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Assessing and managing risks to ecosystem biodiversity

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Abstract Ecosystem conservation is important for biodiversity and for human well-being. Understanding the relative risks to ecosystems is fundamental to well-informed ecosystem management. The IUCN Red List of Ecosystems protocol provides an adaptable framework for risk assessment across terrestrial, subterranean, freshwater and marine ecosystems. I review a series of detailed case studies, published in this special edition of *Austral Ecology*, that apply the Red List of Ecosystems criteria to a broad range of ecosystem types. These studies show that detailed risk assessments are especially valuable as forerunners to strategic ecosystem management. Key components of Red List assessments that contribute to development of management strategies include critical diagnosis of trends and their causes, identification of dependencies that influence ecosystem responses to environmental change and selection of ecosystem-specific diagnostic variables that can be useful monitoring tools for evaluating the performance of management. Ecosystem Red List assessments are crucial to regulatory processes under environmental legislation in Australia and other countries. The IUCN Red List of Ecosystem criteria will help improve the scientific rigour of statutory listings and could also provide a unifying framework for the suite of listing processes that differ among jurisdictions for historical reasons. When integrated with a comprehensive ecosystem typology, ecosystem Red List assessments can also provide critical input into systematic conservation planning. The case studies demonstrate a range of analytical approaches to risk assessment framed to accommodate data of varying quality and abundance. I conclude by exploring opportunities and requirements for a systematic continental-scale Red List assessment of Australian ecosystems.

Key words: ecosystem management, environmental legislation, IUCN Red List of Ecosystems, risk assessment, systematic conservation planning, threatened ecological community.

INTRODUCTION

There are many reasons why ecosystem conservation is an important endeavour. Some of these relate to intrinsic values of biological diversity (Beattie 1995), others relate to the dependence of human well-being on nature (Costanza *et al.* 2014). Approaching biodiversity conservation at the ecosystem level allows broadscale ecological processes and important dependencies and interactions among component species to be considered explicitly. Ecosystem-level approaches also shine a light on far-reaching changes in common species. These often define the identity of 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). Such qualities can be overlooked when conservation priorities are heavily fixated on action directed at conservation of individual threatened species (Joseph *et al.* 2009).

Moreover, ecosystem approaches promote the need for *in situ* conservation action, demanding more sophisticated and self-sustaining solutions to conservation–development conflicts than translocations and establishment of *ex situ* populations. This is not to suggest that ecosystem conservation can or should replace species- and population-level conservation action. Rather, when efforts are integrated across multiple levels of biodiversity, conservation outcomes are likely to be much more robust and effective than those that can be delivered by a narrow focus on either species or ecosystems alone.

Risk assessment, the process of estimating probabilities of adverse events over given time frames (Rowe 1977), is a powerful framework for informing decisions about biodiversity conservation. Knowing the relative risks faced by different ecosystems helps to identify which systems are most likely to fail without remedial action. Documenting trends in risks can inform the public and management authorities about how the status of ecosystems is improving or

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deteriorating through time. Quantifying the influence of different factors or processes on overall risks can help to identify causal factors that conservation actions should address. Knowing how risks vary between alternative future scenarios can reveal which management options are likely to reduce risk most effectively.

The Red List of Ecosystems criteria and categories, adopted by IUCN in 2014, forms part of a risk assessment protocol designed to meet these needs. In this special issue, scientists and managers explore applications of the criteria across a diverse range of contrasting ecosystems in Australia and beyond. In this introductory article, I briefly review the structure of the protocol, explore the applications of ecosystem risk assessment in systematic conservation planning, strategic ecosystem management and environmental regulation, and review lessons to be learnt from the case studies. I conclude by considering the opportunities and requirements for a systematic ecosystem risk assessment for the Australian continent.

THE IUCN RED LIST OF ECOSYSTEMS PROTOCOL

Criteria and categories

The Red List protocol includes five criteria for assessing risks to ecosystems (Keith *et al.* 2013; Rodríguez *et al.* 2015). These risks may be caused by a variety of different threatening processes and may be expressed through different symptoms. Two of the criteria relate to spatial characteristics of ecosystems: criterion A, the rate of decline in distribution over specified time frames in the present, future and historic past; and criterion B, the degree that the distribution is restricted according to standard spatial metrics in combination with qualitative evidence of continuing decline. The other criteria relate to functional characteristics of ecosystems: criterion C, the rate of environmental degradation, based on a direct and sensitive abiotic ecosystem-specific variable; and criterion D, the rate of disruption to biotic processes and interactions based on a direct and sensitive biotic ecosystem-specific variable. The declines for both criteria C and D are assessed over specified time frames in the present, future and historic past. The fifth criterion, E, is based on estimates of risk derived from quantitative models of ecosystem dynamics. This allows for an integrated evaluation of multiple threatening processes, as well as interactions between them. These processes may be addressed individually under the other criteria. A summary of the current version (2.1) of the criteria is given in Appendix S1. More comprehensive explanations of the criteria can be found in Keith *et al.* (2013) and Rodríguez *et al.* (2015).

The Red List protocol requires ecosystem types to be assessed against as many of the five criteria for which data are available. Quantitative thresholds specified for each criterion allow a given ecosystem type to be assigned to one of several ordinal categories of risk: 'Critically Endangered', 'Endangered' or 'Vulnerable'. The category 'Near Threatened' is assigned to ecosystem types that almost meet the thresholds of the preceding categories, whereas those not at appreciable risk are assigned to 'Least Concern'. Ecosystem types that have been entirely replaced throughout their range by a novel ecosystem are designated 'Collapsed' (see below). An ecosystem type is assigned to the 'Data Deficient' category if, given the available data, its status is so uncertain that it cannot be assigned to any of the preceding categories; or conversely, none of the preceding categories can be ruled out with reasonable certainty. Ecosystem types that have not been assessed with available data are assigned to 'Not Evaluated'. The overall status of an ecosystem type is the highest category returned by any of the criteria (Keith *et al.* 2013).

Underlying concepts

Correct and productive application of the protocol relies on the interpretation of two conceptual components. First, 'ecosystem types' are the unit of assessment based on Tansley's (1935) concept, incorporating four key characteristics of an ecosystem (i.e. native biota, abiotic environment, key processes and interactions, and spatial distribution).

Tansley (1935) identified the biotic and abiotic elements by describing an ecosystem as a '*whole system (in the sense of physics), including not only the organism-complex, but also the whole complex of physical factors forming what we call the environment of the biome – the habitat factors in the widest sense*'. He further identified ecological processes and interactions as key elements of his ecosystem concept, '*. . . there is constant interchange of the most various kinds within each system, not only between the organisms but between the organic and the inorganic*'. The spatial expression of ecosystems is a logical derivation from the other three characteristics, implicit in Tansley's discussion on isolating ecosystems, and later made explicit by Likens (1992), '*An ecosystem is defined as a spatially explicit unit of the Earth that includes all of the organisms, along with all components of the abiotic environment within its boundaries*'.

Ecosystems (in the broad sense) may be defined at any scale, although the scale sensitivities of assessment are well known (Keith 2009; Nicholson *et al.*

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2009, 2015) and the IUCN Red List protocol has practical limits to the scales of units to which it can be applied (Keith *et al.* 2013, 2015). The applications in this special issue provide some insight into the versatility of the protocol across thematic scales (see below). Ecosystem types are, of course, artificial compartmentalizations of continuous variation in nature. Despite the somewhat arbitrary character of boundaries between spatially adjoining or thematically related ecosystem types, they are nonetheless enduringly useful tools for analyzing, managing and communicating about biodiversity, landscapes and seascapes (Keith 2009). In Tansley's (1935) words, '. . . *for the purposes of study, so that the series of isolates we make become the actual objects of our study, whether the isolate be a solar system, a planet, a climatic region, a plant or animal community, an individual organism, an organic molecule or an atom. Actually the systems we isolate mentally are not only included as parts of larger ones, but they also overlap, interlock and interact with one another. The isolation is partly artificial, but is the only possible way in which we can proceed . . . it is (these) systems . . . which . . . are the basic units of nature . . .*'.

The second important concept for application of the Red List of Ecosystem criteria is that of 'ecosystem collapse' (Keith *et al.* 2013). In essence, this describes a state transformation in which defining features (compositional, structural, functional) of an ecosystem type are lost, and the system is entirely replaced by a novel one with different defining features (Hobbs *et al.* 2006). In some cases, these transitions involve stark contrasts between initial and novel systems, for example from native woody vegetation to pasture grasslands, urban systems or cropfields (see in this issue English & Keith 2015; Tozer *et al.* 2015). In other cases, collapse may involve transitions between less contrasting systems, such as from one kind of forest to another (see in this issue Auld & Leishman 2015; Burns *et al.* 2015). This point of transition, the 'adverse outcome' central to risk assessment is typically uncertain. All risk assessments involve at least some elements of multiple sources of uncertainty (Burgman 2005).

Dealing with uncertainty in ecosystem risk assessment requires: (i) explicit description of defining features of each ecosystem type in its reference state; (ii) quantification of collapse thresholds; and (iii) evaluation of a trajectory from an initial state towards the collapsed state over the assessment time frame. Step (i) earmarks clear description of all four ecosystem components described above as an essential first step to risk assessment. Specifically, assessors must identify the defining features whose loss marks transition to a novel system. Steps (ii) and (iii) require identification of ecosystem-specific diagnostic variables that enable points of transformation and any trends towards them to be quantified. A key feature of the approach is the critical evaluation required to select ecosystem-specific diagnostic variables, which are then standardized to estimate the severity of any decline, relative to the ecosystem-specific collapsed state. This underpins the versatility of the approach for application to a diverse range of contrasting ecosystems (Keith *et al.* 2015), while maintaining superior sensitivity to generic indices of ecosystem change.

APPLICATIONS OF ECOSYSTEM RISK ASSESSMENTS

Systematic conservation planning

A comprehensive ecosystem typology for a specified region of interest allows the Red List criteria for ecosystems to be applied in a systematic context by assessing all ecosystem units globally or in smaller regions such as countries, states, ecoregions, etc. Crespin & Simonetti (2015, this issue), for example, present a preliminary national Red List of Ecosystems for El Salvador that illuminates the major drivers of ecosystem decline and provides baseline information for determining conservation priorities and for tracking the status of ecosystem-level biodiversity into the future. Systematic global and regional Red Lists can thus provide a valuable evidence base to support a range of applications including design of protected area networks, environmental reporting, prioritization of conservation actions, and environmental regulation and legislation (Keith *et al.* 2015).

Red List data are particularly valuable for systematic conservation planning because they allow both the representation and vulnerability of biodiversity features to be incorporated into options for protected area design across entire landscapes and seascapes (Margules & Pressey 2000). In addition, comprehensive Red Lists for a region of interest support predictive analyses to help understand spatial patterns and future trends in risks to ecosystems. Crespin and Simonetti (2015) examined the occurrence of El Salvadoran ecosystems at varied levels of risk in relation to a series of environmental and socio-economic variables and found that most (68%) of the spatial variation in ecosystem status could be explained by soil capability and human population density.

A systematic typology of ecosystem units throughout an area of interest is an important requirement for such applications to frame the assessment units in a consistent manner. While description of individual ecosystem types is always a linchpin for informative risk assessment, some other applications below are

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less reliant on comprehensive typologies than systematic conservation planning for entire landscapes or seascapes.

Strategic ecosystem management

The diverse contributions to this special issue of *Austral Ecology* demonstrate the support and direction that Red List assessments of individual ecosystems can provide to strategic ecosystem management (Table 1). A rigorous risk assessment based on Red List of Ecosystems criteria compels a detailed diagnostic analysis of threats and dependencies. This provides a powerful basis for developing ecosystem management objectives, strategies and actions targeted where they can reduce risks most effectively. To support their diagnoses, all of the assessments in this volume present diagrammatic process models. These are potent diagnostic tools and integral to adaptive management (Williams 2011).

Table 1 provides a synopsis of management strategies to deal with threatening processes diagnosed in each of the case studies. In several peri-urban ecosystems, the major management challenges emerging from risk assessment are to address the legacy effects of fragmentation through control of invasive species with improved technologies and enhanced implementation effort (English & Keith 2015; Tozer *et al.* 2015), or through restoration and connection of ecosystem fragments to improve habitat suitability for ecosystem engineers (Metcalfe & Lawson 2015; Murray *et al.* 2015). Several assessments identified management of disturbance regimes as crucial, either to maintain defining structural features and dependent biota (Burns *et al.* 2015), or to mitigate the impact of other threatening processes (Barrett & Yates 2015). The imperative for climate change mitigation emerged from several risk assessments, although the mechanisms underpinning risks and the scenario planning required for climate change adaptation vary greatly between the contrasting systems under threat (Auld & Leishman 2015; Clark *et al.* 2015; Wardle *et al.* 2015; Williams *et al.* 2015). Even in systems that are not currently under appreciable risks, a Red List assessment points to management strategies that should ensure continuing Least Concern status, for example, by excluding large-scale water extraction and floodplain development from the Lake Eyre Basin to ensure continued variability of water flows to its connected wetlands (Pisanu *et al.* 2015).

As well as providing strategic diagnoses to underpin management, an effective application of the Red List criteria will identify suitable monitoring variables to evaluate the performance of management (Lindenmayer & Likens 2010) and may produce tools such as simulation models that enable scenario analysis to explore the likely effectiveness of alternative management options (Burns *et al.* 2015). Exploration and screening of alternative management options and performance evaluation through targeted monitoring are key elements of adaptive management approaches (Keith *et al.* 2011).

Regulatory applications

Another important application of ecosystem Red Lists is to support legal and regulatory instruments for sustainable land and water use. Red Lists of ecosystems (*sensu lato*) have been a part of Australian environmental legislation since 1992, when a new national 'Endangered Species Protection Act 1992' included provisions for listing threatened ecological communities (Keith 2009). Several of the ecosystems studied in this special issue are currently listed as threatened ecological communities under national and/or state legislation (Auld & Leishman 2015; Barrett & Yates 2015; English & Keith 2015; Tozer *et al.* 2015).

The Australian experience demonstrates that Red List protocols can be applied for regulatory purposes in the absence of a comprehensive typology of assessment units (Keith *et al.* 2015). Most listings originate through an open nomination process and are assessed by committees of scientists (Keith 2009). The schedules therefore comprise a range of entities defined during the listing process at varied scales and based on different types of defining features.

In contrast, the schedules of threatened ecosystems under Norwegian and South African legislation are based on comprehensive systematic typologies (Lindgaard & Henriksen 2011; Driver *et al.* 2012). In a regulatory context, the consistency provided by a systematic typology is administratively attractive and ensures comprehensive coverage across the jurisdiction. On the other hand, the Australian approach has important advantages in promoting public participation in the listing process and providing flexibility to list fine-scale units when they are at risk, irrespective of whether the broader-scale units to which they belong meet criteria for threatened status (Keith 2009). This proves especially important in biodiversity hotspots where high levels of beta and gamma diversity produce mosaics comprised of many contrasting fine-scale biotic assemblages within one biome (English & Keith 2015).

Since its inception, Australian national legislation has been broadened (Endangered Species Protection Act 1992; Environment Protection and Biodiversity Conservation Act 1999) and state jurisdictions have initiated a range of legislation and

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Table 1. Threat diagnoses and management strategies emerging from ecosystem Red List assessments

Ecosystem type (source)	Habitat loss and fragmentation	Climate change	Altered fire regimes	Altered water regimes	Invasive plants	Feral herbivores and granivores	Invasive pathogens	Harvesting	Pollution
Busselton Ironstone Shrublands (English & Keith 2015)	Land-clearing regulations		Limit fire frequency	Manage water extraction within sustainable limits	Weed control (annual grasses)		Continue manual phosphite application to mitigate disease impacts		
Eastern Stirling Range Montane Heath and Thicket (Barrett & Yates 2015)			Limit frequency and extent of canopy fires				Aerial phosphite application to mitigate disease impacts		
Connected wetlands of the Lake Eyre Basin (Pisanu *et al*. 2015)				Limit water extraction and diversion; limit floodplain dams and levees; monitor variability		Limit livestock access to waterholes			
Georgina gidgee woodlands (Wardle *et al*. 2015)		Climate change mitigation (frequency of extreme heat)			Weed control (buffel grass)	Limit livestock access, control feral herbivores			
Mountain ash forest (Burns *et al*. 2015)		Reduce greenhouse gas emissions (extreme fire weather)	Fire management to reduce frequency of widespread canopy fires					Continue exclusion of logging from old-growth, limit logging of advanced regrowth	
Snow patch herbfields (Williams *et al*. 2015)		Targeted monitoring and scenario planning for climate warming			Monitoring and possible control of shrubs	Minimize disturbance by feral and domestic herbivores			
Cumberland Plain Woodland (Tozer *et al*. 2015)	Maintain land-clearing regulations				Weed control (olives, perennial grasses)				Mitigate eutrophication (through drainage management)
Gnarled Mossy Cloud Forest (Auld & Leishman 2015)		Reduce greenhouse gas emissions (cloud lift hypothesis)				Rat eradication			
Coastal lowland tropical rainforests (Metcalfe & Lawson 2015)	Limit expansion of agricultural and urban land uses; restore and link rainforest fragments								
Antarctic benthic sponge community (Clark *et al*. 2015)		Climate change mitigation (sea ice duration)							
Yellow Sea tidal flats (Murray *et al*. 2015)	Limit shoreline development and reclamation, expand protected area network			Enhance fluvial sediment outflow from impounded rivers					Reduce pollution and eutrophication of coastal waters

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policy instruments to meet their regulatory responsibilities for land and water management (reviewed by Nicholson *et al.* 2015). The semi-independent development of these regulatory tools across jurisdictions resulted in a range of inconsistencies in listing criteria, typologies and processes. These inconsistencies partly reflect the development history of scientific assessment methods over time and partly the differing sociopolitical contexts that prevailed in respective jurisdictions. More recently, imperatives for harmonization of listing processes have been recognized, with the IUCN Red List criteria for ecosystems providing a global standard for more uniform listing processes across jurisdictions (Nicholson *et al.* 2015).

A similar situation occurs in other multi-jurisdictional regions, such as Europe, where the European Habitat Directive (1972) is juxtaposed with national Red Lists of habitat types and ecosystems in countries such as Germany, Finland, Norway and others (Riecken *et al.* 2006; Kontula & Raunio 2009; Lindgaard & Henriksen 2011). Here too, the listing criteria and processes differ between jurisdictions. As well as improving scientific methods and the reliability of their outcomes, harmonization of listing processes with an international standard will promote more accurate up-scaling of listing data for reporting at various levels from sub-national to global jurisdictions (Nicholson *et al.* 2015). It will also improve the effectiveness and consistency, and reduce costs and duplication, of environmental assessments for developments that unavoidably cross-jurisdictional boundaries (Nicholson *et al.* 2015).

DATA REQUIREMENTS – LESSONS FROM APPLICATIONS

The case studies presented in this special issue demonstrate the versatility of the Red List of Ecosystems protocol to accommodate a wide range of data availability. For example, there are sufficient data to support a rudimentary assessment of shallow-water marine benthic invertebrate systems off the Antarctic coast (Clark *et al.* 2015), one of the most remote, challenging and poorly known locations on earth. Although such rudimentary assessments are possible, their outcomes must be treated with caution, as assessments of the same ecosystem type based on more, or better-quality data may produce a different outcome. The same is true of species Red List assessments (Keith *et al.* 2000). In both cases, the number of criteria assessed provides initial guidance on the reliability of assessment outcomes. The assessment of the Antarctic marine benthic invertebrate ecosystem type, for example, was based on quantitative data for criterion B (Least Concern), and qualititative evidence suggesting Near Threatened status under criterion C, with the remaining three criteria Data Deficient. As Clark *et al.* (2015) point out, better coverage of quantitative data for any of criteria A, C, D or E could change the status of the system.

An important outcome of many Red List assessments is the identification of the most important knowledge gaps that need to be filled to support more certain risk assessments. For the Antarctic marine benthic invertebrate system, a longer and more spatially representative time series of sea ice duration, and improved capacity to model it under future climates, would produce a more certain Red List outcome (Clark *et al.* 2015). In contrast, better ecosystem mapping is unlikely to improve certainty of the Red List status because even though the ecosystem distribution is poorly known, coarse spatial data are sufficient to be certain about its Least Concern status under criterion B when the system is treated as one continental-scale assessment unit.

Towards the other extreme, where large volumes of data and detailed ecological understanding are available, assessors can fully exploit the capability of the criteria to produce strongly justified risk assessments. Burns *et al.* (2015) were able to produce such an assessment for Central Victorian Mountain Ash Forests based on more than 30 years of research and long-term monitoring, which was used to support modelling and assessment of all five Red List criteria. Even though uncertainties stem from data scarcity, a wealth of data brings different challenges, such as increased complexity of analyses and time required to compile and analyze data. The availability of multiple alternative variables to assess functional declines can also create difficult choices for data-rich assessments (Burns *et al.* 2015), although the most suitable variables will be those that are most direct, sensitive, involve the simplest assumptions and produce the highest estimate of risk (Keith *et al.* 2013).

Remote sensing provides an important source of data for risk assessment, with several studies using time series of imagery to assess changes in ecosystem distribution under criterion A (English & Keith 2015; Murray *et al.* 2015; Tozer *et al.* 2015). Expanding archives of satellite imagery (e.g. Hansen *et al.* 2013) provide an increasingly powerful basis for evaluating ecosystem change under criteria A, C and D and have been identified as important resources for strengthening a number of assessments (Clark *et al.* 2015; Murray *et al.* 2015). For the Yellow Sea tidal flats, Murray *et al.* (2014, 2015) were able to compile extended time series from several different types of imagery to assess changes in ecosystem distribution over more than 50 years.

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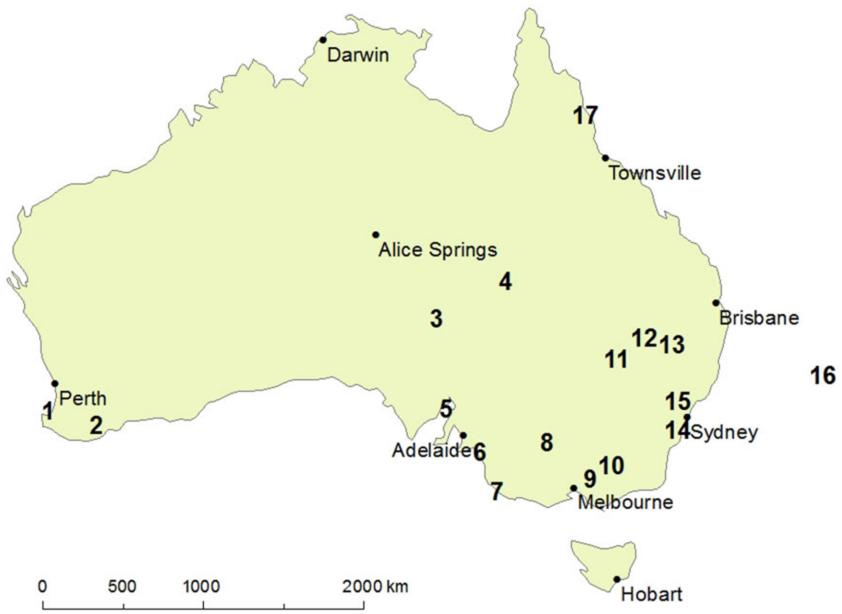

Fig. 1. Locations of Australian ecosystem types for which published Red List assessments are available. (1) Busselton Ironstone Shrublands (English & Keith 2015), (2) Eastern Stirling Range Montane Heath and Thicket (Barrett & Yates 2015), (3) connected wetlands of the Lake Eyre Basin (Pisanu *et al.* 2015), (4) Georgina gidgee woodlands, (5) seagrass meadows of South Australia (Bonifacio & Pisanu in Keith *et al.* 2013), (6) Coorong Lagoon (Lester & Fairweather in Keith *et al.* 2013), (7) Karst Rising Springs (Bonifacio & Pisanu in Keith *et al.* 2013), (8) River Red Gum and Black Box floodplain forests and woodlands (Mac Nally *et al.* in Keith *et al.* 2013), (9) mountain ash forests (Burns *et al.* 2015), (10) snow patch herbfields (Williams *et al.* 2015), (11) swamps, marshes and lakes in the Murray-Darling Basin (Kingsford in Keith *et al.* 2015), (12) coolibah – Black Box woodland (Keith in Keith *et al.* 2013), (13) semi-evergreen vine thicket (Benson in Keith *et al.* 2013), (14) coastal sandstone upland swamps (Keith in Keith *et al.* 2013), (15) Cumberland plain woodland (Tozer *et al.* 2015), (16) Gnarled Mossy Cloud Forest (Auld & Leishman 2015) and (17) coastal lowland tropical rainforests (Metcalfe & Lawson 2015).

A number of ecosystem risk assessments in this special issue use time series of ecological data gathered directly from the field to evaluate changes in the abiotic or biotic features of ecosystems. Auld and Leishman (2015) for example evaluate trends in cloud cover since 1945 on the mountains of Lord Howe Island, which is critical to maintaining the cloud forest system there, whereas Barrett and Yates (2015) inferred trends in disease impacts on mountain heathlands in the Stirling range from annual vegetation monitoring. Their photographs of that system, spanning almost 50 years, show the potential for quantitative analysis of change in vegetation structure and populations of dominant plants.

High levels of natural variability, evident in case studies of ecosystem types in arid climates (Pisanu *et al.* 2015; Wardle *et al.* 2015), can make it difficult to quantify directional trends with certainty. In such cases, long time series of data, a mechanistic understanding of processes that may drive directional and cyclical changes, and analytical techniques such as quantile regressions (Pisanu *et al.* 2015) enhance diagnostic power to distinguish declines from fluctuations.

TOWARDS A SYSTEMATIC ASSESSMENT OF RISKS TO AUSTRALIAN ECOSYSTEMS

The ecosystem risk assessments presented in this special issue of *Austral Ecology* contribute to a suite of case studies on Australian ecosystems. Together with previously published studies, these cover a diverse range of 17 ecosystem types including rainforests, eucalypt forests, woodlands, heathlands, alpine herbfields, arid shrublands, wetlands and benthic marine ecosystems throughout much of the Australian continent, associated islands and coastal waters (Fig. 1). Collectively, however, these assessments fall short of a satisfactory overview of the status of Australian ecosystems. There are four main requirements for a systematic ecosystem risk assessment across Australia, such as those undertaken in some other countries (Riecken *et al.* 2006; Kontula & Raunio 2009; Lindgaard & Henriksen 2011; Driver *et al.* 2012).

First, and most fundamentally, a comprehensive risk assessment requires a systematic spatially explicit typology of Australian ecosystems. This is needed to provide a united and consistent framework

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for assessment and scaling, and clear delineation between ecosystem types. A number of typologies of terrestrial vegetation are currently available for the Australian continent (e.g. Beadle 1981; Specht *et al.* 1995; NLWRA 2001); however, none is ideally suited to Red List assessment either because the units do not consistently conform to the required ecosystem concept (Keith *et al.* 2013), they lack appropriate ecological attribution or because they do not extend to appropriate thematic scales to support detailed ecosystem risk assessments of national scope. Nonetheless, there is a very substantial body of data, including ground observations and spatial data, that could be synthesized into an ecosystem typology to support a consistent continental-scale set of Red List assessments. Classifications of Australian freshwater (Larmour 2001; WWF/TNC 2013) and marine (Heap *et al.* 2005; Lyne & Hayes 2005; Spalding *et al.* 2007) ecosystems are less developed than those for terrestrial systems, but there are similar opportunities to develop suitable typologies for Red List assessments.

Second, high to medium resolution spatial data on ecosystem distribution are required to support the spatial components of risk assessments. Additionally, time series of spatial data are required to estimate rates of ecosystem change. Remote sensing is a key source of such data, with accumulating archives of satellite imagery and increasing digital availability of historical aerial photography providing important resources for time series analyses of ecosystem distributions (e.g. Keith *et al.* 2011), abiotic characteristics (e.g. Bormann *et al.* 2012) and biotic characteristics (e.g. Cunningham *et al.* 2009). GeoScience Australia (http://www.ga.gov.au/) and the AusCover facility within Australia's Terrestrial Ecosystem Research Network (http://www.auscover.org.au/) provide access to a diverse range of spatial data relevant to ecosystem risk assessment, supplementing data sets of global scope (Hansen *et al.* 2013).

Third, the importance of long-term ecological monitoring data for both risk assessment and management of ecosystems cannot be understated (Lindenmayer *et al.* 2012). There is often a perception that the coverage of such data is too thin to be representative of broad-scale change, but the published case studies show that a reasonable search effort sometimes reveals a surprising cache of site-specific time series data that can provide powerful insights into ecological responses to environmental change at landscape scales. For example, a series of long-term ecological data sets on Australian heathlands were recently assembled to produce a continent-wide synthesis of ecosystem change (Keith *et al.* 2014). When rigorously designed and focussed on relevant questions that directly address defining features of ecosystems, these studies can provide a strong basis for assessing ecosystem degradation under criteria C, D and E of the Red List protocol. Initiatives such as Australia's Long Term Ecological Research Network (http://www.ltern.org.au) have begun building valuable repositories for these types of data.

Finally, as ecosystem declines may be driven by a wide range of contrasting processes, effective cross-disciplinary collaborations are imperative to ensure robust and thorough risk assessments. A risk assessment may draw on expertise, not only in plant and animal ecology, but also hydrology, soil science, climatology, oceanography, mathematical modelling and other disciplines. A systematic ecosystem risk assessment for any region will also require broadly based expertise, capacity building, training, collaboration and intellectual leadership among the scientific community. Although the requirements identified above are not unique to an Australian ecosystem assessment, the broad spectrum of authors and collaborators contributing to this special edition of *Austral Ecology* suggests that Australian scientists are up to the task. The substance of their work shows how risk assessments can contribute strategically to better outcomes of ecosystem management.

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SUPPORTING INFORMATION

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

Appendix S1. Summary of IUCN Red List of Ecosystems categories and criteria version 2.1.

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