_4-P) concentration (μg/L) of Lake Burullus, Egypt for the period from 1973 to 2017 (Ghoraba et al., 2019).

Year	*PO_4-P (μg/L)*	*> 50 μg/L*	*Year*	*PO_4-P (μg/L)*	*> 50 μg/L*	*Year*	*PO_4-P (μg/L)*	*> 50 (μg/)L*
1978	1.23	no	2001	2.70	no	2013	932.60	yes
1979	0.60	no	2003	297.30	yes	2014	591.06	yes
1985	1.33	no	2006	270.00	yes	2015	716.46	yes
1987	2.32	no	2010	375.75	yes	2016	514.00	yes
1997	2.90	no	2011	660.59	yes	2017	1872.00	yes
2000	3.40	no	2012	1117.12	yes			

If the threshold of degradation is 50 μg/L, we count *b* = 10 observations above the threshold, out of n = 17 measurements. The value of relative severity is:

$$RS = 100 * \frac{10}{17} = 58.8\%$$

7.3.5. Calculating extent of degradation

Estimating the extent of functional decline (abiotic degradation or biotic disruption) can be based on spatial data, expert-derived estimates or inferences, depending on the available data and context of the assessment.

The extent of degradation is compared to the ‘full extent’ of the ecosystem type or assessment unit to obtain a percentage value for assessment against the thresholds. Estimation should take into account if the ecosystem distribution has been reduced while the remaining area is undergoing degradation, i.e. areas where the ecosystem type has been extirpated should be excluded from the estimated extent of degradation to avoid double counting. To resolve any ambiguity there should be explicit definitions that distinguish the collapsed state from various states and degrees of degradation.

Options for estimating the extent of degradation vary depending on available spatial data, as well as whether the available estimates of relative severity are calculated from one or few samples:

- If relative severity is calculated for multiple states of degradation (either from spatial data or from multiple sampled sites), then assessors need to estimate the approximate area in each state of degradation.
- If the estimated relative severity of degradation represents a maximum value, then assessors need to estimate the approximate area subject to this maximum value.
- If the value of relative severity has been calculated for the whole assessment unit (value represents the total for the whole ecosystem extent or a representative mean value) the extent of degradation is considered to be 100%.

Where spatial data are available, the extent of degradation can be estimated for different levels of relative severity. This enables a severity-extent frontier to be plotted against the risk category thresholds (Figure 15), with the status for criterion C or D, respectively, determined by the maximum risk category intersected by the frontier. For example, Bland et al. (2017) modelled three biotic indicators of the Meso-American Coral Reef ecosystem under alternative future climates that generated different levels of coral bleaching to assess criterion D2, showing that the status of the ecosystem varied from Near Threatened (NT) to Critically Endangered (CR) for all three indicators, depending on the climate change scenario.

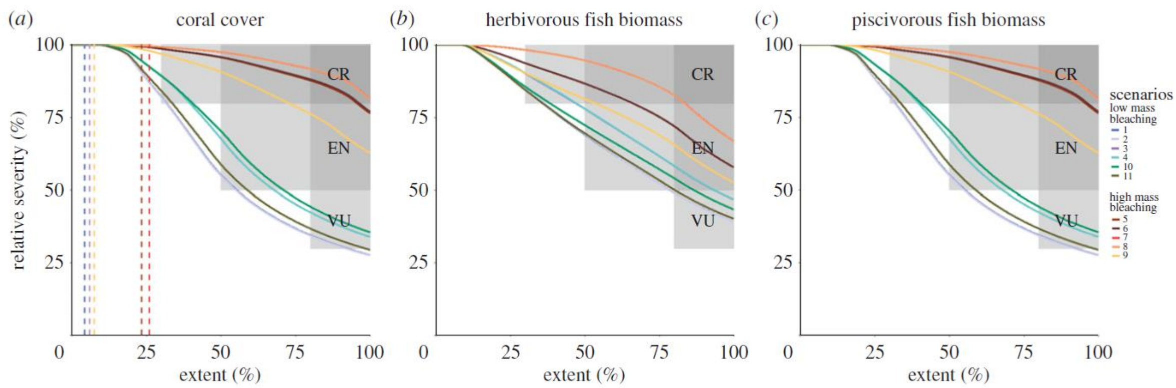


Figure 15. Relative severity and extent of degradation frontiers plotted against risk category thresholds for criterion D2 under future climate scenarios that generate different levels of coral bleaching. Source: Bland et al., 2017.

If multiple estimates of relative severity are available from different locations within the distribution of the ecosystem type, then a similar approach may be followed to that above, using spatial data or elicited expert opinion on the proportional extent of degradation represented by each sampled location. Box 17 offers further guidance on the calculations.

In the absence of sufficient spatial data to calculate a severity-extent frontier, assessors may either: estimate the extent of degradation at a level of relative severity that is believed to result in the highest risk category, or estimate the relative severity of degradation on average throughout the entire extent of the ecosystem type, i.e. for extent of degradation = 100% (see Box 16 for an example).

In some cases, the distinction between collapsed and extant states may be ambiguous and it may be appropriate to frame an integrated assessment of degradation under criterion C or D. For example, the expected future degradation in extent of snow beds in Norway is estimated to be almost 30% due to the increase in tree-line altitudes eroding snow bed area upslope, which is partially compensated by increased snow melt in higher elevations (Box 10). However, warming temperature will also trigger changes in biotic composition in the remaining snow bed areas. The complete extent of this additional decline is more difficult to estimate with current data and projections, but if the potential area loss is considered as part of the biotic change, then the extent of degradation will necessarily be higher than 30%.

Box 17. Aggregation of relative severity values from samples.

During assessment, the relative severity (RS) formula can be applied to many different sites representing sampling or observation units, and a summary value is calculated over the entire assessment unit to describe the mean level of degradation over the whole extent (Ferrer-Paris & Keith, 2024). Assuming that the assessment unit is comprised of n sites, and that RS_i represents the value of *RS* for site i, then, the average *RS* can be calculated as a weighted mean:

$$\underline{RS} = \sum_{i=1}^{n} RS_i \, w_i$$

Weights w_i refer to the relative contribution of each site i to the ecosystem extent or physical measure of interest, and $\sum_{i=1}^{n} w_i = 1$. If all units have equal weights, the average is equal to the arithmetic mean:

$$\underline{RS} = \frac{\sum_{i=1}^{n} RS_i}{n}$$

Example calculation: From the table with estimates of mean standing stock of juvenile finfish in three regions of the Indian Sundarbans (Sievers et al., 2020), we have three values of *RS* for three sites, if all sites have equal weights:

$$\underline{RS} = \frac{31.8\,\% \,+\, 51.9\,\% \,+\, 57.2\,\%}{3} = 46.97\,\%$$

However, site A has a much lower initial value than sites B and C, if we estimate weights of each site based on their initial values (for example 0.15, 0.40 and 0.45 respectively), we can recalculate as:

$$\underline{RS} = (31.8\,\% \times 0.15) + (51.9\,\% \times 0.40) + (57.2\,\% \times 0.45) = 51.27\,\%$$

Spatial variability in relative severity values

In most assessments, however, the distribution of RS_i values across the full spatial extent of the ecosystem type is of interest for expressing different levels of risk or pathways towards collapse, for example: high levels of degradation in a small part of their distribution, moderate levels of degradation in most of their distribution, or high levels of degradation in most of their distribution. The weighted cumulative extent of degradation (or proportional extent of degradation, *ED*) is the sum of weights for all sites above a threshold x:

$$ED(x) = 100 \times \sum_{i=1}^{n} \{w_i \times (1 \; if \; RS_i \geq x, or \; 0 \; if \; RS_i < x)\}$$

for w_i and RS_i as defined above (Ferrer-Paris & Keith, 2024). For assessment of functional criteria, the value of *ED* is calculated for a discrete set of thresholds relevant to the selected time frame.

Example calculation: From the example above with three sites with weights of 0.15, 0.40 and 0.45 respectively, the extent of degradation for a threshold of 50% is

$$ED(50\%) = 100 \times ((0.15 \times 0) + (0.40 \times 1) + (0.45 \times 1)) = 85\,\%$$

7.3.6. Assumptions

Determining an initial and a collapsed value for the indicator variable relies on assumptions about collapsed states of the ecosystem type. Such uncertainty in the point of collapse can be represented with bounded thresholds of the values of the variable. Relative severity can be calculated for both bounds and a best estimate, providing a lower and upper estimate for the risk category (Box 18). Similarly, uncertainty in the extent of degradation can be assessed with the use of upper and lower estimates. The use of bounded values yields an estimate of the extent and severity of degradation while clearly expressing uncertainty.

Similar to the declines of extent required for assessing under criterion A, the application of criteria C and D assume a functional form of decline. The simplest case illustrated above applies when there is a linear relationship between the assessment variable and the trajectory towards a collapsed state. Other scenarios are possible, for example, where collapse proceeds more slowly or more rapidly than indicated by changes in the assessment variable. In such cases a suitable transformation of the assessment variable should be used in the calculation of relative severity (Figure 16).

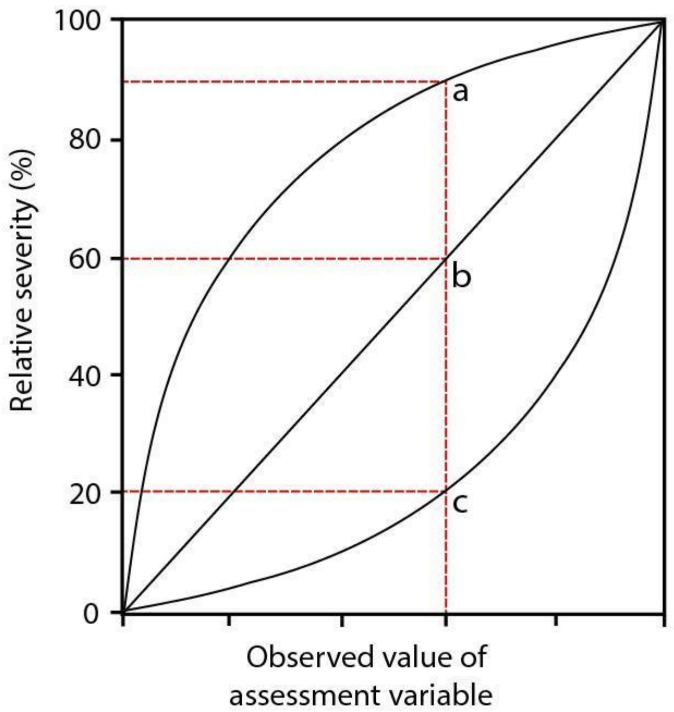


Figure 16. An observed value for a variable assessing degradation can be mapped to different values of relative severity depending on the functional form considered.
Source: Authors.
The red line indicates an observed value which can be mapped to a relative severity of 20%, 60%, or 90% depending on the functional form. This corresponds to a risk category of LC, EN or CR if the degradation occurs over ≥ 80% of the ecosystem type.

Determining whether the degradation is constant, accelerating or decelerating can be informed by time-series data. Assessors should evaluate whether the available data are sufficiently representative to characterise the shape of the decline in the indicator variable, ideally through appropriate statistical methods (Di Fonzo et al., 2013; Connors et al., 2014). Where time-series data are unavailable, it may be possible to infer changes in degradation using expert elicitation or space-for-time substitution with appropriate reference sites (Pickett, 1989). To overcome uncertainty due to this assumption, sensitivity analyses that include estimates produced from multiple shapes of decline can provide a bounded estimate for the risk assessment outcome.

7.3.7. Documentation

Assessors should document: (i) the selection of the indicator(s) with respect to the conceptual model of ecosystem dynamics; (ii) the justification of the bounded collapse threshold for the indicator; (iii) the calculation of relative severity; (iv) the estimation of the extent of degradation; (v) assumptions and appropriate sensitivity analyses (e.g. regarding the collapse definition or shape of decline); (vi) the final risk categories and plausible bounds. Temporal variation in ecosystem degradation is best shown in a graph that depicts changes in the variable over time, and includes any interpolation or extrapolation to match the relevant time frame (Box 18).

Box 18: Assessing disruption of biotic processes (criterion D).

Alaskan Giant Kelp Forests are structurally and functionally diverse assemblages, characterised by species of brown algae in the Order Laminariales. These create complex and dynamic layered forest architecture up to 15 m tall that provides substrate, shelter and foraging resources for a diverse fauna assemblage of epibenthic invertebrate herbivores and pelagic vertebrate predators.

The most serious disruption to biotic interactions occurs through trophic cascades involving sea otters, their predators (killer whales) and their prey (urchins, which consume kelp). Given that densities of kelp are inversely related to densities of urchins, and that phase shifts between forests and urchin barrens are related to a threshold abundance of otters (Estes et al., 2010), any of these variables is potentially suitable for assessing criterion D. Although data are available on population changes in great whales and pinnipeds (alternative prey for killer whales), these were not used because: (i) data on more proximal response variables are available; (ii) the causal relationship linking great whales and pinnipeds with otter abundance via killer whale predation is less certain than the link between otters, urchins and kelp.

Survey data for kelp stipe densities were available between 1987 and 2000 from seven islands (Estes et al., 2009). It was assumed that the seven islands, scattered across the Aleutian chain, were representative of the full distribution of the ecosystem. Ecosystem collapse occurs when kelp density is close to zero across all sites, consistent with kelp replacement by urchin barrens throughout the distribution. Rates of change in kelp density were calculated for each island assuming an exponential model. A weighted average across all sites indicated that kelp densities declined on average by 49.2% between 1987 and 2000. Allowing for some decline prior to 1987 or after 2000 suggests that the decline in kelp density over the past 50 years was at least 50% across the full ecosystem extent.

Aerial survey data for sea otters were available for 55 islands along the Aleutian chain between 1959 and 2000 (Doroff et al., 2003). Ecosystem collapse occurs when otter populations reach zero across all sites. The total population was estimated to be 55,000–74,000 prior to decline in the mid-1980s. By 2000 there were a total of 3,924–13,580 animals based on extrapolation from the aerial survey (Doroff et al., 2003). The lower and upper bounds of otter population decline are:

$$100 \times (55000 - 13580) / 55000 = 75.3\%$$

$$\text{and } 100 \times (74000 - 3924) / 74000 = 94.7\%$$

Evidence from trends in kelp density and sea otter sightings suggest a decline in biotic function of 50–95% relative severity across 100% of the ecosystem extent. The upper bound of this range may overestimate the severity of decline because: (i) the surveys may have underestimated the population due to detectability issues (Doroff et al., 2003); (ii) the calculations assume that otter and kelp populations have not recovered since 2000, in spite of qualitative evidence for some recovery. The most likely status of the ecosystem under criterion D1 is Endangered (EN), although a status of

Critically Endangered (CR) is possible. No projections are currently available for any of the biotic variables. The status of the ecosystem is Data Deficient (DD) under criterion D2.

The otter population in 1750 was comparable or slightly larger than its peak in the mid-1980s (Doroff et al., 2003). Based on this assumption, the decline in otter populations throughout the distribution of the kelp forest was 75–95% since 1750. The status of the ecosystem type under criterion D3 is therefore Endangered (plausible range Endangered – Critically Endangered). Thus, the Alaskan Giant Kelp Forests ecosystem type is listed as Endangered (plausible range Endangered – Critically Endangered).

8. Criterion E. Quantitative risk analysis

8.1. Theory

Criterion E serves two purposes. First it can be used to list an ecosystem type by implementing models that integrate multiple mechanisms of decline and their interactions into the risk assessment (as described below). Second, it provides an anchor for risk assessment and an overarching framework for the other criteria, as its analogue does in Red List criteria for species. Criterion E specifies the level of risk that corresponds to each category of threat, by defining the probability of collapse and the specified time frame for Critically Endangered (CR), Endangered (EN) and Vulnerable (VU) ecosystem types.

8.2. Thresholds

An ecosystem may be listed under Criterion E if it meets the thresholds for the criterion, a quantitative analysis that estimates the probability of ecosystem collapse to be:

CR	≥ 50% within 50 years
EN	≥ 20% within 50 years
VU	≥ 10% within 100 years

8.3. Applying criterion E

8.3.1. Methods

The probability of ecosystem collapse can be estimated with stochastic simulation models incorporating key ecosystem processes (see Box 19 for an example). The models should:

1. Produce estimates of an ecosystem variable for which a threshold of collapse has been estimated.
2. Produce quantitative estimates of risk of ecosystem collapse over a 50–100-year time frame.
3. Incorporate stochasticity in key processes that determine ecosystem properties.
4. Be applied with scenarios that represent plausible future scenarios of ecosystem dynamics.

A wide range of models can be used to apply criterion E. We provide broad recommendations for the application of criterion E in the form of nine steps to ensure that models are based on sound assumptions, scientifically credible and transparent (Figure 17).

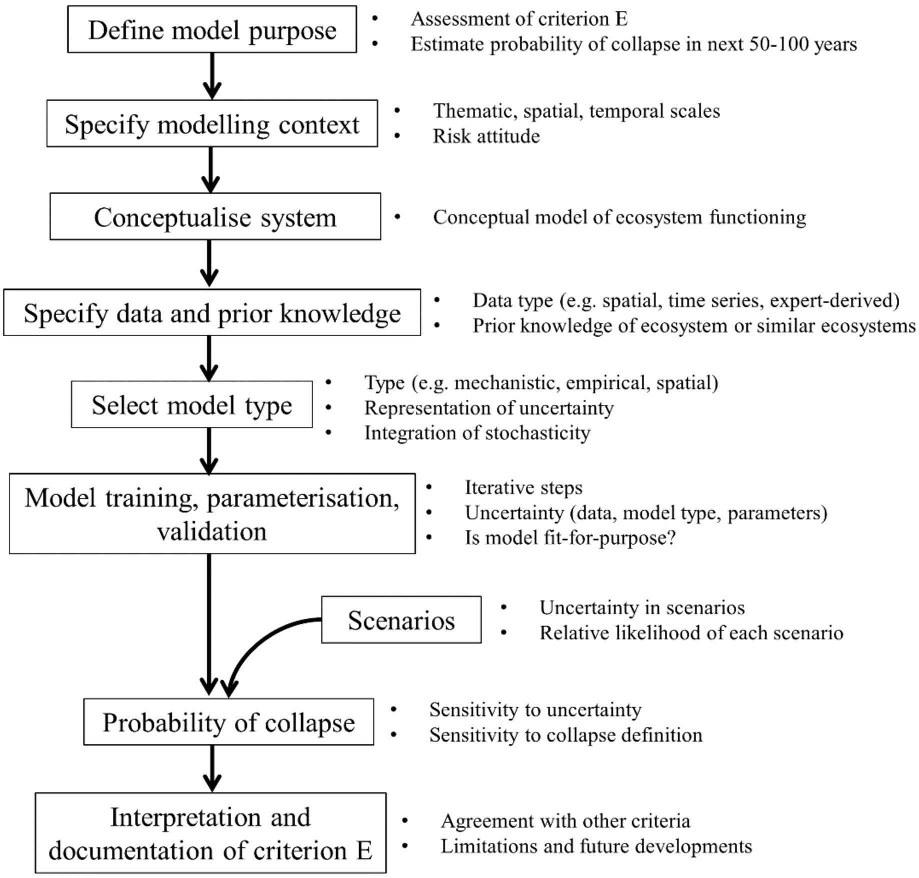


Figure 17. Nine steps to apply criterion E. Source: Authors.

1. **Define model purpose**. Models for criterion E should provide an adequate prediction of the risk of collapse over a period of 50–100 years. If the model used for criterion E is being adapted from a model with different objectives (e.g. providing guidance for management and decision-making), it may be necessary to modify its objectives and implementation. Although other objectives may be important in model-building, models for criterion E will be reviewed based on the quality of their predictions for the purpose of the RLE assessment.

2. **Specify modelling context.** Ecosystems are inherently scale-dependent, so the thematic, spatial and temporal scales of ecosystem processes may affect model-building and predictions. Adequately defining the boundaries of the ecosystem under assessment is crucial – external forcing and external outputs should be clearly labelled as such. The model should aim to spatially represent all occurrences of the ecosystem; if not, adequate inferences should be made to assess the representativeness of final predictions. The time frame of predictions for criterion E is 50–100 years, which is longer than other subcriteria (A2a, A2b, C2a, C2b, D2a, D2b) and may therefore require a different understanding of future threats.

   Modelling may involve decisions relying on the risk attitude of the assessor, i.e. the relative costs of under or overestimating the risk of collapse. A precautionary but realistic risk attitude is advocated when implementing criterion E. Such decisions should be thoroughly documented within the criterion E documentation, and if possible underpinned by quantitative measures of risk aversion.

3. **Conceptualise system.** Models for criterion E should rely on a sound understanding of ecosystem dynamics and function, underpinned by data and relevant inferences from similar ecosystems. Conceptual models can help identify key ecosystem processes and variables indicating collapse. The conceptual model may depict cause-and-effect relationships or transitions among reference and collapsed ecosystem states. The conceptual model used for criterion E may differ from the general conceptual model used in the ecosystem description (Section 4.2.5 Conceptual models), as it may depict more complex relationships and include measurable variables. Deciding on an appropriate level of abstraction for key processes is a key component of conceptualisation and should consider the model purpose, context, required resolution of output and effort required for model building. A critical component of assessment under criterion E is the explicit definition of collapse as it relates to the conceptual model of ecosystem dynamics and measured variables (Section 3.2 Ecosystem collapse).

4. **Specify data and prior knowledge.** Applying criterion E requires the levels of key ecosystem variables to be predicted over specified time frames. These variables can represent spatial distribution (as in criteria A and B), abiotic environment (criterion C), and/or biotic interactions (criterion D). Suitable variables can be selected by following the processes outlined in the application sections relevant to each criterion. The data may be quantitative measurements (e.g. spatial data, time series) or expert derived. At this stage the degree of spatial and temporal aggregation of data and predictions may be revised, to match ecosystem dynamics to the modelling context. For example, it may be appropriate to aggregate daily or monthly data to yearly time steps. In data-poor situations, it may be possible to infer processes and data from similar ecosystems (Maxwell et al., 2015). This should be clearly indicated and discussed within the model documentation.

5. **Select model type.** A diverse range of simulation models of ecosystem dynamics allow the probability of ecosystem collapse to be estimated directly. Selection of an appropriate model type will depend on: (i) ecosystem dynamics; (ii) data availability; (iii) representation of uncertainty; and (iv) integration of stochasticity. Some models may be more appropriate to represent specific ecosystems and their dynamics (e.g. hydrologic models for wetlands, global vegetation models for forests). The type of input data may also constrain model choice (e.g. some model types may be unable to handle missing data or expert-derived data). Models should be chosen or adapted so that appropriate uncertainty and sensitivity analyses can be conducted. Ideally, model uncertainty should be addressed by implementing multiple models representing alternative interpretations of ecosystem dynamics. Finally, ecosystem dynamics rely on stochastic processes, so models should be chosen or adapted so as to integrate stochasticity (see Coorong Lagoon case study in Appendix S2.19 in Keith et al., 2013).

   Candidate model types for the application of criterion E include:

   - State-and-transition models (Lester & Fairweather, 2009; Rumpff et al., 2011, Maxwell et al., 2015).
   - Mass-balance models (e.g. Ecopath, Models of Intermediate Complexity) (Christensen & Walters, 2004, Plagányi et al., 2014).
   - Bifurcation plots (Holdo et al., 2013).
   - Network theory (e.g. Community Viability Analysis) (de Visser et al., 2011).
   - Dynamic Global Vegetation Models (Scholze et al., 2006).

- Dynamic species distribution and population models (Midgley et al., 2010, Keith et al., 2008).
- Spatial models (e.g. cellular automata) (Soares-Filho et al., 2002).
- General ecosystem models (e.g. the Madingley model; Harfoot et al., 2014).

6. Model training, parameterisation, validation. Models should follow best practice recommendations for each model type, and should be appropriately trained, parameterised and/or validated. For example, the data-derived state-and-transition model of the Coorong Lagoon was validated through multiple pathways, so that neither states nor transitions were determined *a priori* (Lester & Fairweather, 2011). For some models, full validation may not be possible. In these cases, model performance can be evaluated with relevant performance indicators, e.g. satisfactory reproduction of observed behaviour, absence of correlation in model residuals (Jakeman et al., 2006). Model training, parameterisation and validation may occur in iterative steps that should be thoroughly documented. It may be appropriate to assess the effects of data uncertainty, parameter uncertainty and model uncertainty through sensitivity analyses. Overall, assessors should demonstrate that the model is fit for purpose for application in criterion E.

7. Scenarios. Future scenarios representing likely threats and changes to ecosystem dynamics should be identified. It is important to recognise that concepts and data underpinning scenarios may be subject to high levels of uncertainty, the effects of which may be difficult to track in large models (e.g. climate change projections; Kujala et al., 2013). Often, the relative likelihood of each future scenario will not be known (Peterson et al., 2003), so the final likelihood of collapse may be expressed as a range of values rather than a single estimate.

8. Probability of collapse. The estimate of the probability of collapse may be a single value, but in most cases in may be expressed as a range of values representing uncertainty in model-building. Sensitivity analyses of the probability of collapse may be done relevant to: (i) data, model and parameters uncertainty; (ii) scenario uncertainty; and (iii) other forms of uncertainty that may affect modelling outcomes, e.g. the choice of variables to assess ecosystem collapse. A sensitivity analysis on the threshold of collapse should be conducted in all models, as the final outcome for criterion E may be particularly sensitive to the definition of collapse. In simulations of the Mountain Ash Forest (Burns et al., 2015), for example, the collapse threshold would need to decrease from an average of one hollow-bearing tree per hectare to 0.7 to change the risk assessment outcome.

9. Interpretation. Criterion E provides an overarching framework for the application of the other criteria, and includes ecosystem dynamics that may not be captured by other criteria. It may therefore be useful to compare the outcome for criterion E with the outcomes of other criteria and provide insights into possible reasons for differences in assessment outcomes.

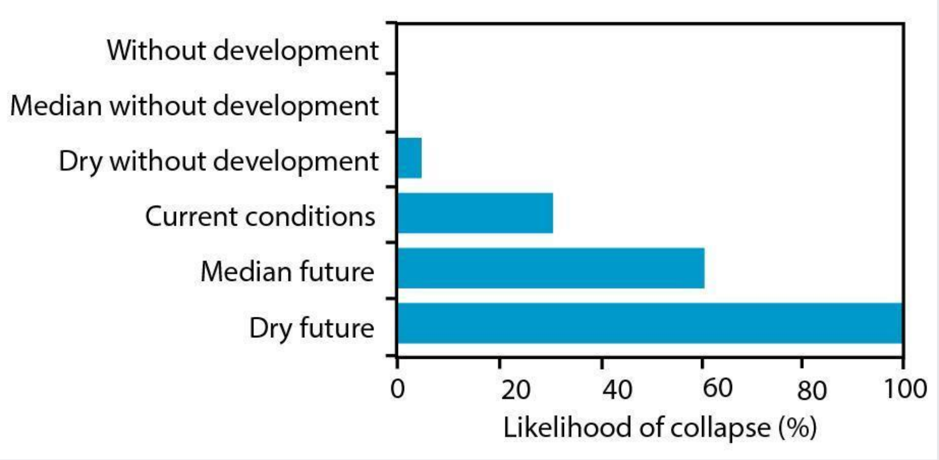
8.3.2. Documentation

A greater level of documentation is required for criterion E than for other criteria, given the scientific nature of modelling and the effects of uncertainty. It is recommended that assessors publish their models in the peer-reviewed literature and place their materials (data, code) in data repositories to allow full scrutiny of models and their outcomes. Within the RLE peer review, risk assessment and modelling experts will review models against strict criteria and may request additional analyses. Specific guidance and examples of the application of criterion E are currently under development, and will be made available on the IUCN RLE website (www.iucnrle.org).

Box 19. Developing a quantitative model of ecosystem dynamics (criterion E).

The probability of ecosystem collapse has been estimated for the **Coorong Lagoon** of South Australia, through the adaptation of an empirically derived state-and-transition model (Appendix S2.19 in Keith et al., 2013; Lester & Fairweather, 2011). Ecosystem collapse occurred when half of the modelled years occurred either in degraded ecosystem states or in a period of recovery following the occurrence of degraded states.

The quantitative assessment of the likelihood of ecosystem collapse in the Coorong was undertaken with a chain-of-models (Lester & Fairweather, 2011). Downscaled simulations from multiple global climate models were applied to hydrologic models for the Murray-Darling Basin to estimate a time series of flows. Six scenarios were investigated to quantify the likelihood of ecological collapse in the Coorong based on three climate projections for 2030 and two extraction levels (i.e. with, and without current infrastructure and extraction). All scenarios were run for a period of 114 years (Lester & Fairweather, 2009). Given that each scenario should be interpreted as 114 years of possible variability due to climatic fluctuations, the proportion of years occurring in degraded or recovery states provides an assessment of the stochasticity within the system.



Likelihood of collapse of the Coorong Lagoon under six scenarios of climate change and water extraction.
The three climate scenarios are: historical sequence since 1895; the median future climate projection based on three climate change scenarios from 15 global climate models; and a dry future climate projection based on the 10^{th} percentile of the same models.

Of the six scenarios investigated, ecological collapse occurred in four. Water extraction will not cease in the Murray-Darling Basin, so the 'without development' scenarios can be discounted from the overall calculation of risk of collapse. The likelihood of ecological collapse ranges from 30% to 100% across three scenarios representing current levels of development. The Coorong Lagoon is thus listed as Critically Endangered (plausible range Endangered – Critically Endangered) under criterion E.

9. Guidance on specific drivers of ecosystem collapse

9.1. Climate change

Climate change is a major driver of ecosystem degradation and exacerbates the impacts of other threats (e.g., habitat loss, invasive species) (IPBES, 2019). Although warming is certain, the trajectory of climate change remains uncertain as it is largely driven by social, political and technological factors (IPCC, 2022). Early impacts are being observed in many ecosystem types. Even under the best-case-scenario, some additional level of impact on ecosystems is inevitable due to lags between alterations in greenhouse gas emissions and climate changes (IPCC, 2022). Therefore, climate change is an important threatening process to consider and capture in IUCN Red List of Ecosystems (RLE) risk assessments to ensure they accurately reflect collapse risk. Explicitly considering the uncertainties in how climate change and its impacts may manifest in ecosystems is vital to identify ecosystem types at risk of collapsing due to climate change.

The RLE risk assessment protocol is well suited for assessing the diverse range of impacts of climate change on the risk of ecosystem collapse. Climate change impacts can be captured across all criteria as it can affect an ecosystem type's distribution (criteria A, B and E), environmental components (criteria C and E) and biota (criterion D and E) (Rowland et al., 2023). Climate change effects on ecosystems may manifest slowly or episodically, yet the assessment timeframes in the RLE criteria are sufficiently long to detect past and predicted future changes. The RLE protocol requires assessors to consider the ecosystem-specific mechanisms response to climate change.

9.1.1. Identify the pathways and impacts of climate change

The first step in assessing risks posed by climate change is to identify how it may alter the characteristic features and processes of the ecosystem type (Rowland et al., 2023). Changes may be non-linear, delayed by time lags, influenced by stochastic extreme events, or exhibit threshold effects (Camill et al., 2000; Walther, 2010; Wethey et al., 2011). Assessors should consider the most likely and most severe pathways of impact from climate change and use these to inform the most appropriate criteria for assessment.

Potential biotic responses to climate change include alterations to ecological interactions among species, or between species and the environment (Foden et al., 2013; Fontúrbel et al., 2021; Williams & Jackson, 2007; Walther, 2010), reduced persistence of key species, or failure to track suitable climates (Paquette & Hargreaves, 2021). Table 15 gives examples. Inferences should ideally be drawn from the target ecosystem type, but data and knowledge from analogous systems in the same functional group may also be relevant (Keith et al., 2020, 2022). Inferences may also be drawn from areas that currently have similar climates to those projected (Dobrowski et al., 2021) (e.g., Analogue Atlas Database: https://plus2c.org) or from space-for-time substitution along climatic gradients (Lester et al., 2014).

Table 15. Examples of mechanisms and indicators of ecosystem response to climate change from Red List of Ecosystems assessments.

Biotic indicator variables	*Abiotic drivers (measured or assumed)*	*Relationship to climate change*
Changes in algal or invertebrate abundance	Warming sea temperature and earlier sea ice breakup	Algae beds are replacing Antarctic marine invertebrate communities under warming temperatures and earlier sea ice breakup, which facilitates greater light influx promoting algal growth and competitive exclusion of benthic invertebrates (Clark et al., 2015).
Frequency of coral bleaching events	Increasing frequency of extremely high sea surface temperature episodes	Algal symbionts evacuate coral polyps during episodes of extreme high sea-surface temperatures, resulting in mass coral bleaching and mortality. Increased frequency of bleaching limits coral recovery and leads to continued degradation and collapse (Hughes et al., 2017; Bland et al., 2017; Obura et al., 2021).
Disruption of dispersal and pollination	Divergence in environmental suitability for pairs of interacting species	Change in the distributional overlap of key plants and animals due to climate change may cause the loss of key dispersal-pollination processes essential for maintaining Colombian terrestrial ecosystems (Etter et al., 2015, 2017).
Abundance of hollow-bearing trees	Increase in frequency of severe fires	Increasing frequency, duration and severity of extreme fire weather as climate changes is increasing the rates of mortality and collapse of hollow-bearing trees. Hollows provide critical habitat for arboreal marsupials and birds in mountain ash forests (Burns et al., 2015).
Declining oyster abundance	Increasing sea surface temperatures	Oysters are ecosystem engineers in oyster reefs and are affected by changes in climate that cause thermal stress and higher prevalence of disease (Gillies et al., 2020). Other drivers (e.g. coastal pollution, turbidity, oyster harvest) may also reduce oyster abundance, independent of climate change.
Shrub encroachment	Earlier snow melt and longer growing season	Increased shrub cover and resulting competitive exclusion of smaller herbaceous plants from snowpatch herbfields results from warming temperatures and earlier snow melt, which enables the establishment of shrubs and changes the characteristic composition and structure of the ecosystem (Williams et al., 2015).
Weed invasion	Decline in rainfall and increased frequency of fires	Declines in rainfall can promote more frequent fires which enable the entry and establishment of invasive alien plants, degrading the shrublands (English & Keith, 2015). Alien plant invasions may occur independently of climate change in response to fragmentation and enrichment of nutrients and water from adjacent agricultural areas and transport corridors.

Where possible, the adaptive capacity of characteristic native biota and ecosystem resilience, and any uncertainty in this capacity should be incorporated into predictions of climate change responses (Bland et al., 2017), particularly for components that are essential to the functioning of the ecosystem type (Angeler et al., 2019; Thompson & Fronhofer, 2019), such as foundation species.

Predicted ecosystem responses should also address interactions between climate change and other threats that may increase risk of collapse (Scheffer et al., 2015; Titeux et al., 2016). For example, habitat loss, degradation or overexploitation may reduce ecosystem resilience to climate change (Moomaw et al., 2018; Keith et al., 2023a). Climate change may similarly make ecosystems more vulnerable to other threats, such as loss of sea ice due to warming temperatures increasing human activity in polar regions (Corell, 2006) or increasing the suitability of land for more intensive uses.

Conceptual models may help to summarise understanding and simplify potentially complex dependencies and uncertainties to inform climate-related Red List assessments (Section 4.2.5 Conceptual models). These may be developed into more quantitative models to implement the assessment. For example, Bayesian belief networks may incorporate different plausible pathways, or causal networks to estimate interactions within the model and the resulting impacts on collapse risk (Peeters et al., 2022). These predicted relationships and the level of uncertainty in each can be based on published evidence and expert knowledge (Hemming et al., 2017). Assessors should identify and report all assumptions.

9.1.2. Select abiotic and biotic indicators to assess climate change impacts

RLE risk assessments for ecosystem types likely to be affected by climate change should include analysis of climate-sensitive indicators under criterion C and/or criterion D (Rowland et al., 2023). These indicators should also fulfil the other criteria for analysis in an RLE risk assessment (Table 11; Section 7.3.1 Selection of indicator variables). Indicators of climate change response should therefore be ecosystem-specific, direct, informative and sensitive. Understanding of functionally similar ecosystem types may inform the selection of suitable indicators (Keith et al., 2022). Raw climate variables (e.g. temperature, precipitation) are not acceptable indicators unless a clear mechanistic link with ecosystem collapse is demonstrated (e.g. sea surface temperature extremes and mass coral bleaching; Hughes et al., 2017) or unless they are influential predictors in a habitat suitability model (e.g. precipitation as part of the hydrological budget of mire ecosystems; Keith et al., 2014; Section 7.3.2 Multiple indicator variables).

Assessors are encouraged to evaluate multiple indicators where relevant for each criterion to represent the range of ecosystem degradation pathways that may plausibly be driven by climate change (Tonmoy et al., 2014; Rowland et al., 2018). Assessors may use sensitivity analyses to identify which variables or relationships most affect the risk outcome (Bland et al., 2017, 2018; Murray et al., 2020). See Table 12 and Table 13 for examples of indicators used to assess the impacts of climate change under criterion C and criterion D, respectively.

9.1.3. Setting collapse thresholds

Setting reliable collapse thresholds for variables used to estimate future climate change impacts may be hampered by uncertainty in ecosystem resilience and the adaptive capacity of component biota under climate change (Thompson & Fronhofer, 2019). There are several approaches assessors can take to inform collapse thresholds for indicators under climate change:

1. Empirical evidence on the limits of physiological tolerance of characteristic native biota, particularly structural or functional dominants (e.g. Körner & Paulsen, 2004; Berdanier, 2010; Schoepf et al., 2015; Shu et al., 2019).
2. Observations of local ecosystem collapse under extreme conditions. These include observations of ecosystem response under extreme climate events. For example, Micklin and Aladin (2008) documented salinity levels that resulted in loss of characteristic native biota from a freshwater lake as its volume declined and salts became more concentrated. Keith et al. (2023) documented soil moisture levels in drying mires that rendered their peaty sediments prone to combustion in bushfires.
3. Inferences drawn from the bioclimatic limits of an ecosystem type, where the fundamental niche and realised niche are likely to be aligned. See point 5 for a quantitative approach based on habitat suitability models.
4. Trait-based generalisations about the vulnerability of characteristic native biota, particularly structural or functional dominants (Foden et al., 2013; Pacifici et al., 2015; Lankau et al., 2015).
5. Models of ecosystem distribution or dynamics (Keith et al., 2014; Matías & Jump, 2014; Bland et al., 2017). Potentially suitable correlative habitat suitability models are described in Section 7.3.2 Multiple indicator variables, while mechanistic models of ecosystem dynamics are described in Section 8.3 Applying criterion E. Assessors may use the increasing number of experimental studies that aim to identify causal relationships between environmental changes and collapse (Shu et al., 2019).

Inherent uncertainty in thresholds of collapse under climate change should be represented with a best estimate with lower and upper bounds (Bland et al., 2018). A sensitivity analysis could be undertaken to quantify the impact of uncertainty in the collapse threshold on the collapse risk (Bland et al., 2017).

9.1.4. Quantifying climate-related risks

Climate change impacts on ecosystems may be assessed via the same approach as for any other ecosystem change (see Section 5.3 Applying criterion A and Section 7.3 Applying criteria C and D) over the recent past (subcriteria A1, C1, D1), historical past (subcriteria A3, C3, D3) or near future (subcriteria A2, C2, D2) (see Section 3.3.1 Time frames). As the RLE assesses overall risks to ecosystems, quantifying declines in distribution or degradation is more important than their attribution to climate change *per se*, except to inform the initial choice of indicators that represent the main mechanisms of decline (Section 4.2.5 Conceptual models and Section 7.3.1 Selection of indicator variables).

Data sources for time series of the relative severity and extent of degradation are largely determined by the choice of climate-related indicators and may include the following:

1. Time-series observations from on-ground samples or remote sensing, with interpolation or extrapolation to the standard time frames, subject to well justified assumptions.
2. Experimental studies that test ecosystem responses to potential future climates, subject to well-justified assumptions that the data are appropriate, realistic estimates of ecosystem responses to plausible trajectories of climate change.
3. Estimated climate responses for component biota, including structural and functional dominants, ecosystem engineers and keystone species. The IUCN Red List of Threatened Species (IUCN SPC, 2024) may be a useful source of evaluations of species vulnerability to climate change and potential changes to their population and/or distribution (Carpenter et al., 2008; Brummitt et al., 2015; Foden et al., 2019) for

estimating potential changes to the distribution (subcriterion A2) or biota (subcriterion D2), or as the basis for a quantitative risk analysis (criterion E).

4. Inferences drawn from responses to historical climates may be used to estimate potential changes in distribution or integrity due to shifts in the suitability of environmental conditions (Fernández et al., 2015).
5. Expert judgments, informed by available evidence and personal experiences, may supplement empirical data (Morgan et al., 2001; McBride et al., 2012; Bland et al., 2018). Expert elicitation can be used to assess the relevance and reliability of climate projections or provide additional estimates of climate change and the associated uncertainty based on local ecological processes (Dessai et al., 2018; Grainger et al., 2022). Subject to RLE standards (Section 3.3.3 Standards of evidence and dealing with uncertainty, Section 3.3.4 Quantitative data and expert knowledge and Section 4.4.1 Dealing with uncertainty), expert-derived estimates of variables may be used to evaluate the RLE criteria or to underpin the links between ecosystem dynamics and environmental change in ecological models (Korell et al., 2019). Robust approaches based on structured elicitation of estimates, aggregation across multiple experts and bounded estimates of uncertainty (Martin et al., 2012; Hemming et al., 2017) are strongly recommended.
6. Projections from ecological and climate models, subject to requirements in Section 9.1.5 Forecasting with models.

9.1.5. Forecasting with models

Climate projections are often used as a data source in RLE risk assessments. They can be used independently to assess criterion C2, or as input data for ecological models for criteria D2 or E, or to quantify continuing decline of environmental quality under the condition (a)ii of subcriteria B1 and B2. Capturing and reporting the uncertainty stemming from these models is vital to outlining the level of confidence in the risk outcome. There are several factors that can influence the goodness of predictions of collapse risk using models:

1. **Climate scenarios.** Global circulation models that produce climate projections are parameterised by different scenarios of greenhouse gas emissions that are based on socio-economic factors such as population growth, energy use, technological development and policy (IPCC, 2021). The Intergovernmental Panel on Climate Change (IPCC) develops and periodically updates a wide range of these scenarios known as Relative Concentration Pathways (RCPs). Assessors should calculate the risks of ecosystem collapse based on a range of the most up-to-date plausible high and low emission scenarios to represent the plausible bounds of climate projections and ecosystem responses. Assessors should clearly report which scenarios were selected, the assumptions in each scenario, and justify those used.

2. **Climate model selection.** A broad range of climate models are available and their projections, and hence the ecological predictions and risk outcomes in an RLE risk assessment, can be strongly affected by model structure and parameterisation (Harris et al., 2014; Rising et al., 2022). Moreover, climate models vary in skill between regions and depending on the climate variable projected (Hawkins & Sutton, 2009; Freer et al., 2018). Therefore, a given model may perform better for some variables or regions and worse for others.

   Assessors should examine projections from multiple climate models, ideally a subset of the most skilled for the region and variable(s) of interest, to estimate plausible bounds of the resulting risk outcome from each. Where projections representing a multi-model mean are used, as in climate model ensembles, assessors should present the variability around the mean, where available, to capture the range of potential futures.

Projections derived from climate models may be used in ecological models, for example to project habitat suitability or simulate ecosystem properties under future climate scenarios (e.g. Ruiz-Labourdette et al., 2012; Keith et al., 2014; Bland et al., 2017; Briscoe et al., 2019; Ferrer-Paris & Keith, 2024). The issues discussed above for climate models similarly apply to ecological models. The uncertainties in model structure and parameterisation should be explored and incorporated into bounded estimates of indicators and risk outcomes. Further guidance on use of ecological models is available in Section 12 of the *Guidelines for Using the IUCN Red List Categories and Criteria* (IUCN SPC, 2024).

3. Model realisations. Variation among model realisations (or runs) is another source of uncertainty stemming from using model projections (Freer et al., 2018). Climate models include stochastic elements to estimate a statistical probability distribution of potential outcomes (as model realisations) and account for random variation (with plausible bounds) in one or more parameters. The variability among model realisations in global or regional climate projections is often summarised as average and dispersion of model outputs. Yet the extreme predictions are important for assessing risk, as these may be the most limiting factors for the persistence of an ecosystem type. Where information on the variability among model realisations is available, assessors should incorporate this information in the RLE risk assessment in the plausible bounds of the risk category. Where this information is not available, the limitations of using the mean projections should be considered and reported in the assessment.

4. Resolution of projections. Climate projections are typically produced at a coarse spatial scale, e.g. 250 km (IPCC, 2021). This makes it challenging to reliably link the projections to impacts in specific ecosystem types, particularly for ecosystem types that are strongly driven by microclimates or patchily distributed, even where there are clear ecological relationships.

   Projections may be downscaled to a finer spatial resolution that may be suitable for applying in RLE risk assessments. Downscaling approaches include dynamical downscaling into regional climate models, statistical downscaling or simple scaling. However, assessors must be careful to avoid misinterpreting the accuracy and precision of these down-scaled data (Harris et al., 2014). The extent of the ecosystem type may affect the importance of spatial uncertainty on the risk outcome (Harris et al., 2014). Ecosystem types that are broadly defined or cover large areas may be less prone to effects of coarse resolution climate data and downscaling.

   Assessors must articulate and justify assumptions for the use of coarse or downscaled climate projections in Red List assessments of the ecosystem type(s) of interest.

9.2. Fragmentation

9.2.1. Fragmentation theory

Ecosystem fragmentation, in these Guidelines, refers to the disruption of ecosystem processes due to the breaking apart of an ecosystem distribution into patches for a given amount of ecosystem loss (Figure 18; Fletcher et al., 2023), i.e. 'fragmentation *per se*' of Fahrig (2003). Thus, the same ecosystem depleted to a given extent may be exposed to different levels of fragmentation, depending on the spatial configuration (number, size, shape, isolation) of the resulting patches, as well as the properties of the matrix and their degree of contrast with patch properties, the traits of biota (persistence, dispersal), and interactions among biota including dependencies, competition, predation, etc. (Laurance, 2008; Fletcher et al., 2023).

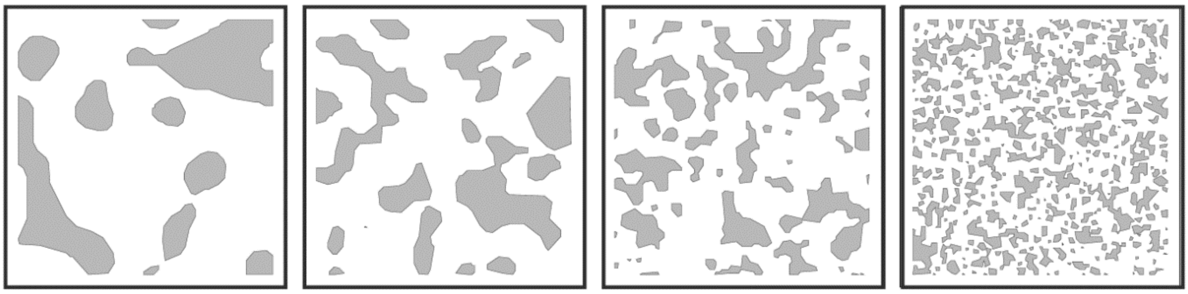


Figure 18. Schematic representation of landscapes showing increasing levels of fragmentation for the same ecosystem extent (grey patches), left to right.
Source: Fletcher et al., 2023.

Fragmentation affects ecological processes at all levels of organisation (Haddad et al., 2015) in terrestrial, subterranean, freshwater, marine and transitional ecosystem types. Mechanisms of effects are diverse (Table 16), as are the symptoms expressed through alterations to movement of biota, rates of extinction, rescue and recolonisation, changes to habitat suitability, interactions and dependencies among species, nutrient retention, succession, trophic dynamics and disturbance regimes (Haddad et al., 2015; Ntshanga et al., 2021; Fletcher et al., 2023). Fragmentation may involve short-term and lagged long-term effects on ecosystem composition, structure and function, including extinction debts and function debts (Haddad et al., 2015).

Table 16. Mechanisms of fragmentation effects and indicators to assess them. Source: Adapted from Fletcher et al., 2023.

Mechanism of fragmentation	*Direction of effect*	*Rationale*	*Scale*	*Mediating factors*	*Direct biotic indicators*	*Indirect spatial indicators*	*References*
Edge effects (attrition)	Negative	Attrition of core-dependent properties of ecosystem as edges increase (responses to changes in nutrients, light, pollutants, physical disturbance, exposure).	Within patch	Edge width	Abundance or proportion of core-dependent biota	Core area, Edge density	Ries et al. (2004); Kemper et al. (1999); Ewers & Didham (2008)
Edge effects (invasions)	Negative	Greater edge and smaller core facilitates entry of new competitors, predators and pathogens.	Within patch	Edge width	Abundance and extent of invasive species or edge opportunists	Core area, Largest patch index, Edge density	Horn et al. (2011)
Patch diminution and degradation	Negative	Reduced habitat suitability and carrying capacity of biota.	Patch	Patch size, dispersal ability of different biota	Area-weighted species diversity	Largest patch index, Core area	Cowling & Bond (1991); Hanski (1999); Fletcher et al. (2018); Chase et al. (2020)
Conspecific attraction or aggregation	Negative	Behavioural aggregation leading to reduced occupation of small patches and antagonistic effects on conspecifics and other biota.	Patch	Patch size	Area-weighted evenness of behaviour-ally aggregated biota	Largest patch index	Fletcher (2006)
Movement effects	Negative	Reduced movement through matrix compared to within patches, reducing gene flow, trophic subsidies, rescue and recolonisation.	Between patch	Matrix permeability, patch connectivity, dispersal ability	Abundance of dispersal-limited biota	Aggregation index, Proximity index	Doak et al. (1992); Haddad et al. (2015)

Mechanism of fragmentation	*Direction of effect*	*Rationale*	*Scale*	*Mediating factors*	*Direct biotic indicators*	*Indirect spatial indicators*	*References*
Competition-dispersal trade-offs	Negative	Strong competitors are disproportionately disadvantaged by declining patch size and increasing isolation.	Between patch	Trait trade-offs for competition and dispersal abilities	Abundance of dispersal-limited biota	Largest patch index, Aggregation index, Proximity index	Tilman et al. (1997)
Destabilisation of predator-prey interactions	Negative	Increased predator foraging efficiency in matrix, increased predator density in patches, reduced prey refuges within patches.	Between patch	Contrast in prey exposure and predator movement between core, edge and matrix	Abundance of prey or prey refuges	Core area, Largest patch index	
Reduced within-patch niche diversity and refuge availability	Negative	Reduced carrying capacity (especially specialised resources e.g. tree hollows), greater exposure to predation and extreme events.	Within patch	Core-edge contrast	Abundance of limiting resources or their dependent biota	Core area, Largest patch index	
Reduced biotic interactions and mutualisms	Negative	Reduced abundance of partners in facilitation or mutualistic interactions (e.g. pollination, epiphytes).	Within patch	Dispersal ability	Abundance of partner-dependent biota	Core area, Largest patch index	Pauw et al. (2007)
Susceptibility to catastrophes (fires, storms, floods, tsunami)	Negative	Increased exposure, increased initiation or reduced recovery with increasing edge length.	Patch	Patch-matrix contrast, exposure	Abundance of catastrophe-sensitive biota, structural disruption	Core area, Largest patch index, Edge density	Bellingham (2008); Cochrane & Laurance (2002)

Mechanism of fragmentation	*Direction of effect*	*Rationale*	*Scale*	*Mediating factors*	*Direct biotic indicators*	*Indirect spatial indicators*	*References*
Disruption of disturbance regimes	Negative	Reduced contagion of disturbances between patches (across matrix).	Landscape	Matrix permeability relative to within patch	Abundance of disturbance-dependent biota, area-weighted disturbance frequency	Aggregation index, Proximity index	Yates & Ladd (2010); Slingsby et al. (2020)
Risk spreading or insurance effects	Positive	Environmental disturbances or stochasticity is less synchronous between than within patches.	Landscape			Aggregation index, Proximity index, Patch number	den Boer (1968); Kallimanis et al. (2005)
Geometric fragmentation effects	Positive	More, smaller patches have greater spread across landscape, better sampling clustered species distributions.	Landscape			Aggregation index, Proximity index, Patch number	May et al. (2019)
Landscape complementation	Positive	Access to spatially separated resources increases with number of patches due to a greater proportion of edge.	Landscape			Edge Density	Dunning et al. (1992)
Habitat diversity	Positive	Greater resource variability across patches than within owing to non-stationary or autocorrelated environmental gradients.	Landscape			Aggregation index, Proximity index, Patch number	Lasky & Keitt (2013)

9.2.2. Assessing risks from fragmentation

Application of the RLE protocol to assess risks pertaining to fragmentation-related threats requires the following steps:

1. **Justify fragmentation processes as cause of patchy distribution**. Ecosystem risk assessment based on the RLE protocol must first diagnose the spatial pattern of patches as the outcome of a fragmentation process, whereby the distribution of an ecosystem type results from the breaking apart of a more continuous distribution into patches and/or patches of occurrence become smaller over time. In most cases, this results from the expansion of anthropogenic infrastructure or activity (e.g. agriculture, urban/industrial/service infrastructure, barriers to surface or stream flows, disruption of aquifers, coastal or marine infrastructure, deposition of anthropogenic debris, etc.). Ecosystem types with naturally patchy distributions or with patch configurations that have not been affected by any fragmentation processes cannot be assessed for risks related to fragmentation. Non-fragmentation processes that may result in patchy ecosystem distributions include unassisted colonisation and assembly of biota into naturally isolated suitable habitat patches that occur within a matrix of unsuitable or less suitable habitat. Examples include wetlands in a dryland matrix, long-established islands in an aquatic matrix, rocky outcrops on plains or hills, distinctive substrate types or topographic features within a contrasting land or the seafloor matrix, etc. To justify an analysis of fragmentation in an RLE assessment, assessors must first identify the evidence supporting an inference that fragmentation processes at least partially caused the derivation of a patchy ecosystem distribution from a more continuous one. The nature of evidence may relate to anthropogenic origin of the matrix and its likely expansion replacing suitable habitat for the ecosystem type and/or time series or qualitative historical information about the former distribution of the ecosystem type.

2. **Assess criterion A: Accelerated decline of area due to fragmentation and edge effects.** Fragmentation can lead to an accelerated decline in core area (due to an increased edge:area ratio). If there is evidence of a strong edge effect (ecosystem functions are severely degraded in the edge areas), assessors could calculate decline in total area and decline in core area as lower and upper bounds of declines for criterion A. The definition and calculation of edges and core area should be consistent between all data points used for estimation of decline.

3. **Assess criterion B: Restricted distribution and threatening process**. Fragmentation may exacerbate risks to ecosystem types with restricted distributions. To assess risks related to fragmentation under criterion B, the distribution of the ecosystem type must first be assessed against spatial thresholds for threatened status under either subcriterion B1 (extent of occurrence) or B2 (area of occupancy). If these thresholds are met for any threat category, then qualitative evidence of fragmentation may be assessed to determine whether it meets the requirements for a threatening process (see Section 6.3 Applying criterion B). That is, ongoing fragmentation effects or ongoing legacies of fragmentation (e.g. extinction debt, functional debt; Haddad et al., 2015) "that [are] likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years" (IUCN, 2016). Continuing declines are not likely if the biophysical and biotic properties of the ecosystem fragments have reached a stable equilibrium with the matrix. Ecosystem types that meet both conditions (spatial thresholds and evidence of ongoing threats posed by fragmentation or its legacies) are eligible for threatened status under criterion B1b or B2b.

4. **Assess criterion D: Disruption of biotic processes and interactions.** Quantitative evidence of fragmentation effects may be assessed under criterion D as a symptom of disruption of biotic processes and interactions within the ecosystem type over any of three time frames (D1 recent past, D2 future, D3 historic past; see Section 3.3.1 Time frames). Subject to appropriate justification for the ecosystem type, criterion D may be assessed either with an indicator that represents a biotic ecosystem property responsive to the effects of fragmentation (examples in Table 11 and Table 16) or a spatial proxy that measures the severity of change in the spatial configuration of the ecosystem type (examples in Table 16 and Table 17). Criterion D should be assessed according to the following steps (a–f).

    a. **Diagnosis of mechanisms of fragmentation effects.** The mechanisms that result in fragmentation effects or symptoms must be identified. For guidance, Table 16 identifies and describes candidate mechanisms, but may not be exhaustive. Evidence on mechanisms may be drawn from landscape experiments (empirical studies), indirect (non-experimental) observational studies, inferred from functionally similar ecosystems exposed to similar threats, or projected using models and plausible scenarios of future landscape or seascape change (Tilman et al., 1997; Laurance et al., 2002; Haddad et al., 2015).

    b. **Select appropriate ecosystem-specific temporal indicators of fragmentation effects and/or processes.** Indicators of fragmentation for assessment under criterion D must be based on the diagnosed mechanisms of fragmentation effects (Table 16; Step 4a above). Effects of ecosystem fragmentation on the risk of ecosystem collapse may be assessed directly via a spatially structured time series of ecosystem response variables or indirectly by time series of proxy spatial indicators (Table 17). Direct indicators should generally be given higher priority and weight in an assessment than indirect proxies because direct observations of effects usually involve fewer assumptions about fragmentation effects on ecosystem processes and biota (Laurance et al., 2002; Haddad et al., 2015). Where use of both direct and indirect indicators of fragmentation is justified, the indicator producing the greatest estimate of relative severity and extent of ecosystem degradation should determine the assessment outcome under criterion D unless limitations in the data warrant weighted averaging of the estimates. Weighting decisions and methods should be documented and explored with sensitivity analyses to evaluate the robustness of the assessment outcome.

Table 17. Examples of spatial proxies for fragmentation effects.
Source: After Ene & McGarigal, 2023.

Metric	*Definition*	*Method of calculation*
Core area	The summed core area of all patches, based on specified buffer width (i.e. edge depth), as a percentage of the total area of all patches.	CORE = $\Sigma(A_{core(e)})/A_{total}*100$, where $A_{core(e)}$ is the core area for a specified edge width to be summed for all patches, A_{total} is the total extent of the ecosystem type at the relevant times specified by the subcriterion.
Largest patch index	Area of the largest patch as a percentage of the total extent of the ecosystem type.	LPI = $A_{largest}/A_{total}$, where $A_{largest}$ is the area of the largest patch and A_{total} is the total extent of the ecosystem type.
Aggregation index (landscape)	The number of like adjacencies involving the corresponding class, divided by the maximum possible number of like adjacencies involving the corresponding class (i.e. when the class is maximally clumped into a single, compact patch), summed for all patches; multiplied by the proportion of the study area occupied by the ecosystem type and by 100 (to convert to a percentage).	AI = ((Σ(g/max(g)))*P)*100, where g is the number of adjacent grid cells, max(g) is the maximum adjacencies possible when cells are grouped into a single patch, and P is the proportion of the study area occupied by the ecosystem type. If A is the area of the ecosystem type (in terms of number of cells) and n is the side of a largest integer square smaller than A, and m = A - n2, then the largest number of shared edges, the: max(g) = 2n(n-1), when m = 0, max(g) = 2n(n-1) + 2m -1, when m ≦ n, or max(g) = 2n(n-1) + 2m -2, when m > n.
Edge density	The total length of edges resulting from fragmentation processes (excludes edges or boundaries with other non-anthropogenic ecosystem types).	ED = $\Sigma(E_{frag})/A_{total}$, where E_{frag} is the total length of edges of the ecosystem type created by fragmentation processes and A_{total} is the total area of the ecosystem type in the study area at the same time E_{frag} is measured.
Proximity index	The sum of patch area divided by the square of edge-to-edge distance to the nearest patch (within a specified neighbourhood threshold), summed across all patches of the ecosystem type in the study area.	PROX = $\Sigma(A_{patch}/d_{min})$, where A_{patch} is the area of a patch and d_{min} is the edge-to-edge distance to the nearest patch within a specified neighbourhood distance threshold.

c. **Analysis of indirect spatial proxies.** Indirect spatial proxies (Table 17) may only be assessed under criterion D if each of the following conditions are met and justified in documentation:
   i. A metric appropriate to the ecosystem type is justified with reference to the mechanism of fragmentation effects with reference to step A and Table 17; AND
   ii. Data for direct indicators are unavailable, or there are identified limitations in the available data on direct indicators that limit inferences on trends; AND

   iii. The spatial data delineating patches is justified as appropriate to the ecosystem type. For example, whether woody cover spatial layers adequately delineate forest ecosystem patches from an anthropogenic matrix depends on habitat suitability of the matrix for key biota, movement ability of key forest biota through the matrix, thresholds of woody cover used to differentiate forest from the matrix, and whether elements such as small clumps of trees or individual trees are interpreted and mapped as part of the forest or matrix.

   Where use of two or more spatial proxies for fragmentation is justified, the indicator producing the greatest estimate of relative severity and extent of ecosystem degradation should determine the assessment outcome under criterion D unless limitations in the data or relevance of the proxy warrant weighted averaging of the estimates. Weighting decisions and methods should be documented and explored with sensitivity analyses to evaluate the robustness of the assessment outcome.

d. **Determine edge buffers for spatial proxies.** Some spatial proxies of fragmentation (e.g. Core area, Table 17) require delineation of edge buffers. Buffers must be applied:
   i. Only to edges derived from a fragmentation process, e.g. calculation of core area should only discount edge buffers adjoining an anthropogenic matrix and no buffers should be applied to discount edges along boundaries with other natural, or similarly semi-natural ecosystem types.
   ii. Using a buffer width appropriate to the fragmentation mechanism, patch properties and matrix properties. Appropriate buffer widths vary from 10 m (Laurance et al., 2002) to more than a kilometre (Ewers & Didham, 2008). A combination of different buffer widths may be used to calculate a spatial proxy (e.g. core area, edge:area ratio) if justified by different matrix types of mechanisms of fragmentation effect.

e. **Assemble fragmentation time series.** Assessment of criterion D requires estimates of fragmentation indicators from at least two time points to calculate the rate of change over any of the three time frames (Section 3.3.1 Time frames), i.e. assessments cannot be based on data from a single point in time. Contemporary estimates may be drawn from maps of extant ecosystem distribution (subject to requirements in Step 4ciii). Estimates for the recent past (subcriterion D1) may be based on time series of imagery (e.g. Murray et al., 2019), those for the future (subcriteria D2a & b) may be based on models of plausible future scenarios (e.g. Swenson & Franklin, 2000), and those for historic past at the advent of industrialisation (subcriterion D3) may be based on appropriate hindcasting (e.g. Bickford & Mackey, 2004). Where estimates of the fragmentation indicator(s) are available for multiple time steps, one or more regressions should be fitted to the time series assuming plausible models of change (e.g. linear, exponential, segmented, etc.). Methods of interpolation, extrapolation and scenario analysis should be applied based on guidelines (Section 3.3.3 Standards of evidence and dealing with uncertainty; Section 4.4.1 Dealing with uncertainty; Section 5.3 Applying criterion A; Section 8.3 Applying criterion E).

f. **Calculate relative severity and extent of degradation due to fragmentation.** The relative severity of decline due to fragmentation is calculated using the ratio of observed decline in the fragmentation indicator value to the decline that would occur if the ecosystem collapsed (see formula in Section 7.3 Applying criteria C and D):

   Relative severity of fragmentation = 100(It - I0)/(Ic - I0)*

   where *I0*, *It* and *Ic* is the fragmentation indicator values for the beginning of the assessment time frame (Section 3.3.1 Time frames), the end of the assessment time frame and for the collapsed state of the ecosystem. In some cases, standardisation may be necessary to separate the effect of fragmentation *per se* from decline in extent, which is assessed under criterion A. If this is the case, the standardisation method should be evaluated with sensitivity analysis to ensure that relative severity of declines is not underestimated or overestimated.

In most cases fragmentation metrics should be calculated for the entire distribution of the ecosystem type, and hence the relative severity of fragmentation will be for 100% of the extent (Section 7.3.5 Calculating extent of degradation). In special circumstances, assessors may interpret fragmentation as the cause of a patchy distribution of an ecosystem type in one portion of its range and not in another part of its range. This may occur, for example, where the socio-economic drivers and history of fragmentation differ, between countries or where a large, long-established secure protected area has prevented fragmentation from occurring in part of the range. There must be a very high level of certainty (virtually certain or very likely; Table 4) that fragmentation will not occur in such areas in the future. Where this is the case, assessors should estimate the extent of degradation in criterion D as the proportion of the total area in which fragmentation is interpreted as the cause of the patchy ecosystem distribution, and the relative severity is calculated as per steps a–f above.

10. Databasing, peer review and publication

The IUCN Red List of Ecosystems (RLE) Database compiles global ecosystem assessments and selected sub-global assessments based on the *IUCN Red List of Ecosystems Categories and Criteria*. The Red List of Ecosystems Database Management Committee is responsible for the technical development, implementation and long-term management of the database, and oversees the Red List of Ecosystems Database Hub, which in turn carries out day-to-day management of the database and coordinates the publication of assessments on the web portal (https://assessments.iucnrle.org).

There are three general routes by which assessments reach the database:

1. **RLE assessments published in scientific journals.** Global RLE assessments and key sub-global assessments that have undergone external peer-review and are published in scientific journals, and that have been identified as compliant with the RLE guidelines, are prioritised for incorporation into the database with cooperation of the authors.

2. **RLE assessments led by RLE partners.** Global RLE assessments or key sub-global assessments carried out by RLE partners, collaborator organisations and institutions, or experts from the RLE Thematic Group. An RLE expert involved with project coordination or implementation is usually responsible for ensuring that these assessments comply with RLE guidelines. Nonetheless, they must go through review process by at least one reviewer appointed by the RLE Scientific Standards Committee, to ensure conceptual consistency, scientific rigour and compliance with RLE guidelines.

3. **External projects.** Global RLE assessments or key sub-global assessments resulting from projects carried out by individuals, government agencies, research institutions and non-government organisations outside of the RLE partnership. These assessments must go through a review process by at least two reviewers, with the reviews and responses sent to the RLE Database Management Committee, to ensure the assessment complies with RLE guidelines, conceptual consistency and scientific rigour.

All three routes use the same basic process for preparing and submitting assessments for publication (Figure 19). Assessors apply the *IUCN Red List of Ecosystems Categories and Criteria* to assess the ecosystem type and document the assessment (as outlined in Section 4.5 Documentation). The assessment information is transcribed into the appropriate database formats (including XML core file and online form or other tools and formats approved by the Database Management Committee). The assessment is subject to peer review arranged by the Database Management Committee. Assessments accepted after review are submitted to the Hub Manager for final format and consistency checks, and accepted assessments are published on the IUCN RLE Database web portal.

It is recommended that authors of Red List assessments assign a unique Digital Object Identifier (DOI) to each ecosystem type, regardless of their hosting in the RLE Database, to facilitate tracking versions and updates of individual assessments. This may not be practical for some assessments with large numbers of units, but in these cases a single DOI is recommended for the collective assessment. Global RLE assessments that are not published in scientific journals should register in EcoEvoRxiv. EcoEvoRxiv (EcoEvo “archive”, https://ecoevorxiv.org/) is a not-for-profit repository for works related to ecology, evolution and conservation, that hosts pre-prints, post-prints, reports and datasets. EcoEvoRxiv is the preferred repository because it has a dedicated editor for the IUCN RLE. Other repositories will be considered in exceptional circumstances. Submitted assessments should use the standard template provided at https://iucnrle.org and include explicit mention of 'Red List of Ecosystems' in the title and keywords.

10.1. Considerations for the review process

Inclusion of assessments into the RLE Database will be subject to appropriate peer reviews determined by the RLE Committee for Scientific Standards. RLE assessors or project leads are encouraged to contact the Database Management Hub (information@iucnrle.org) to submit ecosystem risk assessments for review prior to inclusion into the database. The IUCN RLE Committee for Scientific Standards will coordinate independent peer reviews of risk assessments for the global IUCN RLE. Reviews of sub-global assessments will generally be the responsibility of project managers, though they are encouraged to seek advice from the Committee for Scientific Standards via the Database Management Hub. Assessments will be reviewed by at least two experts: one with expertise in the ecology of the ecosystem type(s) under assessment, and another familiar with the *IUCN Red List of Ecosystems Categories and Criteria*.

Appointed reviewers will take into consideration the following criteria for inclusion of assessments in the RLE database:

1. Whether the units of assessment are consistent with the conceptual definition of an ecosystem type (Section 3.1 Ecosystem types: the units of assessment and Section 4.2 Describing the unit of assessment), and hence valid for assessment using the IUCN RLE criteria.
2. Whether documentation includes an adequate description of the ecosystem type or satisfactory references to published descriptions (Section 4.2 Describing the unit of assessment). Descriptions should include attribution to the IUCN Global Ecosystem Typology at least at level 3 (Keith et al., 2022; https://global-ecosystems.org/), crosswalks to other relevant classifications, a description of characteristic ecosystem properties, representative images, distribution map(s), and an account of key ecological processes and threats with a graphical conceptual model.
3. Whether all accessible data and information relevant to IUCN RLE assessment of the ecosystem type have been addressed.
4. Whether the quality of underlying data has been evaluated and found to be adequate.
5. Whether definitions and concepts in the RLE guidelines have been correctly interpreted and applied, particularly whether indicator selection has been appropriately justified, linked to thresholds of collapse, and correctly applied and calculated.
6. Whether methods and calculations have been validly applied, and whether alternative methods are more suitable.
7. Whether estimates of variables for past, present, future, and collapsed states are complete and supported by evidence.
8. Whether inferences related to the IUCN RLE criteria are justified and transparently communicated.
9. Whether uncertainties have been adequately incorporated into the assessment.

10.2. Timing of publication on the database

Global-scale ecosystem assessments based on the *IUCN Red List of Ecosystems Categories and Criteria* will be published on the IUCN RLE Database. Sub-global assessments may also be published on the IUCN RLE Database, subject to meeting the standards specified in these guidelines, agreement from the data custodian and resources available to the RLE team. The following principles apply to global RLE assessments.

Databasing prioritisation principles:

1. **The conservation imperative.** A primary purpose of RLE assessments is to inform the public about the current status of ecosystems in a timely manner. Consequently, assessments will be made available online, on the IUCN RLE Database, to inform ecosystem conservation and management actions. For this primary reason, IUCN and its RLE partners will strive to publish Red List data on the IUCN RLE Database at the earliest opportunity and work to ensure that publication is not unnecessarily delayed, subject to available resources and capacity in the RLE team, the need to comply with internal consistency checks, and technology constraints.
2. **The value of scientific publishing.** IUCN recognises the value of scientific publications and strongly encourages the production of scientific papers and analyses using Red List data by Red List partner institutions, commission members, and ecosystem scientists involved in the assessment process. However, publication of assessment data on the RLE Database will not usually be delayed until after publication of analyses in the scientific literature unless specifically requested.
3. **Assessments and peer-reviewed publications.** In exceptional circumstances, on recommendation of the RLE Partnership Committee, IUCN may delay publication of assessments on the IUCN RLE Database for a period of no more than six months to enable publication of assessment results (or a summary thereof) in a peer-reviewed scientific journal. Once the six-month period has lapsed, assessments will be processed and published at the next opportunity.
4. **Assessments and publicity.** Publicity and media exposure of the RLE is encouraged. However, publishing assessments on the IUCN RLE Database does not necessitate publicising the results in the media. Media outreach and press releases can be delayed until a peer-reviewed paper is ready for release based on a recommendation by the RLE Partnership Committee. IUCN recognises that a potential advantage of such a delay is the scientific credibility afforded by peer-reviewed publication that may be expounded in publicity and media coverage. It also recognises that a potential disadvantage of such delay is that it may consequently delay urgent conservation action that may otherwise have been triggered by more rapid and widespread communication. In making a recommendation, the RLE Partnership Committee must weigh up the benefits of delaying publicity. In the absence of a recommendation from the Committee, publicity of the assessment results will not normally be delayed beyond the date of publication on the RLE website.

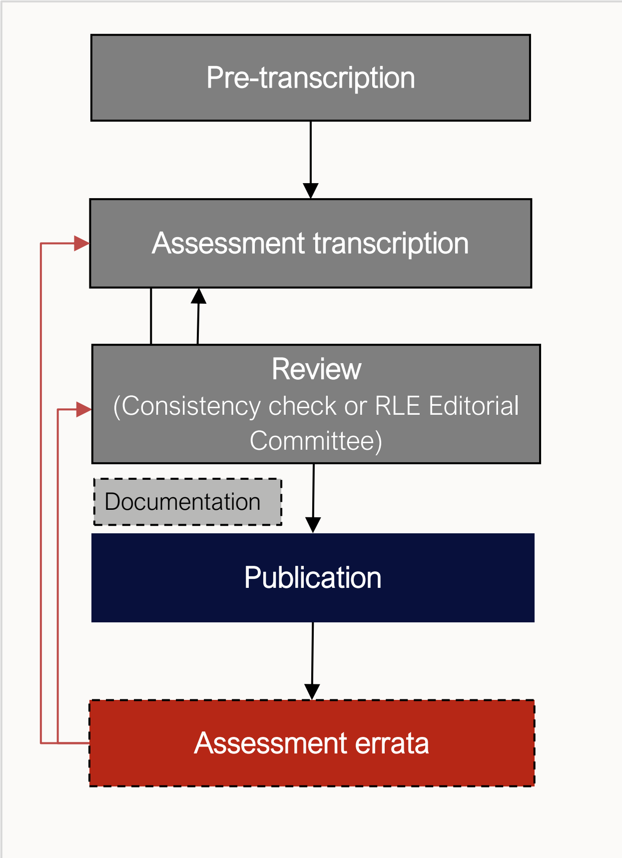


Figure 19. General publication process for the Red List of Ecosystems Database.
Source: Authors.

References

Aarrestad, P.A., Evju, M., Høitomt, T., Ihlen, P. & Grytnes, J.A. (2018). Snow bed, Mountains and rocks. Norwegian red list for nature types 2018. Species data bank, Trondheim. https://artsdatabanken.no/RLN2018/46

Akcakaya, H.R., Ferson, S., Burgman, M.A., Keith, D.A., Mace, G.M. & Todd, C.R. (2000). Making Consistent IUCN Classifications under Uncertainty. *Conservation Biology* 14:1001–1013. https://doi.org/10.1046/j.1523-1739.2000.99125.x

Aladin, N. & Plotnikov, I. (1993). Large saline lakes of former USSR: a summary review. *Hydrobiologia* 267:1–12. https://doi.org/10.1007/978-94-011-2076-0_1

Alaniz, A.J., Galleguillos, M. & Perez-Quezada, J.F. (2016). Assessment of quality of input data used to classify ecosystems according to the IUCN Red List methodology: The case of the central Chile hotspot. *Biological Conservation* 204: 378-385. https://doi.org/10.1016/j.biocon.2016.10.038

Alevizon, W.S. (2010). *Introduction To Caribbean Coral Reefs.* Gainsville, USA: William Alevizon.

Allen, C., Gunderson, L. & Johnson, A.R. (2005). The Use of Discontinuities and Functional Groups to Assess Relative Resilience in Complex Systems. *Ecosystems* 8: 958–966. https://doi.org/10.1007/s10021-005-0147-x

Allen, T.F. & Hoekstra, T.W. (1990). The confusion between scale-defined levels and conventional levels of organization in ecology. *Journal of Vegetation Science* 1:5–12. https://doi.org/10.2307/3236048

Angeler, D.G., Fried-Petersen, H.B., Allen, C.R., Garmestani, A., Twidwell, D., Chuang, W.C., Donovan, V.M., Eason, T., Roberts, C.P., Sundstrom, S.M. & Wonkka, C.L. (2019). Adaptive capacity in ecosystems. *Advances in Ecological Research*, 1st edition. Elsevier Ltd. http://dx.doi.org/10.1016/bs.aecr.2019.02.001

Arias-González, J.E., Acosta-González, G., Membrillo, N., Garza-Pérez, J.R. & Castro-Pérez, J.M. (2012). Predicting spatially explicit coral reef fish abundance, richness and Shannon–Weaver index from habitat characteristics. *Biodiversity and Conservation* 21:115–130. https://doi.org/10.1007/s10531-011-0169-y

Aronson, R.B. & Blake, D.B. (2001). Global climate change and the origin of modern benthic communities in Antarctica. *American Zoologist* 41:27–39. https://doi.org/10.1093/icb/41.1.27

Auld, T.D. & Hutton, I. (2004). Conservation issues for the vascular flora of Lord Howe Island. *Cunninghamia* 8:490–500.

Auld, T.D. & Leishman, M.R. (2015). Ecosystem risk assessment for Gnarled Mossy Cloud Forest, Lord Howe Island, Australia. *Austral Ecology* 40:364–372. https://doi.org/10.1111/aec.12202

Austnes, K., Lund, E., Sample, J. E., Aarrestad, P. A., Bakkestuen, V. & Aas, W. (2018). *Overskridelser av tålegrenser for forsuring og nitrogen for Norge.* Oppdatering med perioden 2012–2016 Rapport M-966|2018. Miljødirektoratet.

Bairos-Novak, K.R., Hoogenboom, M.O., van Oppen, M.J.H. & Connolly, S.R. (2021). Coral adaptation to climate change: Meta-analysis reveals high heritability across multiple traits. *Global Change Biology* 27, 5694-5710. https://doi.org/10.1111/gcb.15829

Bakkestuen, V., Erikstad, L. & Halvorsen, R. (2008). Step-less models for regional biogeoclimatic variation in Norway. *Journal of Biogeography* 35: 1906-1922. https://doi.org/10.1111/j.1365-2699.2008.01941.x

Barrett, S. & Yates, C.J. (2015). Risks to a mountain summit ecosystem with endemic biota in southwestern Australia. *Austral Ecology* 40:423–432. https://doi.org/10.1111/aec.12199

Beck, M.W., Brumbaugh, R.D., Airoldi, L., Carranza, A., Coen, L.D., Crawford, C., Defeo, O., Edgar, G.J., Hancock, B., Kay, M.C., Lenihan, H.S., Luckenbach, M.W., Toropova, C. L., Zhang, G. & Guo, X. (2011). Oyster Reefs at Risk and Recommendations for Conservation, Restoration, and Management. *BioScience* 61:107–116. https://doi.org/10.1525/bio.2011.61.2.5

Bellingham, P.J. (2008). Cyclone effects on Australian rain forests: An overview. *Austral Ecology* 33:580–584. https://doi.org/10.1111/j.1442-9993.2008.01914.x

Benítez-López, A., Santini, L., Schipper, A.M., Busana, M. & Huijbregts, M.A.J. (2019). Intact but empty forests? Patterns of hunting-induced mammal defaunation in the tropics. *PLoS Biology* 17: e3000247. https://doi.org/10.1371/journal.pbio.3000247

Berdanier, A.B. (2010). Global Treeline Position. *Nature Education Knowledge* 3(10):11. https://doi.org/10.14201/eks.7506

Bergstrom, D.M., Wienecke, B.C., van den Hoff, J., Hughes, L., Lindenmayer, D.B., Ainsworth, T.D., Baker, C.M., Bland, L., Bowman, D.M., Brooks, S.T. & Canadell, J.G. (2021). Combating ecosystem collapse from the tropics to the Antarctic. *Global Change Biology* 27: 1692–1703. https://doi.org/10.1111/gcb.15539

Bickford, S. & Mackey, B. (2004). Reconstructing Pre-Impact Vegetation Cover in Modified Landscapes Using Environmental Modelling, Historical Surveys and Remnant Vegetation Data: A Case Study in the Fleurieu Peninsula, South Australia. *Journal of Biogeography* 31: 787–805. https://doi.org/10.1111/j.1365-2699.2003.01050.x

Blab, J., Riecken, U. & Ssymank, A. (1995). Proposal on a criteria system for a National Red Data Book of Biotopes. *Landscape Ecology* 10:41–50. https://doi.org/10.1007/BF00158552

Blasi, C., Zavattero, L., Capotorti, G., Bonacquisti, S., Copiz, R., Del Vico, E. & Facioni, L. (2023). Lista Rossa degli Ecosistemi d'Italia. Comitato Italiano IUCN, Ministero dell'Ambiente e della Sicurezza Energetica.

Bland, L.M., Regan, T.J., Dinh, M.N., Ferrari, R., Keith, D.A., Lester, R., Mouillot, D., Murray, N.J., Nguyen, H.A. & Nicholson, E. (2017). Using multiple lines of evidence to assess the risk of ecosystem collapse. *Proceedings of the Royal Society B: Biological Sciences* 284:20170660. https://doi.org/10.1098/rspb.2017.0660

Bland, L.M., Watermeyer, K.E., Keith, D.A., Nicholson, E., Regan, T.J., Shannon, L.J., Coll, M., Spalding, M., Link, J. & Blanchard. J. (2018). Assessing risks to marine ecosystems with indicators, ecosystem models and experts. *Biological Conservation* 227:19–28. https://doi.org/10.1016/j.biocon.2018.08.019

Bland, L.M., Nicholson, E., Miller, R.M., Andrade, A., Carré, A., Etter, A., Ferrer-Paris, J.R. Herrera, B., Kontula, T., Lindgaard, A., Pliscoff, P., Skowno, A., Valderrábano, M., Zager, I. & Keith, D.A. (2019). Impacts of the IUCN Red List of Ecosystems on conservation policy and practice. *Conservation Letters* 12, e12666. https://doi.org/10.1111/conl.12666

Botts, E.A., Skowno, A., Driver, A., Holness, S., Maze, K., Smith, T., Daniels, F., Desmet, P., Sink, K., Botha, M., Nel, J. & Manuel, J. (2020). More than just a (red) list: Over a decade of using

South Africa's threatened ecosystems in policy and practice. *Biological Conservation* 246: 108559. https://doi.org/10.1016/j.biocon.2020.108559

Briscoe, N.J., Elith, J., Salguero-Gómez, R., Lahoz-Monfort, J.J., Camac, J.S., Giljohann, K.M., Holden, M.H., Hradsky, B.A., Kearney, M.R., McMahon, S.M. & Phillips, B.L. (2019). Forecasting species range dynamics with process-explicit models: matching methods to applications. *Ecology Letters* 22:1940–1956. https://doi.org/10.1111/ele.13348

Brummitt, N.A., Bachman, S.P., Griffiths-Lee, J., Lutz, M., Moat, J.F., Farjon, A., Donaldson, J.S., Hilton-Taylor, C., Meagher, T.R., Albuquerque, S. & Aletrari, E. (2015). Green Plants in the Red: A Baseline Global Assessment for the IUCN Sampled Red List Index for Plants. *PLOS ONE* 10(8):1–22. Public Library of Science. https://doi.org/10.1371/journal.pone.0135152

Burgman, M.A. (2015) *Trusting judgements: how to get the best out of experts*. Cambridge University Press, Cambridge. https://doi.org/10.1017/CBO9781316282472

Burns, E.L., Lindenmayer, D.B., Stein, J., Blanchard, W., McBurney, L., Blair, D. & Banks, S.C. (2015). Ecosystem assessment of mountain ash forest in the Central Highlands of Victoria, south-eastern Australia. *Austral Ecology* 40:386–399. https://doi.org/10.1111/aec.12200

Butchart, S.H.M., Stattersfield, A.J., Bennun, L.A., Shutes, S.M., Akçakaya, H.R., Baillie, J.E.M., Stuart, S.N., Hilton-Taylor,C. & Mace, G.M. (2004). Measuring global trends in the status of biodiversity: Red List Indices for birds. *PLoS Biology* 2(12): e383. https://doi.org/10.1371/journal.pbio.0020383

Butchart, S.H,M. Akçakaya H.R., Chanson, J., Baillie, J.E.,M. Collen, B., Quader, S., Turner, W.R., Amin, R., Stuart, S.N. & Hilton-Taylor, C. (2007). Improvements to the Red List Index. *PLoS ONE* 2(1): e140. https://doi.org/10.1371/journal.pone.0000140

Cabezas, A., Comín, F.A., Beguería, S. & Trabucchi, M. (2009). Hydrologic and landscape changes in the Middle Ebro River (NE Spain): implications for restoration and management. *Hydrology and Earth Systems Science* 13:273–284. https://doi.org/10.5194/hess-13-273-2009

Calcagno, V., Sun, C., Schmitz, O.J. & Loreau, M. (2011). Keystone Predation and Plant Species Coexistence: The Role of Carnivore Hunting Mode. *The American Naturalist* 177(1): E1–E13. https://doi.org/10.1086/657436

Camill, P., Clark, J.S., Camill, P. & Clark, J.S. (2000). Long-term perspectives on lagged ecosystem responses to climate change: Permafrost in boreal peatlands and the grassland/woodland boundary. *Ecosystems* 3:534–544. https://doi.org/10.1007/s100210000047

Campbell, C., Yang, S., Shea, K. & Alber, R. (2012). Topology of plant-pollinator networks that are vulnerable to collapse from species extinction. *Physical Review E* 86: 021924. https://doi.org/10.1103/PhysRevE.86.021924

Capotorti, G., Del Vico, E., Copiz, R., Facioni, L., Zavattero, L., Bonacquisti, S., Paolanti, M. & Blasi, C. (2023). Ecosystems of Italy. Updated mapping and typology for the implementation of national and international biodiversity-related policies. *Plant Biosystems* 157:1248–1258. https://doi.org/10.1080/11263504.2023.2284135

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., Daily, G.C., Loreau, M., Grace, J.B., Larigauderie, A., Srivastava, D.S. & Naeem, S. (2012). Biodiversity loss and its impact on humanity. *Nature* 486:59–67. https://doi.org/10.1038/nature11148

Cardinale, B.J., Matulich, K.L., Hooper, D.U., Byrnes, J.E., Duffy, E., Gamfeldt, L., Balvanera, P., O'Connor, M.I. & Gonzalez, A. (2011). The functional role of producer diversity in ecosystems. *American Journal of Botany* 98:572–592. https://doi.org/10.3732/ajb.1000364

Cardinale, B.J., Wright, J.P., Cadotte, M.W., Carroll, I.T., Hector, A., Srivastava, D.S., Loreau, M. & Weis, J.J. (2007). Impacts of plant diversity on biomass production increase through time because of species complementarity. *Proceedings of the National Academy of Sciences* 104:18123–18128. https://doi.org/10.1073/pnas.0709069104

Carpenter, K.E., Abrar, M., Aeby, G., Aronson, R.B., Banks, S., Bruckner, A., Chiriboga, A., Cortes, J., Delbeek, J.C., DeVantier, L. & Edgar, G.J. (2008). One-third of reef-building corals face elevated extinction risk from climate change and local impacts. *Science* 321:560–564. https://doi.org/10.1126/science.1159196

Carpenter, S.R. (2003). *Regime Shifts in Lake Ecosystems: Patterns and Variation.* Excellence in Ecology Series, Volume 15. Olderdorf/Luhe, Germany: International Ecology Institute.

Carpenter, S.R. and Turner, M.G. (2001). Hares and Tortoises: Interactions of Fast and Slow Variables in Ecosystems. *Ecosystems* 3:495–497. https://doi.org/10.1007/s100210000043

Catling, P.M. & Brownell, V.R. (1995). A review of the alvars of the Great Lakes region: distribution, floristic composition, biogeography and protection. *Canadian Field Naturalist* 109:143–171. https://doi.org/10.5962/p.357608

Convention on Biological Diversity (CBD) (2022). Decision Adopted by the Conference of the Parties to the Convention on Biological Diversity 15/5. Monitoring framework for the Kunming-Montreal Global Biodiversity Framework (CBD/COP/DEC/15/5).

Chapelle, G. & Peck, L.S. (1999). Polar gigantism dictated by oxygen availability. *Nature* 399:114–115. https://doi.org/10.1038/20099

Chase, J.M., Blowes, S.A., Knight, T.M., Gerstner, K. & May, F. (2020). Ecosystem decay exacerbates biodiversity loss with habitat loss. *Nature 584*:238–243. https://doi.org/10.1038/s41586-020-2531-2

Chen, G., Wang,X. & Ma, K-P. (2020). Red list of China's forest ecosystems: A conservation assessment and protected area gap analysis. *Biological Conservation* 248:108636. https://doi.org/10.1016/j.biocon.2020.108636

Christensen, V. & Walters, C.J. (2004). Ecopath with Ecosim: methods, capabilities and limitations. *Ecological modelling* 172:109–139. https://doi.org/10.1016/j.ecolmodel.2003.09.003

Clark, G.F., Raymond, B., Riddle, M.J., Stark, J.S. & Johnston, E.L. (2015). Vulnerability of Antarctic shallow invertebrate-dominated ecosystems. *Austral Ecology* 40:482–491. https://doi.org/10.1111/aec.12237

Clarke, P.J., Latz, P.K. & Albrecht, D.E. (2005). Long-term changes in semi-arid vegetation: Invasion of an exotic perennial grass has larger effects than rainfall variability. *Journal of Vegetation Science* 16:237–248. https://doi.org/10.1111/j.1654-1103.2005.tb02361.x

Commonwealth of Australia (2000). Environment Protection and Biodiversity Conservation Regulations, REG 7.02 Criteria for listing threatened ecological communities. Commonwealth of Australia, Canberra, ACT.

Cochrane, M.A. & Laurance, W.F. (2002). Fire as a large-scale edge effect in Amazonian forests. *Journal of Tropical Ecology* 18:311–325. https://doi.org/10.1017/S0266467402002237

Congalton, R.G. & Green, K. (2008). *Assessing the Accuracy of Remotely Sensed Data: Principles and Practices, 2nd Edition*. Boca Raton, Florida, USA: CRC Press/Taylor & Francis Group. https://doi.org/10.1201/9781420055139

Corell, R.W. (2006). Challenges of Climate Change: An Arctic Perspective. *AMBIO: A Journal of the Human Environment* 35:148–152. https://doi.org/10.1579/0044-7447(2006)35[148:COCCAA]2.0.CO;2

Connors, B.M., Cooper, A.B., Peterman, R.M. & Dulvy, N.K. (2014). The false classification of extinction risk in noisy environments. *Proceedings of the Royal Society B* 281:20132935. https://doi.org/10.1098/rspb.2013.2935

Council of the European Commission (1992). Council directive 92/43/EEC of 21 May 1992 on the conservation of natural habitats and of wild fauna and flora. *Official Journal of the European Communities, Series L* 206:7–49.

Cowling, R.M. & Bond, W.J. (1991). How small can reserves be? An empirical approach in Cape Fynbos, South Africa. *Biological Conservation* 58: 243–56. https://doi.org/10.1016/0006-3207(91)90094-P

Crespin, S.J. & Simonetti, J.A. (2015). Predicting ecosystem collapse: Spatial factors that influence risks to tropical ecosystems. *Austral Ecology* 40:492–501. https://doi.org/10.1111/aec.12209

Dayton, P. (1979). Observations of growth, dispersal and population dynamics of some sponges in McMurdo Sound, Antarctica. Biologie des Spongiaires (Sponge Biology). *Colloques Internationaux du Centre National de la Reserch Scientifique* 291:271–282.

Dayton, P.K. (1990). Polar benthos. In: W.O. Smith (ed.) *Polar oceanography*, pp. 631–685. London: Academic Press. https://doi.org/10.1016/B978-0-08-092595-0.50010-6

den Boer, P.J. (1968). Spreading of risk and stabilization of animal numbers. *Acta Biotheoretica* 18:165–194. https://doi.org/10.1007/BF01556726

de Visser, S.N., Freymann, B.P. and Olff, H. (2011). The Serengeti food web: empirical quantification and analysis of topological changes under increasing human impact. *Journal of Animal Ecology* 80:484–494. https://doi.org/10.1111/j.1365-2656.2010.01787.x

Department if Environment and Conservation (DEC) (2007). *Definitions, categories and criteria for threatened and priority ecological communities.* Perth: Government of Western Australia.

Department of Environment and Climate Change (DECC) (2007). *Lord Howe Island Biodiversity Management Plan.* Sydney: New South Wales.

Delarze R., Eggenberg S., Steiger P., Bergamini A., Fivaz F., Gonseth Y., Guntern J., Hofer G., Sager L. & Stucki P. (2016). *Liste rouge des milieux de Suisse.* Abrégé actualisé du rapport technique 2013 sur mandat de l'Office fédéral de l'environnement (OFEV), Berne.

Dessai, S., Bhave, A., Birch, C., Conway, D., Garcia-Carreras, L., Gosling, J.P., Mittal, N. & Stainforth, D. (2018). Building narratives to characterise uncertainty in regional climate change through expert elicitation. *Environmental Research Letters* 13:074005. https://doi.org/10.1088/1748-9326/aabcdd

Di Fonzo, M., Collen, B. & Mace, G.M. (2013). A new method for identifying rapid decline dynamics in wild vertebrate populations. *Ecology and Evolution* 3:2378–2391. https://doi.org/10.1002/ece3.596

Diamond, J. (2007). Easter Island revisited. *Science* 317:1692–1694. https://doi.org/10.1126/science.1138442

Diamond, J.M. (1989). The present, past and future of human-caused extinctions. *Philosophical Transactions of the Royal Society of London B* 325:469–477. https://doi.org/10.1098/rstb.1989.0100

Dickman C., Wardle, G., Foulkes, J. & de Preu, N. (2014). Desert complex environments. In: D. Lindenmayer, E. Burns, N. Thurgate, & A. Lowe (Eds.) *Biodiversity and Environmental Change: Monitoring, Challenges and Direction* (pp 215–285). CSIRO, Melbourne

Dobrowski, S.Z., Littlefield, C.E., Lyons, D.S., Hollenberg, C., Carroll, C., Parks, S.A., Abatzoglou, J.T., Hegewisch, K. & Gage, J. (2021). Protected-area targets could be undermined by climate change-driven shifts in ecoregions and biomes. *Communications Earth and Environment* 2:1–11. https://doi.org/10.1038/s43247-021-00270-z

Doak, D.F., Marino, P.C. & Kareiva, P.M. (1992). Spatial scale mediates the influence of habitat fragmentation on dispersal success: Implications for conservation. *Theoretical Population Biology* 41:315–336. https://doi.org/10.1016/0040-5809(92)90032-O

Domning, D. (1972). Steller's sea cow and the origin of North Pacific aboriginal whaling. *Syesis* 5:187–189.

Doroff, A.M., Estes, J.A., Tinker, M.T., Burn, D.M. & Evans, T.J. (2003). Sea otter population declines in the Aleutian archipelago. *Journal of Mammalogy* 84:55–64. https://doi.org/10.1644/1545-1542(2003)084<0055:SOPDIT>2.0.CO;2

Driver, A., Sink, K.J., Nel, J.N., Holness, S., Van Niekerk, L., Daniels, F., Jonas, Z., Majiedt, P.A., Harris, L. & Maze, K. (2012). *National Biodiversity Assessment 2011: An assessment of South Africa's biodiversity and ecosystems. Synthesis Report*. Pretoria: South African National Biodiversity Institute and Department of Environmental Affairs.

Dunning, J.B., Danielson, B.J. & Pulliam, H. R. (1992). Ecological processes that affect populations in complex landscapes. *Oikos* 65:169–175. https://doi.org/10.2307/3544901

Earl, J. (2003). *The Distribution and Impacts of Lippia (*Phyla canescens*) in the Murray Darling System.* Guyra: Murray Darling Basin Lippia Working Group.

Edens, B. Maes, J., Hein, L., Carl Obst, C., Siikamaki, J., Schenau, S., Javorsek, M., Chow, J., Ying Chan, J., Steurer, A. & Alfieri, A. (2022). Establishing the SEEA Ecosystem Accounting as a global standard. *Ecosystem Services* 54: 101413. https://doi.org/10.1016/j.ecoser.2022.101413

Eisenhauer, N., Milcu, A., Sabais, A.C.W., Bessler, H., Brenner, J., Engels, C., Klarner, B., Maraun, M., Partsch, S., Roscher, C., Schonert, F., Temperton, V.M., Thomisch, K., Weigelt, A., Weisser, W.W. & Scheu, S. (2011). Plant Diversity Surpasses Plant Functional Groups and Plant Productivity as Driver of Soil Biota in the Long Term. *PLoS ONE* 6: e16055. https://doi.org/10.1371/journal.pone.0016055

Elith, J., Burgman, M.A. & Regan, H.M. (2002). Mapping epistemic uncertainties and vague concepts in predictions of species distribution. *Ecological Modelling* 157:313–329. https://doi.org/10.1016/S0304-3800(02)00202-8

Elith, J., Graham, C.H., Anderson, R.P., Dudík, M., Ferrier, S., Guisan, A., Hijmans, R.J., Huettmann, F., Leathwick, J.R., Lehmann, A., Li, J., Lohmann, L.G., Loiselle, B.A., Manion, G., Moritz, C., Nakamura, M., Nakazawa, Y., Overton, J.McC.M., Peterson, A.T., Phillips, S.J., Richardson, K., Scachetti-Pereira, R., Schapire, R.E., Soberón, J., Williams, S., Wisz, M.S. & Zimmermann, N.E. (2006). Novel methods improve prediction of species' distributions from occurrence data. *Ecography* 29:129-151. https://doi.org/10.1111/j.2006.0906-7590.04596.x

Ene, E. & McGarigal, K. (2023). Fragstats: A Spatial Pattern Analysis Program for Categorical Maps. https://fragstats.org/index.php/documentation

English, V. & Keith, D.A. (2015). Assessing risks to ecosystems within biodiversity hotspots: A case study from southwestern Australia. Austral Ecology 40:411–422. https://doi.org/10.1111/aec.12177

Erikstad, L., Bakkestuen, V., Hanssen, F., Stabbetorp, O.E., Evju, M. & Aarrestad, P.A. (2009). *Evaluering av landsdekkende satelittbasert vegetasjonskart.* NINA Rapport 448: 77.

Essl, F. & Egger, G. (2010). Lebensraumvielfalt in Österreich – Gefährdung und Handlungsbedarf. Zusammenschau der Roten Liste gefährdeter Biotoptypen Österreichs. Klagenfurt: Naturwissenschaftlicher Verein für Kärnten und Umweltbundesamt GmbH.

Estes, J.A., Doak, D., Springer, A. & Williams, T. (2009). Causes and consequences of marine mammal population declines in southwest Alaska: a food-web perspective. *Philosophical Transactions of the Royal Society B* 364:1647–1658. https://doi.org/10.1098/rstb.2008.0231

Estes, J.A., Terborgh, J., Brashares, J.S., Power, M.E., Berger, J., Bond, W.J., Carpenter, S. R., Essington, T.E., Holt, R.D. & Jackson, J.B. (2011). Trophic downgrading of planet Earth. *Science* 333:301–306. https://doi.org/10.1126/science.1205106

Estes, J.A., Tinker, M. & Bodkin, J.L. (2010). Using ecological function to develop recovery criteria for depleted species: sea otters and kelp forests in the Aleutian archipelago. *Conservation Biology* 24:852–860. https://doi.org/10.1111/j.1523-1739.2009.01428.x

Etter, A., Andrade, A., Amaya, P. & Arevalo, P. (2015). *State of the Colombian Ecosystems 2014: an application of the Red List of Ecosystems methodology.* Pontificia Universidad Javeriana and Conservación Internacional-Colombia, Bogotá. https://cm.iucnrle.org/assets/69a94638-cb9e-4358-8745-19ebc2761df1

Etter, A., Andrade, A., Saavedra, K., Amaya, P., & Arévalo, P. (2017). *Risk assessment of Colombian ecosystems: An application of the Red List of Ecosystems methodology (Version 2.0).* Final Report. Pontificia Universidad Javeriana and Conservación Internacional-Colombia, Bogotá.

Etter A, Andrade A, Saavedra K, Amaya P, Cortés, J. and Arévalo P. (2020). *Ecosistemas Colobianos: Amenazas y riesgos.* Una aplicación de la Lista Roj de Ecosistemas a los ecosistemas terrestres continentals. Pontificia Universidad Javierana y Conservación Internacional-Colombia, Bogotá.

Ewers, R.M. & Didham, R.K. (2008). Pervasive impact of large-scale edge effects on a beetle community. *Proceedings of the National Academy of Sciences of the United States of America 105*:5426–5429. https://doi.org/10.1073/pnas.0800460105

Faber-Langendoen, D., Keeler-Wolf, T., Meidinger, D., Tart, D., Hoagland, B., Josse, C., Navarro, G., Ponomarenko, S., Saucier, J.-P., Weakley, A. and Comer, P. (2014). EcoVeg: a new approach to vegetation description and classification. *Ecological Monographs* 84:533–561. https://doi.org/10.1890/13-2334.1

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology Evolution and Systematics* 34: 487–515. https://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fernández, M., Hamilton, H.H. & Kueppers, L.M. (2015). Back to the future: Using historical climate variation to project near-term shifts in habitat suitable for coast redwood. *Global Change Biology* 21:4141–4152. https://doi.org/10.1111/gcb.13027

Ferrer-Paris, J., Zager, I., Keith, D.A., Oliveira-Miranda, M.A., Paul Rodriguez, J., Josse, C., Gonzalez-Gil, M., Miller, R.M., Zambrana-Torrelio, C. & Barrow, E. (2019). An ecosystem risk assessment of temperate and tropical forests of the Americas with an outlook on future conservation strategies. *Conservation Letters* 12: e12623. https://doi.org/10.1111/conl.12623

Ferrer-Paris, J.R. & Keith, D.A. (2024). Trade-offs in the use of direct or indirect indicators of ecosystem degradation for risk assessment. *Ecological Indicators* 160:111790. https://doi.org/10.1016/j.ecolind.2024.111790

Fischer, J., Lindenmayer, D.B., Blomberg, S.P., Montague-Drake, R., Felton, A. & Stein, J.A. (2007). Functional Richness and Relative Resilience of Bird Communities in Regions with Different Land Use Intensities. *Ecosystems* 10:964–974. https://doi.org/10.1007/s10021-007-9071-6

Fletcher, R. J. Jr. (2006). Emergent properties of conspecific attraction in fragmented landscapes. *American Naturalist* 168:207–219. https://doi.org/10.1086/505764

Fletcher, R.J. Jr., Didham, R.K., Banks-Leite, C., Barlow, J., Ewers, R.M., Rosindell, J., Holt, R.D., Gonzalez, A., Pardini, R., Damschen, E.I., Melo, F.P.L., Ries, L., Prevedello, J.A., Tscharntke, T., Laurance, W.F., Lovejoy, T. & Haddad, N.M. (2018). Is habitat fragmentation good for biodiversity? *Biological Conservation 226*:9–15. https://doi.org/10.1016/j.biocon.2018.07.022

Fletcher, R.J., Jr., Betts, M.G., Damschen, E.I., Hefley, T.J., Hightower, J., Smith T.A.H., Fortin, M.J. & Haddad, N.M. (2023). Addressing the problem of scale that emerges with habitat fragmentation. *Global Ecology and Biogeography* 32: 828–841. https://doi.org/10.1111/geb.13658

Foden, W.B., Butchart, S.H.M., Stuart, S.N., Vié, J-C., Akçakaya, H.R., Angulo, A., DeVantier, L.M., Gutsche, A., Turak, E., Cao, L. & Donner, S.D. (2013). Identifying the World's Most Climate Change Vulnerable Species: A Systematic Trait-Based Assessment of all Birds, Amphibians and Corals. *PLoS ONE* 8:e65427. https://doi.org/10.1371/journal.pone.0065427

Foden, W.B., Young, B.E., Akçakaya, H.R., Garcia, R.A., Hoffmann, A.A., Stein, B.A., Thomas, C.D., Wheatley, C.J., Bickford, D., Carr, J.A. & Hole, D.G. (2019). Climate change vulnerability assessment of species. *Climate Change* 10:1–36. https://doi.org/10.1002/wcc.551

Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L. & Holling, C.S. (2004). Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology, Evolution, and Systematics* 35:557–581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711

Fontaine, C., Dajoz, I., Meriguet, J. & Loreau, M. (2005). Functional diversity of plant–pollinator interaction webs enhances the persistence of plant communities. *PLoS Biol*ogy 4(1), e1. https://doi.org/10.1371/journal.pbio.0040001

Fontúrbel, F.E., Nespolo, R.F., Amico, G.C. & Watson, D.M. (2021). Climate change can disrupt ecological interactions in mysterious ways: Using ecological generalists to forecast community-wide effects. *Climate Change Ecology* 2:100044. https://doi.org/10.1016/j.ecochg.2021.100044

Foody, G.M. (2011). Impacts Of Imperfect Reference Data on The Apparent Accuracy Of Species Presence-Absence Models And Their Predictions. *Global Ecology and Biogeography* 20(3):498–508. https://doi.org/10.1111/j.1466-8238.2010.00605.x

Freer, J.J., Partridge, J.C., Tarling, G.A., Collins, M.A. & Genner, M.J. (2018). Predicting ecological responses in a changing ocean: the effects of future climate uncertainty. *Marine Biology* 165:1–18. https://doi.org/10.1007/s00227-017-3239-1

Fuller, R.M., Smith, G.M. & Devereux, B.J. (2003). The characterisation and measurement of land cover change through remote sensing: problems in operational applications?. *International Journal of Applied Earth Observation and Geoinformation* 4:243–253. https://doi.org/10.1016/S0303-2434(03)00004-7

Gann, G.D., McDonald, T., Walder, B., Aronson, J., Nelson, C.R., Jonson, J., Hallett, J.G., Eisenberg, C., Guariguata, M.R., Liu, J., Hua, F., Echeverría, C., Gonzales, E., Shaw, N., Decleer, K. & Dixon, K.W. (2019). International principles and standards for the practice of ecological restoration. Second edition. *Restoration Ecology* 27: S1–S46. https://doi.org/10.1111/rec.13035

Gaston, K.J. (1994). *Rarity.* London, UK: Chapman and Hall. https://doi.org/10.1007/978-94-011-0701-3

Gaston, K.J. & Fuller, R.A. (2007). Commonness, population depletion and conservation biology. *Trends in Ecology and Evolution* 23:14–19. https://doi.org/10.1111/j.1365-2664.2008.01596.x

Gaston, K.J. & Fuller, R.A. (2009). The sizes of species' geographic ranges. *Journal of Applied Ecology* 46:1–9.

Ghoraba, S.M.M., Halmy, M.W.A., Salem, B.B. & Badr, N.B.E. (2019). Assessing risk of collapse of Lake Burullus: a Ramsar site in Egypt using IUCN red list of ecosystems. *Ecological Indicators* 104:172–183. https://doi.org/10.1016/j.ecolind.2019.04.075

Gili, J.-M., Arntz, W.E., Palanques, A., Orejas, C., Clarke, A., Dayton, P.K., Isla, E., Teixidó, N., Rossi, S. & López-González, P.J. (2006). A unique assemblage of epibenthic sessile suspension feeders with archaic features in the high-Antarctic. *Deep Sea Research Part II: Topical Studies in Oceanography* 53:1029–1052. https://doi.org/10.1016/j.dsr2.2005.10.021

Gillies, C.L., Castine, S.A., Alleway, H.K., Crawford, C., Fitzsimons, J.A., Hancock, B., Koch, P., McAfee, D., McLeod, I.M. & Ermgassen, P.S.E. (2020). Conservation status of the Oyster Reef Ecosystem of Southern and Eastern Australia. *Global Ecology and Conservation* 22:e00988. https://doi.org/10.1016/j.gecco.2020.e00988

Giri, C., Ochieng, E., Tieszen, L., Zhu, Z., Singh, A., Loveland, T., Masek, J. & Duke, N. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography* 20:154–159. https://doi.org/10.1111/j.1466-8238.2010.00584.x

Goudard, A. & Loreau, M. (2008). Nontrophic interactions, biodiversity, and ecosystem functioning: an interaction web model. *The American Naturalist* 171:91–106. https://doi.org/10.1086/523945

Government of Manitoba (2014). *The Endangered Species and Ecosystems Act.* https://web2.gov.mb.ca/laws/statutes/ccsm/e111e.php?ccsm=e111.

Grainger, S., Dessai, S., Daron, J., Taylor, A. & Siu, Y.L. (2022). Using expert elicitation to strengthen future regional climate information for climate services. *Climate Services* 26:100278. https://doi.org/10.1016/j.cliser.2021.100278

Grebmeier, J.M., Overland, J.E., Moore, S.E., Farley, E.V., Carmack, E.C., Cooper, L.W., Frey, K.E., Helle, J.H., McLaughlin, F.A. & McNutt, S.L. (2006). A major ecosystem shift in the northern Bering Sea. *Science* 311:1461–1464. https://doi.org/10.1126/science.1121365

Green, P.T., O'Dowd, D.J., Abbott, K.L., Jeffery, M., Retallick, K. & Mac Nally, R. (2011). Invasional meltdown: Invader-invader mutualism facilitates a secondary invasion. *Ecology* 92:1758–1768. https://doi.org/10.1890/11-0050.1

Guo, K. Liu, C-C., Xie, Z-Q., Yonghong Li, F., Franklin, S. B., Lu, Z-J. & Ma, K-P. (2016). China Vegetation Classification: concept, approach and applications. *Phytocoenologia* 48:113–120. https://doi.org/10.1127/phyto/2017/0166

Haddad, N.M., Brudvig, L.A., Clobert, J., Davies, K.F., Gonzalez, A., Holt, R.D., Lovejoy, T.E., Sexton, J.O., Austin, M.P., Collins, C.D., Cook, W.M., Damschen, E.I., Ewers, R.M., Foster, B.L., Jenkins, C.N., King, A.J., Laurance, W.F., Levey, D.J., Margules, C.R., … Townshend, J.R. (2015). Habitat fragmentation and its lasting impact on earth. *Science Advances* 1:e1500052. https://doi.org/10.1126/sciadv.1500052

Hannah, J. & Bell, R.G. (2012). Regional sea level trends in New Zealand. *Journal of Geophysical Research: Oceans* 117:C01004. https://doi.org/10.1029/2011JC007591

Hansen, M.C., Potapov, P.V., Moore, R., Hancher, M., Turubanova, S.A., Tyukavina, A., Thau, D., Stehman, S.V., Goetz, S.J., Loveland, T.R., Kommareddy, A., Egorov, A., Chini, L., Justice, C.O. & Townshend, J.R.G. (2013). High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science* 342:850–853. https://doi.org/10.1126/science.1244693

Hanski, I. (1998). Metapopulation dynamics. *Nature* 396:41–49. https://doi.org/10.1038/23876

Hanski, I. (1999). *Metapopulation ecology*. Oxford University Press. https://doi.org/10.1093/oso/9780198540663.001.0001

Harfoot, M.B.J., Newbold, T., Tittensor, D.P., Emmott, S., Hutton, J., Lyutsarev, V., Smith, M.J., Scharlemann, J.P.W. & Purves, D.W. (2014). Emergent Global Patterns of Ecosystem Structure and Function from a Mechanistic General Ecosystem Model. *PLoS Bio*logy 12:e1001841. https://doi.org/10.1371/journal.pbio.1001841

Harpole, W.S. & Tilman, D. (2007). Grassland species loss resulting from reduced niche dimension. *Nature* 446:791–793. https://doi.org/10.1038/nature05684

Harris, R.M.B., Grose, M.R., Lee, G., Bindoff, N.L., Porfirio, L.L. & Fox-Hughes, P. (2014). Climate projections for ecologists. Wiley Interdisciplinary Reviews: *Climate Change* 5:621–637. https://doi.org/10.1002/wcc.291

Hartley, S. & Kunin, W.E. (2003). Scale dependency of rarity, extinction risk, and conservation priority. *Conservation Biology* 17:1559–1570. https://doi.org/10.1111/j.1523-1739.2003.00015.x

Hawkins, E. & Sutton, R. (2009). The potential to narrow uncertainty in regional climate predictions. *Bulletin of the American Meteorological Society* 90:1095–1107. https://doi.org/10.1175/2009BAMS2607.1

Healy, T., Wang, Y. & Healy, J. (eds.) (2002). *Muddy Coasts of the World: Processes, Deposits, and Function,* Amsterdam: Elsevier Science.

Hemming, V., Burgman, M.A., Hanea, A.M., McBride, M.F. & Wintle, B.C (2018). A practical guide to structured expert elicitation using the IDEA protocol. *Methods in Ecology and Evolution* 9: 169–180. https://doi.org/10.1111/2041-210X.12857

Hermoso, V., Januchowski-Hartley, S.R. & Pressey, R.L. (2013). When the suit does not fit biodiversity: loose surrogates compromise the achievement of conservation goals. *Biological Conservation* 159:197–205. https://doi.org/10.1016/j.biocon.2012.11.026

HilleRisLambers, J., Adler, P.B., Harpole, W.S., Levine, J.M. & Mayfield, M.M. (2012). Rethinking community assembly through the lens of coexistence theory. *Annual Review of Ecology, Evolution and Systematics* 43:227–248. https://doi.org/10.1146/annurev-ecolsys-110411-160411

Hobbs, R.J., Arico, S., Aronson, J., Baron, J.S., Bridgewater, P., Cramer, V.A., Epstein, P.R., Ewel, J.J., Klink, C.A., Lugo, A.E., Norton, D., Ojima, D., Richardson, D.M., Sanderson, E.W., Valladares, F., Vilà, M., Zamora, R. & Zobel, M. (2006). Novel ecosystems: theoretical and management aspects of the new ecological world order. *Global Ecology and Biogeography* 15:1–7. https://doi.org/10.1111/j.1466-822X.2006.00212.x

Holdaway, R.J., Wiser, S.K. & Williams, P.A. (2012). Status assessment of New Zealand's naturally uncommon ecosystems. *Conservation Biology* 26:619–629. https://doi.org/10.1111/j.1523-1739.2012.01868.x

Holdo, R.M., Holt, R.D. & Fryxell, J.M. (2013). Herbivore–vegetation feedbacks can expand the range of savanna persistence: insights from a simple theoretical model. *Oikos* 122:441–453. https://doi.org/10.1111/j.1600-0706.2012.20735.x

Hong, S. & Shin, I. (2010). Global Trends of Sea Ice: Small-Scale Roughness and Refractive Index. *Journal of Climate* 23:4669–4676. https://doi.org/10.1175/2010JCLI3697.1

Horn, A., Krug, C.B., Newton, I.P. & Esler, K.J. (2011). Specific edge effects in highly endangered Swartland Shale Renosterveld in the Cape Region. *Ecologia Mediterranea* 37,63–74. https://doi.org/10.3406/ecmed.2011.1339

Hughes, T.P., Kerry, J., Álvarez-Noriega, M., Álvarez-Romero, J.G., Anderson, K.D., Baird, A.H., Babcock, R.C., Beger, M., Bellwood, D.R., Berkelmans, R., Bridge, T.C., Butler, I.R., Byrne, M., Cantin, N.E., Comeau, S., Connolly, S.R., Cumming, G.S., Dalton, S.J., Diaz-Pulido, G. … Wilson, S.K. (2017). Global warming and recurrent mass bleaching of corals. *Nature* 543:373–377. https://doi.org/10.1038/nature21707

Huth, N. & Possingham, H.P. (2011). Basic ecological theory can inform habitat restoration for woodland birds. *Journal of Applied Ecology* 48:293–300. https://doi.org/10.1111/j.1365-2664.2010.01936.x

International Finance Corporation (IFC) (2019). International Finance Corporation’s Guidance Note 6: Biodiversity Conservation and Sustainable Management of Living Natural Resources. World Bank Group.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (2019). *The global assessment report on biodiversity and ecosystem services Summary for Policymakers.* (Díaz, S. et al., eds.). IPBES Secretariat, Bonn, Germany.

Intergovernmental Panel on Climate Change (IPCC) (2021). Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press:3949. https://www.ipcc.ch/report/ar6/wg1/downloads/report/IPCC_AR6_WGI_Full_Report.pdf.

IPCC (2022). *Climate Change 2022: Impacts Adaptation and Vulnerability Summary for Policymakers.* Sixth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge: Cambridge University Press.

International Union for Conservation of Nature (IUCN) (2001). *IUCN Red List Categories and Criteria: Version 3.1,* Gland and Cambridge: IUCN Species Survival Commission. https://portals.iucn.org/library/node/10315

IUCN (2012). *IUCN Red List Categories and Criteria: Version 3.1. Second edition.* Gland and Cambridge: IUCN. https://portals.iucn.org/library/node/10315

IUCN (2016). *An Introduction to the IUCN Red List of Ecosystems: The Categories and Criteria for Assessing Risks to Ecosystems*. Gland: IUCN. https://portals.iucn.org/library/node/45958

IUCN (2022a). *The IUCN Red List of Threatened Species. Version 2022-2*. www.iucnredlist.org

IUCN (2022b). *Guidelines for using A Global Standard for the Identification of Key Biodiversity Areas. Version 1.2.* Prepared by the KBA Standards and Appeals Committee of the IUCN Species Survival Commission and IUCN World Commission on Protected Areas. Gland: IUCN. https://portals.iucn.org/library/node/49979

IUCN SPC (2024). *Guidelines for Using the IUCN Red List Categories and Criteria.* Version 16. Prepared by the Standards and Petitions Committee. Available from https://www.iucnredlist.org/

Jakeman, A.J., Letcher, R.A. & Norton, J.P. (2006). Ten iterative steps in development and evaluation of environmental models. *Environmental Modelling and Software* 21:602–614. https://doi.org/10.1016/j.envsoft.2006.01.004

Jarvis, A. & Mulligan, M. (2011). The climate of cloud forests. *Hydrological Processes* 25:327–343. https://doi.org/10.1002/hyp.7847

Johansen, B. (2009). Vegetasjonskart for Norge basert pa Landsat TM/ETM+ data. NORUT Rapport 4/2009: 87 s.

Kallimanis, A.S., Kunin, W.E., Halley, J.M. & Sgardelis, S.P. (2005). Metapopulation extinction risk under spatially autocorrelated disturbance. *Conservation Biology* 19:534–546. https://doi.org/10.1111/j.1523-1739.2005.00418.x

Kauffman, A. & Gupta, M.M. (1991). *Introduction to fuzzy arithmetic: theory and application*. New York: VanNostrand Reinhold.

Keith, D.A. (2009). The interpretation, assessment and conservation of ecological communities. *Ecological Management and Restoration* 10:S3–S15. https://doi.org/10.1111/j.1442-8903.2009.00453.x

Keith, D.A. (2015). Assessing and managing risks to ecosystem biodiversity. *Austral Ecology* 40:337–346. https://doi.org/10.1111/aec.12249

Keith, D. & Myerscough, P. (1993). Floristics and soil relations of upland swamp vegetation near Sydney. *Australian Journal of Ecology* 18:325–344. https://doi.org/10.1111/j.1442-9993.1993.tb00460.x

Keith, D.A., Holman, L., Rodoreda, S., Lemmon, J. & Bedward, M. (2007). Plant functional types can predict decade-scale changes in fire-prone vegetation. *Journal of Ecology* 95:1324–1337. https://doi.org/10.1111/j.1365-2745.2007.01302.x

Keith, D.A., Akçakaya, H.R., Thuiller, W., Midgley, G.F., Pearson, R.G., Phillips, S.J., Regan, H.M., Araújo, M.B. & Rebelo, T.G. (2008). Predicting extinction risks under climate change: coupling stochastic population models with dynamic bioclimatic habitat models. *Biology Letters* 4:560–563. https://doi.org/10.1098/rsbl.2008.0049

Keith, D.A., Orscheg, C., Simpson, C.C., Clarke, P.J., Hughes, L., Kennelly, S.J., Major, R.E., Soderquist, T.R., Wilson, A.L. & Bedward, M. (2009). A new approach and case study for estimating extent and rates of habitat loss for ecological communities. *Biological Conservation* 142:1469–1479. https://doi.org/10.1016/j.biocon.2009.02.015

Keith, D.A., Elith, J. & Simpson, C.C. (2014). Predicting distribution changes of a mire ecosystem under future climates. *Diversity and Distributions* 20:440–454. https://doi.org/10.1111/ddi.12173

Keith, D.A., Rodríguez, J.P., Rodríguez-Clark, K.M., Nicholson, E., Aapala, K., Alonso, A., Asmussen, M., Bachman, S., Basset, A., Barrow, E.G., Benson, J.S., Bishop, M.J., Bonifacio, R., Brooks, T.M., Burgman, M.A., Comer, P., Comín, F.A., Essl, F., Faber-Langendoen, D. … Zambrano-Martínez, S. (2013). Scientific Foundations for an IUCN Red List of Ecosystems. *PLoS ONE* 8(5):e62111. https://doi.org/10.1371/journal.pone.0062111

Keith, D.A., Rodríguez, J.P., Brooks, T.M., Burgman, M.A., Barrow, E.G., Bland, L., Comer, P.J., Franklin, J., Link, J., McCarthy, M.A., Miller, R.M., Murray, N.J., Nel, J., Nicholson, E., Oliveira-Miranda, M.A., Regan, T.J., Rodríguez-Clark, K.M., Rouget, M. & Spalding, M.D. (2015). The IUCN Red List of Ecosystems: Motivations, Challenges, and Applications. *Conservation Letters* 8:214–226. https://doi.org/10.1111/conl.12167

Keith, D.A., Akcakaya, H.R. & Murray, N.J. (2018). Scaling range sizes to threats for robust predictions of risks to biodiversity. *Conservation Biology* 32, 322–332. https://doi.org/10.1111/cobi.12988

Keith D.A., Ferrer-Paris, J.R., Nicholson, E. & Kingsford, R.T. (2020). *IUCN Global Ecosystem Typology 2.0 Descriptive profiles for biomes and ecosystem functional groups.* https://doi.org/10.2305/IUCN.CH.2020.13.en

Keith, D.A., Ferrer-Paris, J.R., Nicholson, E., Bishop, M.J. Polidoro, B.A., Ramirez-Llodra, E., Tozer, M. G., Nel, J.L., Mac Nally, R., Gregr, E.J., Watermeyer, K.E., Essl, F., Faber-Langendoen, D., Franklin, J., Lehmann, C.E.R., Etter, A., Roux, D., Stark, J.S., Rowland, J.A. … Kingsford, R.T. (2022) A function-based typology for Earth's ecosystems. *Nature* 610, 513–518. https://doi.org/10.1038/s41586-022-05318-4

Keith, D.A., Ghoraba, S.M.M., Kaly, E., Jones, K.R., Oosthuizen, A., Obura, D., Costa, H.M., Daniels, F., Duarte, E., Grantham, H., Gudka, M., Norman, J., Shannon, L.J., Skowno, A. & Ferrer-Paris, J.R. (2023a). Contributions of Red Lists of Ecosystems to risk-based design and management of protected and conserved areas in Africa. *Conservation Biology* e14169. https://doi.org/10.1111/cobi.14169

Keith, D.A, Benson D.H., Baird, I.R.C., Watts, L., Simpson, C.C., Krogh, M., Gorissen, S., Ferrer-Paris, J.R. & Mason, T.J. (2023b). Effects of interactions between anthropogenic stressors and recurring perturbations on ecosystem resilience and collapse. *Conservation Biology* 37: e13995. https://doi.org/10.1111/cobi.13995

Kemper, J., Cowling, R.M. & Richardson D.M. (1999). Fragmentation of South African renosterveld shrublands: effects on plant community structure and conservation implications. *Biological Conservation* 90:103–111. https://doi.org/10.1016/S0006-3207(99)00021-X

Kingsford, R. & Johnson, W. (1998). Impact of water diversions on colonially-nesting waterbirds in the Macquarie Marshes of arid Australia. *Colonial Waterbirds* 21(2):159–170. https://doi.org/10.2307/1521903

Kingsford, R.T. & Auld, K.M. (2005). Waterbird breeding and environmental flow management in the Macquarie Marshes, arid Australia. *River Research and Applications* 21:187–200. https://doi.org/10.1002/rra.840

Kingsford, R.T. & Thomas, R.F. (1995). The Macquarie Marshes in arid Australia and their waterbirds: a 50-year history of decline. *Environmental Management* 19:867–878. https://doi.org/10.1007/BF02471938

Kirby, M.X. (2004). Fishing down the coast: Historical expansion and collapse of oyster fisheries along continental margins. *Proceedings of the National Academy of Sciences of the United States of America* 101:13096–13099. https://doi.org/10.1073/pnas.0405150101

Kontula, T. & Raunio, A. (2009). New method and criteria for national assessments of threatened habitat types. *Biodiversity and Conservation* 18:3861–3876. https://doi.org/10.1007/s10531-009-9684-5

Kontula, T. & Raunio, A. (2019). *Threatened Habitat Types in Finland 2018 – Red List of Habitats Results and Basis for Assessment*. Finnish Environment Institute and Ministry of the Environment, Helsinki.

Korell, L., Auge, H., Chase, J.M., Harpole ,W.S. & Knight, T.M. (2019). We need more realistic climate change experiments for understanding ecosystems of the future. *Global Change Biology* 26:325-327. https://doi.org/10.1111/gcb.14797

Körner, C. & Paulsen, J. (2004). A world-wide study of high altitude treeline temperatures. *Journal of Biogeography* 31:713–732. https://doi.org/10.1111/j.1365-2699.2003.01043.x

Kujala, H., Moilanen, A., Araújo, M.B. & Cabeza, M. (2013). Conservation planning with uncertain climate change projections. *PloS ONE* 8:e53315. https://doi.org/10.1371/journal.pone.0053315

Kullman, L. & Öberg, L. (2009). Post-Little Ice Age tree line rise and climate warming in the Swedish Scandes: a landscape ecological perspective. *Journal of Ecology* 97: 415–429. https://doi.org/10.1111/j.1365-2745.2009.01488.x

Lasky, J.R. & Keitt, T.H. (2013). Reserve size and fragmentation alter community assembly, diversity, and dynamics. *American Naturalist* 182:E142–E160. https://doi.org/10.1086/673205

Laurance, W.F. (2008). Theory meets reality: How habitat fragmentation research has transcended island biogeographic theory. *Biological Conservation* 141: 1731–1744. https://doi.org/10.1016/j.biocon.2008.05.011

Laurance, W.F., Lovejoy, T.E., Vasconcelos, H.L., Bruna, E.M., Didham, R.K., Stouffer, P.C., Gascon, C., Bierregaard, R.O., Laurance, S.G. & Sampaio, E. (2002). Ecosystem decay of Amazonian forest fragments: A 22-year investigation. *Conservation Biology* 16:605–618. https://doi.org/10.1046/j.1523-1739.2002.01025.x

Lehmann, C.E.R., Anderson, T.M., Sankaran, M., Higgins, S.I., Archibald, S., Hoffmann, W.A., Hanan, N.P., Williams, R.J., Fensham, R.J., Felfili, J., Hutley, L.B., Ratnam, J., San Jose, J., Montes, R., Franklin, D., Russell-Smith, J., Ryan, C.M., Durigan, G., Hiernaux, P., Haidar, R., Bowman, D.M.J.S. & Bond, W.J. (2014). Savanna vegetation-fire-climate relationships differ among continents. *Science* 343:548–552. https://doi.org/10.1126/science.1247355

Lankau, R.A., Zhu, K. & Ordonez, A. (2015). Mycorrhizal strategies of tree species correlate with trailing range edge responses to current and past climate change. *Ecology* 96:1451–1458. https://doi.org/10.1890/14-2419.1

Lester, R.E. & Fairweather, P.G. (2009). Modelling future conditions in the degraded semi-arid estuary of Australia's largest river using ecosystem states. *Estuarine, Coastal and Shelf Science* 85:1–11. https://doi.org/10.1016/j.ecss.2009.04.018

Lester, R.E. & Fairweather, P.G. (2011). Ecosystem states: Creating a data-derived, ecosystem-scale ecological response model that is explicit in space and time. *Ecological Modelling* 222:2690–2703. https://doi.org/10.1016/j.ecolmodel.2011.05.009

Lester, R.E., Close, P.G., Barton, J.L., Pope, A.J. & Brown, S.C. (2014). Predicting the likely response of data-poor ecosystems to climate change using space-for-time substitution across domains. *Global Change Biology* 20:3471–3481. https://doi.org/10.1111/gcb.12634

Lever, J.J, van Nes, E.H., Scheffer, M. & Bascompte, J. (2014). The sudden collapse of pollinator communities. *Ecology Letters* 17:350–359. https://doi.org/10.1111/ele.12236

Likens, G.E. (1992). *The Ecosystem Approach: Its Use and Abuse*. Excellence in Ecology Series, Volume 3. Olderdorf/Luhe: Ecology Institute

Lindenmayer, D. (2009). *Forest pattern and ecological process: a synthesis of 25 years of research.* Melbourne: CSIRO Publishing. https://doi.org/10.1071/9780643098305

Lindgaard, A. & Henriksen, S. (eds.) (2011). *The 2011 Norwegian Red List for Ecosystems and Habitat Types.* Trondheim: Norwegian Biodiversity Information Centre.

Loreau, M., Mouquet, N. & Gonzalez, A. (2003). Biodiversity as spatial insurance in heterogeneous landscapes. *Proceedings of the National Academy of Sciences* 100:12765–12770. https://doi.org/10.1073/pnas.2235465100

Ludwig, J.A., Bastin, G.N., Chewings, V.H., Eager, R.W. & Liedloff, A.C. (2007). Leakiness: A new index for monitoring the health of arid and semiarid landscapes using remotely sensed vegetation cover and elevation data. *Ecological Indicators* 7:442–454. https://doi.org/10.1016/j.ecolind.2006.05.001

Lundberg, J. & Moberg, F. (2003). Mobile Link Organisms and Ecosystem Functioning: Implications for Ecosystem Resilience and Management. *Ecosystems* 6:87–98. https://doi.org/10.1007/s10021-002-0150-4

Mac Nally, R., Cunningham, S.C., Baker, P.J., Horner, G.J. & Thomson, J.R. (2011). Dynamics of Murray-Darling floodplain forests under multiple stressors: The past, present, and future of an Australian icon. *Water Resources Research* 47:W00G05. https://doi.org/10.1029/2011WR010383

MacArthur, R.H. & Wilson, E.O. (1967). *The theory of island biogeography.* Princeton, New Jersey: Princeton University Press.

Mace, G.M., Collar, N.J., Gaston, K.J., Hilton-Taylor, C., Akcakaya, H.R., Leader-Williams, N., Milner-Gulland, E.J. & Stuart, S.N. (2008). Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology* 22:1424–42. https://doi.org/10.1111/j.1523-1739.2008.01044.x

MacKinnon, J., Verkuil, Y.I. & Murray, N.J. (2012). IUCN situation analysis on East and Southeast Asian intertidal habitats, with particular reference to the Yellow Sea (including the Bohai Sea). Occasional Paper of the IUCN Species Survival Commission No. 47. Gland and Cambridge: IUCN.

Mathisen, I.E., Mikheeva, A., Tutubalina, O.V., Aune, S. & Hofgaard, A. (2014). Fifty years of tree line change in the Khibiny Mountains,Russia: advantages of combined remote sensing and dendroe-cological approaches. *Applied Vegetation Science* 17: 6–16. https://doi.org/10.1111/avsc.12038

Martin, T.G., Burgman, M.A., Fidler, F., Kuhnert, P.M., Low-Choy, S., McBride, M. & Mengersen, K. (2012). Eliciting Expert Knowledge in Conservation Science. *Conservation Biology* 26:29–38. https://doi.org/10.1111/j.1523-1739.2011.01806.x

Mason, T.J., Krogh, M., Popovic, G.C., Glamore, W. & Keith, D.A. (2021). Persistent effects of underground longwall coal mining on freshwater wetland hydrology. *Science of the Total Environment* 772: 144772. https://doi.org/10.1016/j.scitotenv.2020.144772

Mastrandrea, M.D., Field, C.B., Stocker, T.F., Edenhofer, O., Ebi, K.L., Frame, D.J., Held, H., Kriegler, E., Mach, K.J. & Matschoss, P.R. (2010). *Guidance Note for Lead Authors of the IPCC Fifth Assessment Report on Consistent Treatment of Uncertainties.* Intergovernmental Panel on Climate Change (IPCC).

Matías, L. & Jump, A.S. (2014). Impacts of predicted climate change on recruitment at the geographical limits of Scots pine. *Journal of Experimental Botany* 65:299–310. https://doi.org/10.1093/jxb/ert376

Maxwell, P.S., Pitt, K.A., Olds, A.D., Rissik, D. & Connolly, R.M. (2015). Identifying habitats at risk: simple models can reveal complex ecosystem dynamics. *Ecological Applications* 25:573–587. https://doi.org/10.1890/14-0395.1

May, F., Rosenbaum, B., Schurr, F.M. & Chase, J.M. (2019). The geometry of habitat fragmentation: Effects of species distribution patterns on extinction risk due to habitat conversion. *Ecology and Evolution* 9:2775–2790. https://doi.org/10.1002/ece3.4951

McBride, M.F., Garnett, S.T., Szabo, J.K., Burbidge, A.H., Butchart, S.H., Christidis, L., Dutson, G., Ford, H.A., Loyn, R.H., Watson, D.M. & Burgman, M.A. (2012). Structured elicitation of expert judgments for threatened species assessment: A case study on a continental scale using email. *Methods in Ecology and Evolution* 3:906–920. https://doi.org/10.1111/j.2041-210X.2012.00221.x

McClintock, J.B., Amsler, C.D., Baker, B.J. & Van Soest, R.W. (2005). Ecology of Antarctic marine sponges: an overview. *Integrative and Comparative Biology* 45:359–368. https://doi.org/10.1093/icb/45.2.359

McKnight, M.W., White, P.S., McDonald, R.I., Lamoreux, J.F., Sechrest, W., Ridgely, R.S. & Stuart, S.N. (2007). Putting beta-diversity on the map: broad-scale congruence and coincidence in the extremes. *PLoS Biology* 5:e272. https://doi.org/10.1371/journal.pbio.0050272

Metternicht, G.I. & Zinck, J.A. (2003). Remote sensing of soil salinity: potentials and constraints. *Remote Sensing of Environment* 85:1–20. https://doi.org/10.1016/S0034-4257(02)00188-8

Meyer, J. (2011). Montane cloud forests on remote islands of Oceania: the example of French Polynesia (South Pacific Ocean). In: L.A. Bruijnzeel, F.N. Scatena & L.S. Hamilton (Eds). *Tropical Montane Cloud Forests: Science for Conservation and Management* (pp. 121–129). Cambridge: Cambridge University Press. https://doi.org/10.1017/CBO9780511778384.012

Micklin, P. & Aladin, N.V. (2008). Reclaiming the Aral Sea. *Scientific American* 298:64–71. https://doi.org/10.1038/scientificamerican0408-64

Midgley, G.F., Davies, I.D., Albert, C.H., Altwegg, R., Hannah, L., Hughes, G.O., O'Halloran, L.R., Seo, C., Thorne, J.H. & Thuiller, W. (2010). BioMove–an integrated platform simulating the dynamic response of species to environmental change. *Ecography* 33:612–616. https://doi.org/10.1111/j.1600-0587.2009.06000.x

Miller, G., Friedel, M., Adam, P. & Chewings, V. (2010). Ecological impacts of buffel grass (Cenchrus ciliaris L.) invasion in central Australia – does field evidence support a fire-invasion feedback? *The Rangeland Journal* 32:353–365. https://doi.org/10.1071/RJ09076

Mladenoff, D.J., Dahir, S.E., Nordheim, E.V., Schulte, L.A. & Guntenspergen, G.G. (2002). Narrowing historical uncertainty: probabilistic classification of ambiguously identified tree species in historical forest survey data. *Ecosystems* 5:539-553

Moberg, F. & Folke, C. (1999). Ecological goods and services of coral reef ecosystems. *Ecological Economics* 29:215–233. https://doi.org/10.1016/S0921-8009(99)00009-9

Moomaw, W.R., Chmura, G.L., Davies, G.T., Finlayson, C.M., Middleton, B.A., Natali, S.M., Perry, J.E., Roulet, N. & Sutton-Grier, A.E. (2018). Wetlands In a Changing Climate: Science, Policy and Management. *Wetlands* 38:183–205. https://doi.org/10.1007/s13157-018-1023-8

Morgan, M.G., Pitelka, L.F. & Shevliakova, E. (2001). Elicitation of expert judgments of climate change impacts on forest ecosystems. *Climatic Change* 49:279–307. https://doi.org/10.1023/A:1010651300697

Mucina, L. & Rutherford, M.C. (eds.) (2006). *The Vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19.* Pretoria: South African National Biodiversity Institute.

Mueller-Dombois, D. & Fosberg, F.R. (1998). *Vegetation of the Tropical Pacific Islands.* New York: Springer-Verlag. https://doi.org/10.1007/978-1-4419-8686-3

Murray, N.J., Clemens, R.S., Phinn, S.R., Possingham, H.P. & Fuller, R.A. (2014). Tracking the rapid loss of tidal wetlands in the Yellow Sea. *Frontiers in Ecology and the Environment* 12:267–272. https://doi.org/10.1890/130260

Murray, N.J. & Fuller, R.A. (2015). Protecting stopover habitat for migratory shorebirds in East Asia. *Journal of Ornithology* 156:217-225. https://doi.org/10.1007/s10336-015-1225-2

Murray, N.J., Keith, D.A., Bland, L.M., Nicholson, E., Regan, T.M., Rodriguez, J.P. & Bedward, M. (2017). The use of range size to assess risks to biodiversity from stochastic threats. *Diversity and Distributions* 23: 474-483. https://doi.org/10.1111/ddi.12533

Murray, N.J. (2017) Global 10 x 10-km grids suitable for use in IUCN Red List of Ecosystems assessments (vector and raster format). https://doi.org/10.6084/m9.figshare.4653439.v1

Murray, N.J., Phinn, S.R., DeWitt, M., Ferrari, R., Johnston, R., Lyons, M.B., Clinton, N., Thau, D. & Fuller, R.A. (2019). The global distribution and trajectory of tidal flats. *Nature* 565: 222–225. https://doi.org/10.1038/s41586-018-0805-8

Murray, N.J., Ma, Z. & Fuller, R.A. (2015). Tidal flats of the Yellow Sea: A review of ecosystem status and anthropogenic threats. *Austral Ecology* 40:472–481. https://doi.org/10.1111/aec.12211

Murray, N.J., Phinn, S.R., Clemens, R.S., Roelfsema, C.M. & Fuller, R.A. (2012). Continental Scale Mapping of Tidal Flats across East Asia Using the Landsat Archive. *Remote Sensing* 4:3417–3426. https://doi.org/10.3390/rs4113417

Murray, N.J., Keith, D.A., Duncan, A., Tizard, R., Ferrer-Paris, J.R., Worthington, T.A., Armstrong, K., Hlaing, N., Htut, W.T., Oo, A.H. & Ya, K.Z. (2020). Myanmar's terrestrial ecosystems: status, threats and conservation opportunities. *Biological Conservation* 252:108834. https://doi.org/10.1016/j.biocon.2020.108834

Newton, A.C. (2010). Use of a Bayesian network for Red Listing under uncertainty. *Environmental Modelling and Software* 25:15–23. https://doi.org/10.1016/j.envsoft.2009.07.016

Nicholson, E., Keith, D.A. & Wilcove, D.S. (2009). Assessing the threat status of ecological communities. *Conservation Biology* 23:259–274. https://doi.org/10.1111/j.1523-1739.2008.01158.x

Nicholson, E., Regan, T.J., Auld, T.D., Burns, E.L., Chisholm, L.A., English, V., Harris, S., Harrison, P., Kingsford, R.T., Leishman, M.R., Metcalfe, D.J., Pisanu, P., Watson, C.J., White, M., White, M.D., Williams, R.J., Wilson, B. & Keith, D.A. (2015). Towards consistency, rigour and compatibility of risk assessments for ecosystems and ecological communities. *Austral Ecology* 40:347–363. https://doi.org/10.1111/aec.12148

Nicholson, E., Watermeyer, K.E., Rowland, J.A., Sato, C.F., Stevenson, S.L., Andrade, A., Brooks, T.M., Burgess, N.D., Cheng, S.T., Grantham, H.S. & Hill, S.L. (2021). Scientific foundations for an ecosystem goal, milestones and indicators for the post-2020 global biodiversity framework. *Nature Ecology and Evolution* 5: 1338–1349. https://doi.org/10.1038/s41559-021-01538-5. https://doi.org/10.1038/s41559-021-01538-5

Nicholson, E., Andrade, A., Brooks, T.M., Driver, A., Ferrer-Paris, J. R., Grantham, H., Gudka, M. S., Keith, D.A., Kontula, T., Lindgaard, A., Londono-Murcia, M.C., Murray, N.J., Raunio, A., Rowland, J.A., Sievers, M., Skowno, A.L., Stevenson, S.L., Valderrabano, M., Vernon, C.M., Zager, I. & Obura, D. (2024). Roles of the Red List of Ecosystems in implementing the Kunming-Montreal Global Biodiversity Framework. *Nature Ecology and Evolution.* https://doi.org/10.1038/s41559-023-02320-5. https://doi.org/10.1038/s41559-023-02320-5

Norwegian Biodiversity Information Centre (NBIC) (2018). Norwegian Red List for ecosystems and habitat types (2018). Trondheim: NBIC. https://www.artsdatabanken.no/rodlistefornaturtyper

NSW Scientific Committee (2004). Coolibah – Black Box Woodland of the northern riverine plains in the Darling Riverine Plains and Brigalow Belt South bioregions – endangered ecological community listing. Final Determination. Sydney: NSW Scientific Committee.

Ntshanga, N.K., Proches, S. & Slingsby J.A. (2021). Assessing the threat of landscape transformation and habitat fragmentation in a global biodiversity hotspot. *Austral Ecology* 46, 1052–1069. https://doi.org/10.1111/aec.13037

Obura, D. Gudka, M., Samoilys, M., Osuka, K, Mbugua, J., Keith, D.A., Porter, S., Roche, R., Hooidonk, R.V., Ahmada, S., Araman, A., Karisa, J., Komakoma, J., Madi, M., Ravinia, I., Razafindrainibe, H., Yahya, S. & Zivane, F. (2022). Vulnerability to collapse of coral reef ecosystems in the Western Indian Ocean. *Nature Sustainability* 5:104–113. https://doi.org/10.1038/s41893-021-00817-0

Olofsson, P., Foody, G.M., Herold, M., Stehman, S.V., Woodcock, C.E. & Wulder, M.A. (2014). Good practices for estimating area and assessing accuracy of land change. *Remote Sensing of Environment* 148:42–57. https://doi.org/10.1016/j.rse.2014.02.015

Olofsson, P., Foody, G.M., Stehman, S.V. & Woodcock, C.E. (2013). Making better use of accuracy data in land change studies: Estimating accuracy and area and quantifying uncertainty using stratified estimation. *Remote Sensing of Environment* 129:122–131. https://doi.org/10.1016/j.rse.2012.10.031

Pacifici, M., Foden, W.B., Visconti, P., Watson, J.E., Butchart, S.H., Kovacs, K.M., Scheffers, B.R., Hole, D.G., Martin, T.G., Akçakaya, H.R. & Corlett, R.T. (2015). Assessing species vulnerability to climate change. *Nature Climate Change* 5:215–225. https://doi.org/10.1038/nclimate2448

Paquette, A. & Hargreaves, A.L. (2021). Biotic interactions are more often important at species' warm versus cool range edges. *Ecology Letters* 24:2427–2438. https://doi.org/10.1111/ele.13864

Pauw, A. (2007). Collapse of a pollination web in small conservation areas. *Ecology* 88:1759–69. https://doi.org/10.1890/06-1383.1

Payet, K., Rouget, M., Esler, K.J., Reyers, B., Rebelo, T., Thompson, M.W. & Vlok, J.H. (2013). Effect of Land Cover and Ecosystem Mapping on Ecosystem-Risk Assessment in the Little Karoo, South Africa. *Conservation Biology* 27:531–541. https://doi.org/10.1111/cobi.12065

Peeters, L.J.M., Holland, K.L., Huddlestone-Holmes, C. & Boulton, A.J. (2022). A spatial causal network approach for multi-stressor risk analysis and mapping for environmental impact assessments. *Science of the Total Environment* 802:149845. https://doi.org/10.1016/j.scitotenv.2021.149845

Pekel, J.F., Cottam, A., Gorelick, N. & Belward, A.S. (2016). High-resolution mapping of global surface water and its long-term changes. *Nature* 540:418-422. https://doi.org/10.1038/nature20584

Peterson, G.D., Cumming, G.S. & Carpenter, S.R. (2003). Scenario planning: a tool for conservation in an uncertain world *Conservation Biology* 17:358–366. https://doi.org/10.1046/j.1523-1739.2003.01491.x

Pickett, S.T.A. (1989). Space-for-time substitution as an alternative to long-term studies. In: Likens, G.E. (ed.) *Long-term Studies in Ecology: Approaches and Alternatives,* pp.110–135*.* New York: Springer-Verlag. https://doi.org/10.1007/978-1-4615-7358-6_5

Pickett, S.T.A. & Cadenasso, M.L. (1995). Landscape ecology: spatial heterogeneity in ecological systems. *Science* 269:331–334. https://doi.org/10.1126/science.269.5222.331

Plagányi, É.E., Punt, A.E., Hillary, R., Morello, E.B., Thébaud, O., Hutton, T., Pillans, R.D., Thorson, J.T., Fulton, E.A. & Smith, A.D. (2014). Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish and Fisheries* 15:1–22. https://doi.org/10.1111/j.1467-2979.2012.00488.x

Pliscoff, P. (2015). Aplicación de los criterios de la Unión Internacional para la Conservación de la Naturaleza (IUCN) para la evaluación de riesgo de los ecosistemas terrestres de Chile. Santiago, Chile: Ministerio de Medio Ambiente.

Portillo, C.A. (2004). Cuantificación del riesgo de extinción de los hábitats terrestres de la Zona Protectora de la Cuenca Baja de los Ríos Socuy, Guasare y Cachirí. Trabajo Especial de Grado. La Universidad del Zulia, Maracaibo. 88+xi pp.

Pounds, J.A., Fogden, M.P.L. & Campbell, J.H. (1999). Biological response to climate change on a tropical mountain. *Nature* 398:611–615. https://doi.org/10.1038/19297

Precautionary Principle Project (2005). *Guidelines for Applying the Precautionary Principle to Biodiversity Conservation and Natural Resource Management.* Cambridge, UK: The Precautionary Principle Project – a joint initiative of Fauna & Flora International, IUCN, ResourceAfrica and TRAFFIC.

Puyravaud, J.P. (2003). Standardizing the calculation of the annual rate of deforestation. *Forest Ecology and Management* 177:593–596. https://doi.org/10.1016/S0378-1127(02)00335-3

Rebelo, A.G., Boucher, C., Helme, N., Mucina, L. & Rutherford, M.C. (2006). Fynbos biome. In: Mucina, L. and Rutherford, M.C. (eds.) *The Vegetation of South Africa, Lesotho and Swaziland. Strelitzia 19*, pp.52–219. Pretoria: South African National Biodiversity Institute.

Regan, H.M. & Colyvan, M. (2000). Fuzzy Sets and Threatened Species Classification. *Conservation Biology* 14:1197–1199. https://doi.org/10.1046/j.1523-1739.2000.99130.x

Regan, H.M., Colyvan, M. & Burgman, M.A. (2002). A taxonomy and treatment of uncertainty for ecology and conservation biology. *Ecological Applications* 12:618–628. https://doi.org/10.1890/1051-0761(2002)012[0618:ATATOU]2.0.CO;2

Regan, T.J., Tolsma, A., Rowland, J., Muir, A., Ferrer-Paris, J.R., Tóth, A.B. & White, M. (2020). *Risk assessment and management priorities for alpine ecosystems under climate change:* Milestone 5 Report. Heidelberg, Victoria.

Reid, M., Ogden, R. & Thoms, M. (2011). The influence of flood frequency, geomorphic setting and grazing on plant communities and plant biomass on a large dryland floodplain. *Journal of Arid Environments* 75:815–826. https://doi.org/10.1016/j.jaridenv.2011.03.014

Reisewitz, S.E., Estes, J.A. & Simenstad, C.A. (2006). Indirect food web interactions: sea otters and kelp forest fishes in the Aleutian archipelago. *Oecologia* 146:623–631. https://doi.org/10.1007/s00442-005-0230-1

Republic of South Africa (2004). National Environmental Management: Biodiversity Act, 2004. *Government Gazette Republic of South Africa Cape Town* 467:1–84

Reschke, C., Reid, R., Jones, J., Feeney, T. & Potter, H. (1999). *Conserving Great Lakes Alvars.* Chicago, USA: The Nature Conservancy.

Riecken, U., Finck, P., Raths, U., Schröder, E. & Ssymank, A. (2009). *German Red Data Book on endangered habitats (short version, July 2009).* Bonn-Bad Godesberg: German Federal Agency for Nature Conservation (BfN).

Riecken, U., Heinzel, K., Schlumprecht, H., Boedeker, D. & für Naturschutz, B. (2006). *Rote Liste der gefährdeten Biotoptypen Deutschlands: zweite fortgeschriebene Fassung 2006.* Bonn: Bundesamt für Naturschutz.

Ries, L., Fletcher, R.J., Battin, J. & Sisk, T.D. (2004). Ecological responses to habitat edges: Mechanisms, models, and variability explained. *Annual Review of Ecology Evolution and Systematics 35*:491–522. https://doi.org/10.1146/annurev.ecolsys.35.112202.130148

Rising, J., Tedesco, M., Piontek, F. & Stainforth, D.A. (2022). The missing risks of climate change. *Nature* 610:643–651. https://doi.org/10.1038/s41586-022-05243-6

Robertson, A.I. & Rowling, R.W. (2000). Effects of livestock on riparian zone vegetation in an Australian dryland river. *Regulated Rivers: Research and Management* 16:527–541. https://doi.org/10.1002/1099-1646(200009/10)16:5<527::AID-RRR602>3.0.CO;2-W

Rodríguez, J.P., Keith, D.A., Rodríguez-Clark, K.M., Murray, N.J., Nicholson, E., Regan, T.J., Miller, R.M., Barrow, E.G., Bland, L.M., Boe, K., Brooks, T.M., Oliveira-Miranda, M.A., Spalding, M. & Wit, P. (2015). A practical guide to the application of the IUCN Red List of Ecosystems criteria. *Philosophical Transactions of the Royal Society B* 370: 20140003. https://doi.org/10.1098/rstb.2014.0003

Rodríguez, J.P., Rodríguez-Clark, K.M., Baillie, J.E.M., Ash, N., Benson, J., Boucher, T., Brown, C., Burgess, N.D., Collen, B., Jennings, M., Keith, D.A., Nicholson, E., Revenga, C., Reyers, B., Rouget, M., Smith, T., Spalding, M., Taber, A., Walpole, M., Zager, I. & Zamin, T. (2011). Establishing IUCN Red List Criteria for Threatened Ecosystems. *Conservation Biology* 25:21–29. https://doi.org/10.1111/j.1523-1739.2010.01598.x

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

Rogers, C.S. (1990). Responses of coral reef s and reef organisms to sedimentation. *Marine Ecology Progress Series* 62:185–202. https://doi.org/10.3354/meps062185

Rowland J.A., Nicholson, E., Ferrer-Paris, J.R., Keith, D.A., Murray, N.J., Sato, C., Toth, A.B., Tolsma, A., Venn, S.E., Asmüssen, M., Pliscoff, P., Zambrana-Torrelio, C., Lester, R.E. & Regan T.J. (2023). Assessing risk of ecosystem collapse in a changing climate. EcoEvoRvix https://doi.org/10.32942/X2HS4Q

Rowland, J.A., Nicholson, E., Murray, N.J., Keith, D.A., Lester, R.E. & Bland, L.M. (2018). Selecting and applying indicators of ecosystem collapse for risk assessments. *Conservation Biology* 32:1233–1245. https://doi.org/10.1111/cobi.13107

Rowland J.A., Bland L.M., Keith D.A., Juffe Bignoli D., Burgman M.A., Etter A., Ferrer-Paris J.R., Miller R.M., Skowno A.L. & Nicholson E. (2020). Ecosystem indices to support global biodiversity conservation. *Conservation Letters* 13: 12680. https://doi.org/10.1111/conl.12680

Ruiz-Labourdette, D., Nogués-Bravo, D, Ollero, H.S., Schmitz, M.F.M.F. & Pineda, F.D. (2012). Forest composition in Mediterranean mountains is projected to shift along the entire elevational gradient under climate change. *Journal of Biogeography* 39:162–176. https://doi.org/10.1111/j.1365-2699.2011.02592.x

Rumpff, L., Duncan, D., Vesk, P., Keith, D. & Wintle, B. (2011). State-and-transition modelling for adaptive management of native woodlands. *Biological Conservation* 144:1224–1236. https://doi.org/10.1016/j.biocon.2010.10.026

Salafsky, N., Salzer, D., Stattersfield, A.J., Hilton-Taylor, C., Neugarten, R., Butchart, S.H.M., Collen, B., Cox, N., Master, L.L., O'Connor, S. & Wilkie, D. (2008). A standard lexicon for biodiversity conservation: Unified classifications of threats and actions. *Conservation Biology* 22:897–911. https://doi.org/10.1111/j.1523-1739.2008.00937.x

Salomaa, A. & Arponen, A. (2023). The role of the Red Lists of Ecosystems in leveraging sustainability changes in Finland – perceptions of the assessors. *Ecosystems and People* 19: 2222185. https://doi.org/10.1080/26395916.2023.2222185

Schaefer, N., Mayer-Pinto, M., Griffin, K.J., Johnston, E.L., Glamore, W. & Dafforn, K.A. (2020). Predicting the impact of sea-level rise on intertidal rocky shores with remote sensing. *Journal of Environmental Management* 261: 110203. https://doi.org/10.1016/j.jenvman.2020.110203

Scheffer, M., Carpenter, S.R., Foley, J., Folke, C. & Walker, B.H. (2001). Catastrophic shifts in ecosystems. *Nature* 413:591–596. https://doi.org/10.1038/35098000

Scheffer, M., Barrett, S., Carpenter, S.R., Folke, C., Green, A.J., Holmgren, M., Hughes, T.P., Kosten, S., Van de Leemput, I.A., Nepstad, D.C. & van Nes, E.H. (2015). Creating a safe operating space for iconic ecosystems: Manage local stressors to promote resilience to global change. *Science* 347:1317–1319. https://doi.org/10.1126/science.aaa3769

Schoepf, V., Stat, M., Falter, J. & McCulloch, M.T. (2015). Limits to the thermal tolerance of corals adapted to a highly fluctuating, naturally extreme temperature environment. *Scientific Reports* 5:17639. https://doi.org/10.1038/srep17639

Scholze, M., Knorr, W., Arnell, N.W. & Prentice, I.C. (2006). A climate-change risk analysis for world ecosystems. *Proceedings of the National Academy of Sciences* 103:13116–13120. https://doi.org/10.1073/pnas.0601816103

Shapiro, A.C., Grantham, H.S., Aguilar-Amuchastegui, N., Murray, N.J., Gond, V., Bonfils, D. & Rickenbach, O. (2021). Forest condition in the Congo Basin for the assessment of ecosystem conservation status. *Ecological Indicators* 122: 107268. https://doi.org/10.1016/j.ecolind.2020.107268

Shi, J., Ma, K., Wang, J., Zhao, J. & He, K. (2010). Vascular plant species richness on wetland remnants is determined by both area and habitat heterogeneity. *Biodiversity and Conservation* 19:1279–1295. https://doi.org/10.1007/s10531-009-9757-5

Shu, A., Duan, G., Rubinato, M., Tian, L., Wang, M. & Wang, S. (2019). An experimental study on mechanisms for sediment transformation due to riverbank collapse. *Water* (Switzerland) 11. MDPI AG. https://doi.org/10.3390/w11030529

Sievers M., Ryan M. Pearson, R. M., Turschwell, M.P., Bishop, M.J., Bland, L., Brown, C.J., Tulloch, V.J.D., Haig, J.A., Olds A.D., Maxwell, P.S. & Connolly, R.M. (2020). Integrating outcomes of IUCN red list of ecosystems assessments for connected coastal wetlands. *Ecological Indicators* 116:106489. https://doi.org/10.1016/j.ecolind.2020.106489

Simmons, C.L., Auld, T.D., Hutton, I., Baker, W. J. & Shapcott, A. (2012). Will climate change, genetic and demographic variation or rat predation pose the greatest risk for persistence of an altitudinally distributed island endemic? *Biology* 1:736–765. https://doi.org/10.3390/biology1030736

Skowno, A.L. & Monyeki, M.S (2021). South Africa's Red List of Terrestrial Ecosystems (RLEs). *Land* 10: 1–14. https://doi.org/10.3390/land10101048

Slingsby, J.A., Moncrieff, G.R., Rogers, A.J. & February E.C. (2020). Altered ignition catchments threaten a hyperdiverse fire-dependent ecosystem. *Global Change Biology* 26:616–628. https://doi.org/10.1111/gcb.14861

Soares-Filho, B.S., Coutinho Cerqueira, G. & Lopes Pennachin, C. (2002). DINAMICA—a stochastic cellular automata model designed to simulate the landscape dynamics in an Amazonian colonization frontier. *Ecological Modelling* 154:217–235. https://doi.org/10.1016/S0304-3800(02)00059-5

Spalding, M.D., Fox, H.E., Halpern, B.S., McManus, M.A., Molnar, J., Allen, G.R., Davidson, N., Jorge, Z.A., Lombana, A.L., Lourie, S.A., Martin, K.D., McManus, E., Recchia, C.A. & Robertson, J. (2007). Marine ecoregions of the world: A bioregionalization of coastal and shelf areas. *BioScience* 57:573–583. https://doi.org/10.1641/B570707

Speirs-Bridge, A., Fidler, F., McBride, M., Flander, L., Cumming, G. & Burgman, M. (2010). Reducing overconfidence in the interval judgments of experts. *Risk Analysis* 30: 512–523. https://doi.org/10.1111/j.1539-6924.2009.01337.x

Springer, A.M., Estes, J.A., van Vliet, G.B., Williams, T.M., Doak, D.F., Danner, E.M., Forney, K.A. & Pfister, B. (2003). Sequential megafaunal collapse in the North Pacific Ocean: An ongoing legacy of industrial whaling?. *PNAS* 100:12223–12228. https://doi.org/10.1073/pnas.1635156100

Steinbauer, M.J., Grytnes, J-A., Jurasinski, G., Kulonen, A., Lenoir, J., Pauli, H., Rixen, C., Winkler, M., Bardy-Durchhalter, M., Barni, E., Bjorkman, A.D., Breiner, F.T., Burg, S., Czortek, P., Dawes, M.A., Delimat, A., Dullinger, S., Erschbamer, B., Felde, V.A., … Wipf, S. (2018). Accelerated increase in plant species richness on mountain summits is linked to warming. *Nature* 556: 231-234. https://doi.org/10.1038/s41586-018-0005-6

Steneck, R. & Watling, L. (1982). Feeding capabilities and limitation of herbivorous molluscs: a functional group approach. *Marine Biology* 68:299–319. https://doi.org/10.1007/BF00409596

Steneck, R.S., Graham, M.H., Bourque, B.J., Corbett, D., Erlandson, J.M., Estes, J.A. & Tegner, M.J. (2002). Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation* 29:436–459. https://doi.org/10.1017/S0376892902000322

Swenson, J.J. & Franklin, J. (2000). The effects of future urban development on habitat fragmentation in the Santa Monica Mountains. *Landscape Ecology* 15:713–730. https://doi.org/10.1023/A:1008153522122

Tansley, A.G. (1935). The use and abuse of vegetational concepts and terms. *Ecology* 16:284–307. https://doi.org/10.2307/1930070

Taylor, B. & Ganf, G.G. (2005). Comparative ecology of two co-occurring floodplain plants: the native Sporobolus mitchellii and the exotic Phyla canescens. *Marine and Freshwater Research* 56:431–440. https://doi.org/10.1071/MF04196

Taylor, B.L. & Gerrodette, T. (1993). The uses of statistical power in conservation biology: The Vaquita and Northern Spotted Owl. *Conservation Biology* 7: 489-500. https://doi.org/10.1046/j.1523-1739.1993.07030489.x

Thébault, E. & Loreau, M. (2005). Trophic Interactions and the Relationship between Species Diversity and Ecosystem Stability. *The American Naturalist* 166:E95–E114. https://doi.org/10.1086/444403

Thompson, J.N. (1997). Conserving Interaction Biodiversity. In: Pickett, S.T.A., Ostfeld, R.S., Shachak, M. & Likens, G.E. (eds.) *The Ecological Basis of Conservation,* pp.285–293. Springer US. https://doi.org/10.1007/978-1-4615-6003-6_28

Thompson, P.L. & Fronhofer, E.A. (2019). The conflict between adaptation and dispersal for maintaining biodiversity in changing environments. *Proceedings of the National Academy of Sciences of the United States of America* 116:21061–21067. https://doi.org/10.1073/pnas.1911796116

Thoms, M. & Sheldon, F. (2000). Water resource development and hydrological change in a large dryland river: the Barwon–Darling River, Australia. *Journal of Hydrology* 228:10–21. https://doi.org/10.1016/S0022-1694(99)00191-2

Thoms, M.C. (2003). Floodplain–river ecosystems: lateral connections and the implications of human interference. *Geomorphology,* 56, 335–349. https://doi.org/10.1016/S0169-555X(03)00160-0

Tilman, D., Lehman, C.L. & Yin, C.J. (1997). Habitat destruction, dispersal, and deterministic extinction in competitive communities. *American Naturalist* 149:407–435. https://doi.org/10.1086/285998

Titeux, N., Henle, K., Mihoub, J.B. & Brotons, L. (2016). Climate change distracts us from other threats to biodiversity. *Frontiers in Ecology and the Environment* 14:291. https://doi.org/10.1002/fee.1303

Tonmoy, F.N., El-Zein, A. & Hinkel, J. (2014). Assessment of vulnerability to climate change using indicators: A meta-analysis of the literature. *Climate Change* 5:775–792. https://doi.org/10.1002/wcc.314

Turner II, B. L. & Sabloff, J. A. (2012). Classic Period collapse of the Central Maya Lowlands: Insights about human–environment relationships for sustainability. *Proceedings of the National Academy of Sciences of the United States of America* 109: 13908–13914. https://doi.org/10.1073/pnas.1210106109

United Nations Environment Programme (UNEP) (2001). Partow, H. *The Mesopotamian marshlands: demise of an ecosystem*. Early Warning and Assessment Technical Report, UNEP/DEWA/TR.01-3 Rev. 1. Division of Early Warning and Assessment. Nairobi: UNEP.

United Nations Committee of Experts on Environmental-Economic Accounting (UNCEEA) (2021). System of Environmental-Economic Accounting—Ecosystem Accounting. New York: UNCEEA, United Nations Statistical Division.

Valavi, R., Guillera-Arroita, G., Lahoz-Monfort, J.J., & Elith, J. (2022). Predictive performance of presence-only species distribution models: a benchmark study with reproducible code. *Ecological Monographs* 92: e01486. https://doi.org/10.1002/ecm.1486

Varian, H.R. (2005). Bootstrap Tutorial. *The Mathematica Journal* 9:768-775.

Vázquez, D.P. & Simberloff, D. (2003). Changes in interaction biodiversity induced by an introduced ungulate. *Ecology Letters* 6:1077–1083. https://doi.org/10.1046/j.1461-0248.2003.00534.x

Walker, B., Kinzig, A. & Langridge, J. (1999). Plant Attribute Diversity, Resilience, and Ecosystem Function: The Nature and Significance of Dominant and Minor Species. *Ecosystems* 2:95–113. https://doi.org/10.1046/j.1461-0248.2003.00534.x

Walther, G.R. (2010). Community and ecosystem responses to recent climate change. *Philosophical Transactions of the Royal Society B: Biological Sciences* 365:2019–2024. https://doi.org/10.1098/rstb.2010.0021

Wang, Y. & Zhu, D. (1994). Tidal flats in China. *Oceanology of China Seas* 2:445–456. https://doi.org/10.1098/rstb.2010.0021

Wang, Y.P., Gao, S., Jia, J., Thompson, C.E.L., Gao, J. & Yang, Y. (2012). Sediment transport over an accretional intertidal flat with influences of reclamation, Jiangsu coast, China. *Marine Geology* 291–294:147–161. https://doi.org/10.1016/j.margeo.2011.01.004

Watling, L. & Norse, E.A. (1998). Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology* 12:1180–1197. https://doi.org/10.1046/j.1523-1739.1998.0120061180.x

Wethey, D.S., Woodin, S.A., Hilbish, T.J., Jones, S.J., Lima, F.P. & Brannock, P.M. (2011). Response of intertidal populations to climate: Effects of extreme events versus long term change. *Journal of Experimental Marine Biology and Ecology* 400:132–144. https://doi.org/10.1016/j.jembe.2011.02.008

Wiens, J.A. (1989). Spatial scaling in ecology. *Functional Ecology* 3:385–397. https://doi.org/10.2307/2389612

Wieser, G. & Tausz, M. (2007). *Trees at their upper limit: Treelife limitation at the alpine timberline.* Springer, Dordrecht. https://doi.org/10.1007/1-4020-5074-7

Williams, J.W. & Jackson, S.T. (2007). Novel climates, no-analog communities, and ecological surprises. *Frontiers in Ecology and the Environment* 5:475–482. https://doi.org/10.1890/070037

Williams, R.J., Wahren, C.H., Stott, K.A.J., Camac, J.S., White, M., Burns, E., Harris, S., Nash, M., Morgan, J.W., Venn, S. & Papst, W.A. (2015). An International Union for the Conservation of Nature Red List ecosystems risk assessment for alpine snow patch herbfields, South-Eastern Australia. *Austral Ecology* 40:433–443. https://doi.org/10.1111/aec.12266

Wood, J., Low, A.B., Donaldson, J.S. & Rebelo, A.G. (1994). Threats to plant species diversity through urbanization and habitat fragmentation in the Cape Metropolitan Area, South Africa. In: Huntley, B.J. (ed.) *Botanical Diversity in Southern Africa. Strelitzia 1*, pp.259–274. Pretoria: South African National Biodiversity Institute.

Xiao, H., Driver, A., Etter, A., Keith, D.A., Obst, C., Traurig, M.J. & Nicholson. E. (2022). Synergies and complementarities between ecosystem risk assessment and ecosystem accounting. Preprint. *EcoEvoRxiv*. https://doi.org/10.32942/OSF.IO/Y86AR

Yates, C.J. & Ladd, P.G. (2010). Using population viability analysis to predict the effect of fire on the extinction risk of an endangered shrub *Verticordia fimbrilepis* subsp. *fimbrilepis* in a fragmented landscape. *Plant Ecology* 211:305–319. https://doi.org/10.1007/s11258-010-9791-0

Zhao, H.L., Zhao, X.Y., Zhou, R.L., Zhang, T.H. & Drake, S. (2005). Desertification processes due to heavy grazing in sandy rangeland, Inner Mongolia. *Journal of Arid Environments* 62:309–319. https://doi.org/10.1016/j.jaridenv.2004.11.009

Appendix 1. IUCN Red List of Ecosystems Criteria, Version 2.1

A. Reduction in geographic distribution over **ANY** of the following time periods:

	A1. the past 50 years	**A2a.** the next 50 years	**A2b.** any 50-year period including the past, present and future	**A3.** since 1750
CR	≥ 80%	≥ 80%	≥ 80%	≥ 90%
EN	≥ 50%	≥ 50%	≥ 50%	≥ 70%
VU	≥ 30%	≥ 30%	≥ 30%	≥ 50%

B. Restricted geographic distribution indicated by **ANY OF** B1, B2 or B3:

B1. Extent of a minimum convex polygon enclosing all occurrences (extent of occurrence, EOO) is no larger than:

CR	2,000 km^2	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at 1 threat-defined location.
EN	20,000 km^2	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; OR ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at ≤ 5 threat-defined locations.
VU	50,000 km^2	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at ≤ 10 threat-defined locations.

B2. The number of 10 × 10 km grid cells occupied (area of occupancy, AOO) is no more than:

CR	2	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at 1 threat-defined location.
EN	20	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at ≤ 5 threat-defined locations.
VU	50	AND at least one of the following (a–c):	a) An observed or inferred continuing decline in **ANY OF**: i. a measure of spatial extent appropriate to the ecosystem; **OR** ii. a measure of environmental quality appropriate to characteristic biota of the ecosystem; **OR** iii. a measure of biotic interactions* appropriate to the characteristic biota of the ecosystem. b) Observed or inferred threatening processes that are likely to cause continuing declines in geographic distribution, environmental quality or biotic interactions within the next 20 years. c) Ecosystem exists at ≤ 10 threat-defined locations.

B3. The number of threat-defined locations is:

VU	Very small (generally fewer than 5 threat-defined locations) **AND** prone to the effects of human activities or stochastic events within a very short time period in an uncertain future, and thus capable of Collapse or becoming Critically Endangered within a very short time period (B3 can only lead to a listing as VU).

** Note: The full text of clauses B1(a)iii and B2(a)ii of the criteria, states "a measure of disruption to biotic interactions appropriate to…" (IUCN 2016). Alternative phrasing given here avoids potential misinterpretation associated with double-negative expression when that phrase is read in context with clause B1(a) or B2(a), respectively, "continuing decline" in "... a measure of [disruption to] biotic interactions".*

C. **Environmental degradation** over **ANY** of the following time periods:

Criterion		Relative severity (%)		
C1. The past 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
C2a. The next 50 years, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** **C2b.** Any 50-year period including the past, present and future, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:		Relative severity (%)		
	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
C3. Since 1750, based on change in an abiotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:		Relative severity (%)		
	Extent (%)	≥ 90	≥ 70	≥ 50
	≥ 90	CR	EN	VU
	≥ 70	EN	VU	
	≥ 50	VU		

D. **Disruption of biotic processes or interactions** over **ANY** of the following time periods:

Criterion		Relative severity (%)		
D1. The past 50 years, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
D2a. The next 50 years, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table; **OR** **D2b.** Any 50-year period including the past, present and future, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:		Relative severity (%)		
	Extent (%)	≥ 80	≥ 50	≥ 30
	≥ 80	CR	EN	VU
	≥ 50	EN	VU	
	≥ 30	VU		
D3. Since 1750, based on change in a biotic variable affecting a fraction of the extent of the ecosystem and with relative severity, as indicated by the following table:		Relative severity (%)		
	Extent (%)	≥ 90	≥ 70	≥ 50
	≥ 90	CR	EN	VU
	≥ 70	EN	VU	
	≥ 50	VU		

E. **Quantitative analysis that estimates the probability of ecosystem collapse to be:**

CR	≥ 50% within 50 years
EN	≥ 20% within 50 years
VU	≥ 10% within 100 years

Appendix 2. Colour codes

Category	Color	RGB color code		
		R	G	B
CO	Black	0	0	0
CR	Red	255	0	0
EN	Orange	255	165	0
VU	Yellow	255	255	0
NT	Green-Yellow	173	255	47
LC	Green	0	128	0
DD	Grey	128	128	128
NE	White	255	255	255

Appendix 3. Revision history

Version & date		*Revisions*	*Section*
1.0	27/10/2016	Guidelines v1.0 launched	
1.1	16/06/2017	Updated guidelines to include new research published between v1.0 and v1.1	Throughout
		New guidance on making the most of quantitative data and expert knowledge	3.3.4
		New Fig. 8 and Fig. 9	4.4
		Incorporated recent research on performance of range size measures	5.2.3
		Change of language from 'location' to 'threat-defined location' to avoid ambiguity with locality	Glossary
2.0	12/07/2024	Additional terms and edits to definitions	Glossary
		Addition of tier 4 headings to assist navigation	Throughout & Table of contents
		Updates to development of RLE	1.2
		Updates to ecosystem classification and typologies, including the IUCN Global Ecosystem Typology	3.1.1, 4.2.1, Table 1, Table 7
		Additional guidance on the assessment of semi-natural and anthropogenic ecosystem types	3.1.2
		Updates to guidance on the thematic, spatial and temporal scales of RLE assessments	3.1.3
		Clarifications to guidance on diagnosing ecosystem collapse	3.2
		Additional guidance on the interpretation of historical declines in ecosystem properties	3.3.1, 5.3.1.2
		Additional guidance on structured elicitation and the use of expert judgement	3.3.4
		Additional guidance on subglobal RLE assessments (including national Red Lists)	4.1.1
		Additional guidance on use of Not Evaluated, Data Deficient, and Least Concern Categories	4.4
		Additional guidance on weight of evidence for interpretation of bounded estimates spanning two or more Red List categories	4.4.1
		Additional theoretical background on criterion B	6.1
		Clarification of grammatical expression for subcriteria B1(a)iii and B2(a)iii	6.2
		Minor clarifications to guidance for application of spatial metrics in criterion B	6.3.2
		Additional guidance on distinguishing continuing declines from fluctuations	6.3.2
		Additional guidance on interpreting threatening processes in subcriteria B1(b) and B2(b)	6.3.2
		Additional guidance and a new figure for interpreting the number of threat-defined locations in subcriteria B1(c), B2(c) and B3.	6.3.2, Figure 13, Box 13

Version & date	*Revisions*	*Section*
	Guidance for criteria C and D has been integrated into one major section to avoid repetition and excessive cross-referencing	7
	Elaborated theoretical context on criteria C & D	7.1
	New guidance on the selection of indicators for criteria C and D	7.3.1, Table 11
	New guidance on assessing multiple indicators for criterion C or D	7.3.2, Box 15
	New guidance on use of correlative habitat suitability models for assessing subcriteria D2	7.3.3
	New guidance on calculating relative severity and extent of degradation under criteria C and D	7.3.4, 7.3.5, Box 17
	New guidance for assessing risks related to climate change climate under subcriteria A2, C2 and D2 and criterion E	9.1, Box 10
	New guidance for assessing risks related to ecosystem fragmentation under criterion D	9.2
	Additional guidance on processes for databasing, peer review and publication	10
	New appendix on Frequently asked questions with preliminary content for future updates	Appendix 4
	Updated literature to incorporate new research	Throughout & 11

Appendix 4. Frequently Asked Questions

This rubric is continually updated on the IUCN Red List of Ecosystems (RLE) website as new queries emerge (www.iucnrle.org). Please consult the website for the most recent version.

1. **How do I construct descriptions and maps of historical reference states of ecosystem types within a study area?**

   Answer: Historic reference states refer to ecosystem properties (biotic and abiotic features, ecological processes and geographic distribution) that are largely unaffected by intensive or broad-scale human activity. For practical purposes, the year 1750 marks a reference date at the beginning of the industrial era when broad-scale exploitation of ecosystems began to accelerate markedly, noting that major ecosystem transformations had occurred in some areas at earlier times. The properties of historical reference states may be inferred from models based on environmental relationships or evidence on relictual present-day occurrences and distributions and/or historical information, including that available in historical documents, surveys, maps, oral histories and expert elicitation. Properties of reference states exhibit natural variability in space and time that should be considered in description and analysis. Section 5.3.1 Data requirements (subheading 'Historical distributions') provides further guidance.

2. **All ecosystem types within the study area met criterion B for Critically Endangered status. Can I adjust the thresholds within the criteria to get a better spread of ecosystem types among the categories?**

   Answer: No, the thresholds cannot be adjusted. Assessments based on adjusted thresholds do not meet the global standard for ecosystem risk assessment as endorsed by IUCN Council. A single set of thresholds for all ecosystem types is essential to ensure consistency between different areas of assessment and over time.

   Thresholds of the criteria cannot be retro-fitted to any particular area of assessment. There are many problems and consequences associated with such arbitrary adjustments, including:

   a) Departure from the principle that conservation risks within a given area increases as the size of the area is reduced.
   b) Failure to maintain global consistency of assessment outcomes and meanings of threatened categories (e.g. 'Endangered' does not mean the same everywhere).
   c) If taken to extreme, such adjustments could result in every assessment area having its own idiosyncratic set of thresholds.
   d) Divergence of methods between the IUCN RLE and the IUCN Red List of Threatened Species protocols.

   These types of outcomes from RLE assessments may occur because the resulting threat categories more strongly reflect the small area of assessment, rather than the properties of individual ecosystem types (see Section 3.1.3 The influence of scale on assessment outcomes). The risks are real because the area of assessment is small. Just as species extirpation from a small portion of their range is more likely than extinction across their entire range, the same is true of ecosystem collapse. Such outcomes, on their own, may not be informative in setting priorities for ecosystem management, but they are not incorrect and do not require adjustments to the criteria or thresholds.

Even in small areas of assessment where threat categories are apparently uninformative, however, a key motivation for assessing risks to ecosystems is to set priorities for ecosystem protection, regulation, management and restoration (e.g. Etter et al., 2020). If the purpose of an RLE assessment is to set priorities, then assessors may:

a) Use the raw indicator estimates (e.g. rate of decline, area of occupancy, etc.) to obtain finer resolution rankings of ecosystem types than is possible from ranking by the threat categories alone.
b) Address the full context of priority setting for ecosystem management, protection, regulation or restoration by considering socio-economic values, feasibility, cost, in addition to the risk of ecosystem collapse (Miller et al., 2006).

See Section 4.1 Area of assessment for further guidance and commentary on the area of assessment.

3. Which maps best support a Red List of Ecosystems assessment?

Answer: Ecosystem maps most suitable for Red List assessments should:

a) Be based on ecosystem classification units that are faithful to the ecosystem concept (Section 3.1 Ecosystem types: the units of assessment), supporting information (descriptions of map units), spatial resolution, temporal specifications.
b) Directly represent the distribution of ecosystem types based on direct mapping of diagnostic properties (e.g. via remote sensing and/or modelling), rather than indirect proxies (Keith et al., 2022). For example, alpine ecosystems are best delineated by mapping the tree line, rather than using an altitudinal threshold as a proxy.
c) Accurately represent ecosystem distributions, preferably supported by a rigorous, quantitative accuracy assessment.
d) Be well-documented with metadata.
e) Within a suitable range of spatial resolution for Red List assessment (e.g. 1 km^2 pixels or smaller) and commensurate with requirements for accuracy, noting that excessively fine spatial resolution can introduce noise resulting in lower accuracy than a coarser resolution map (Section 3.1.3 The influence of scale on assessment outcomes & Section 4.2.2 Spatial distribution).
f) Represent ecosystem distributions at time frames relevant to the Red List criteria (i.e. present day, 50 years in the past, the time of industrialisation, or 50 years into the future – see Section 3.3.1 Time frames).
g) Cover the full extent of the area of assessment (Section 4.1 Area of assessment).

4. Is ecosystem collapse reversible?

Answer: In theory, if all the key components of an ecosystem still exist, they could re-assemble or be re-assembled into a system that conforms with the description of the ecosystem in its reference state (Section 4.2 Describing the unit of assessment), effectively reversing its collapse. In practice, this rarely occurs, either because: (i) some components of the reference state no longer exist; (ii) threatening processes that drove ecosystem collapse are still operating; (iii) there are ecological barriers, homeostasis or long lags that impede reversal of collapse; (iv) restoration actions are technically infeasible or prohibitively expensive; or (v) ongoing intervention is essential, yet rarely sustained to prevent the system converging on a novel equilibrium that differs from the reference state. In general, restoration of an ecosystem to its reference state is more feasible before major degradation occurs along a pathway to collapse (see Gann et al., 2019).

5. Can I use a general index of ecosystem condition for a combined assessment of criteria C and D, given that abiotic and biotic causes of degradation interact?

   Answer: No. Abiotic and biotic mechanisms of ecosystem degradation must be assessed separately based on different symptoms of degradation, and hence the indicator(s) used to measure them (i.e. measures of abiotic symptoms are assessed under the criterion C and measures of biotic symptoms are assessed under criterion D). Any interactions between abiotic and biotic processes must be understood and assessed in a manner that represent risks from interacting degradation processes in an ecologically and mathematically valid manner. Valid approaches to assessing known interactions between two or more indicators include the use of combinatorial or rules, incorporation of abiotic and biotic predictors into habitat suitability models, and incorporation of abiotic and biotic variables into a simulation model of ecosystem dynamics (see Section 7.3 Applying criteria C and D & Section 8.3 Applying criterion E).

   General indices of ecosystem condition are not recommended for RLE assessments because they: fail to take into account ecosystem-specific mechanisms of threat; are based on arbitrary, very indirect or poorly justified selections of component measures; are based on methods of aggregation that fail to take into account interactions and dependencies among component measures; and/or are not based on an ecologically valid justification for weighting (equal or otherwise) the index components (see Section 7.3 Applying criteria C and D 'Aggregated indices of ecosystem integrity or condition').

6. Can I use patch size as an indicator to assess fragmentation effects under criterion D?

   Answer: Yes, but only by following the guidelines to estimate the change in patch size over the assessment time frame(s) (Section 9.2 Fragmentation). Assessors should consider other spatial metrics that may represent ecological responses to fragmentation more directly or more sensitively (Table 16; Table 17).

INTERNATIONAL UNION FOR
CONSERVATION OF NATURE

WORLD HEADQUARTERS
Rue Mauverney 28
1196 Gland, Switzerland
Tel +41 22 999 0000
Fax +41 22 999 0002
www.iucn.org
www.iucnrle.org
www.iucn.org/resources/publications