NSW SCIENTIFIC COMMITTEE

Dr Allan Hawke
C/- Secretariat
Independent review of the EPBC Act 1999
GPO Box 787
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Dear Dr Hawke,

The NSW Scientific Committee is responsible for assessing threatened species, populations, ecological communities and key threatening processes in NSW in accordance with the NSW Threatened Species Conservation Act. The Committee must also review any listings under the Environment Protection and Biodiversity Conservation (EPBC) Act that relate to species and ecological communities that occur in NSW.

Please find attached a submission from the Committee for your consideration. This submission addresses only matters related to the listing process for threatened species, ecological communities and key threatening processes.

Yours sincerely

Lesley Hughes

Professor Lesley Hughes
Chairperson
Scientific Committee

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Submission to the Independent Review of EPBC Act 1999
Prepared by NSW Scientific Committee, December 2008

The NSW Scientific Committee is an independent statutory body responsible for maintaining schedules of threatened species, populations and ecological communities and key threatening processes under the NSW Threatened Species Conservation Act 1995. The Committee receives and assesses nominations for additions, deletions and changes to the status of listed entities, applying criteria defined in the NSW Threatened Species Conservation Regulation 2002. This submission addresses only matters related to the listing process for threatened species, ecological communities and key threatening processes under the Commonwealth Environment Protection and Biodiversity Conservation Act 1999.

1. Separation of risk assessment from decision making on conservation action.

The Schedules of listed species and communities under the EPBC Act perform two separate functions: a reporting role, to inform the public about the status of Australia's biodiversity; and a planning role to inform decisions about conservation of the listed species and communities. The question of whether a species or ecological community is at a substantial risk of extinction within a given time frame (and therefore eligible for listing under the EPBC Act) is purely a scientific one, to be determined from available evidence on relevant biological characteristics and threatening processes. In contrast to the listing process, the decision-making process for conservation actions must consider broader socio-economic issues, in addition to scientific evidence. In this case, relevant scientific considerations include level of extinction risk, the extent that alternative actions are likely to reduce those risks and likelihood that each action will be successful in mitigating risk. The socio-economic issues include costs of actions, social values associated with the species or community at risk and the implications of conservation actions for competing interests. These socio-economic considerations are legitimate concerns for elected governments in deciding how best to deal with threatened biodiversity, but they have no role in determining whether or not a species or community is at sufficiently high risk of extinction to warrant listing.

Under current EPBC arrangements, the Department and Minister both play central roles in both the listing process and implementation of actions (e.g. through recovery planning, threat abatement planning, assessment of development proposals, etc.). The current Threatened Species Scientific Committee (TSSC) has an advisory role only (Clause 504, EPBC Act) and is dependent on the Department for all administrative and technical support. All of its listing decisions are subject to Ministerial approval and the Committee relies heavily on Departmental staff to carry out assessments that form the basis of its listing decisions on nominated species and communities.

The lack of clear separation of responsibility for these different tasks (listing and action), generates a public perception that socio-economic considerations infiltrate the listing process and potentially leaves the listing process open to future abuse. This perception has been strengthened by periodic statements and addresses from the TSSC about balancing social considerations and priorities in both listing and management of threatened species and communities (e.g. TSSC 2004).

Public confidence in the listing process would be greatly strengthened if there was a clear legal separation of listing responsibilities from decision-making on conservation actions. This would require establishment a committee of specialists whose primary responsibility was assessment and listing, with a legislative mandate to make determinative (rather than advisory) listing decisions, and functionally independent of the Minister and Department responsible for prioritisation and implementation of conservation actions. The Committee would need to be equipped with scientific expertise and technical support required to undertake this work in a rigorous and timely fashion (see 4. below). This may also require adjustment to the Committee's advisory roles on other matters (e.g. recovery planning) to accommodate the increased workload required to undertake assessment and listing independently. Such a model has been implemented successfully by IUCN Species Survival Commission for many years (though its Red List Specialist Groups, Red Listing Authorities and the Red List Criteria Standards and Petitions Working Group) and, more recently (since 1996), under the NSW Threatened Species Act (through the NSW Scientific Committee).

Recommendation: That the functions of the Threatened Species Scientific Committee be revised to: i) establish its primary responsibility as a statutory listing authority with powers to make determinative (rather than advisory) listing decisions; ii) improve the focus of the Committee's functions on its assessment and listing roles; and iii) establishing the Committee's functional independence of the Minister and Department; through dedication of secretariat and technical support staff that report directly to the Committee Chair. This would require reform of Clause 504 of the EPBC Act, which currently establishes the Committee with a purely advisory role on an extremely broad and open range of subjects.

2. Public participation and processes for prioritising nominations

Public participation is an important aspect of biodiversity legislation in Australia and overseas. Under the EPBC Act, the public has the opportunity to participate in the listing process by nominating species, ecological communities and key threatening processes for listing and by making submissions on nominations. While the intent of the Act encourages public participation, the current administrative arrangements may have the reverse effect by generating perceptions of undue bureaucratic control over the listing process. Moreover, the nomination and assessment process (see Clause 194A of the Act) includes a series of at least four filters that could be seen to dilute public involvement in the listing process. These include the following: nominations must follow a rigid annual cycle; the Minister determines a conservation theme for nominations; nominations are vetted by the Department for compliance with EPBC Regulations; priorities for assessment are set by TSSC, and must then be approved (with modification if deemed necessary) by the Minister; listing advice from TSSC is subject to vetting and approval by the Department/Minister. For example, Clause 194G, provides for the TSSC to exclude a nomination from the priority assessment list if TSSC considers it lacks capacity to undertake an assessment while performing its other duties or any other reason that TSSC considers appropriate. Such provisions would seem obstructive to fair and reasonable consideration of public nominations. There are several aspects of the current administrative process that could be managed in an alternative way that promotes more effective public engagement and confidence in the listing process.

The recently implemented annual nomination cycle, would seem to impose unnecessary constraints on public participation for the convenience of managing nominations. Listing processes in other jurisdictions in Australia and abroad operate effectively in the absence of such an imposition.

The Minister "may determine a conservation theme or themes for an assessment cycle [Clause 194D of the Act]. For example, the conservation theme for the assessment cycle that commenced on 1 October 2007 was 'rivers, wetlands and groundwater dependent species and ecosystems of inland Australia'. The Minister may seek advice from the Threatened Species Scientific Committee (TSSC), in establishing the theme" [excerpt from EPBC Review Discussion Paper]. An implicit corollary of such a call is that other species or communities nominated by the public that do not conform with the current theme are less likely to be considered as a priority for assessment. This may have the effect of discouraging some important nominations that could make important contributions to the protection of Australia's biodiversity.

It is unclear how rigidly nominations are vetted by the Department for compliance with the EPBC Regulations. If applied too rigidly, this could also provide an obstacle to public nominations by imposing unnecessary requirements on the format of information and placing too much onus on the nominator to provide and comprehensively assess certain technical information. Conversely, poorly substantiated nominations require a greater effort to investigate and therefore may involve significant opportunity costs in progression of other nominations. Ideally, the public should be made aware that any 'reasonable' effort to compile relevant information in support of a nomination will be diligently assessed under the EPBC Act.

Under current administrative arrangements, only those nominations on the "finalised priority assessment list" (Clause 194K of the Act) are advertised for public comment and progressed to assessment by the TSSC. The "finalised priority assessment list" is initially determined by the TSSC and is subject to change and approval by the Minister (in having regard to any matters that the Minister considers appropriate", Clause 194K(3)) before public release. The EPBC website states that the TSSC considers 'applicable' nominations against 'a set of criteria', however, no details of these criteria appear to be available. There are several potential consequences of such an approach: the public (including relevant external experts) may have information relevant to prioritisation that is not initially available to TSSC or the Minister; prioritisation may be open to abuse with little public accountability; nominations assessed by the TSSC or Minister (rightly or wrongly) as low-priority may languish indefinitely unless there are safeguards; and prioritisation itself may be viewed by external stakeholders as cherry-picking in the absence of a more transparent process.

Recommendation: That consideration be given to the following initiatives: i) the Department prepares, or commissions nominations which address its priority themes (in lieu of the Minister announcing a preferred theme); ii) public comment is sought, by advertisement, and specific invitation to relevant experts, on all nominations (except those considered vexatious) within two months of receipt; iii) the submissions received are considered in the prioritisation of nominations; iv) transparent prioritisation criteria are developed and implemented and reasons for priorities are made publicly available; v) TSSC determines its assessment priorities and manages its workload independently of the Department and Minister; and vi) additional resources are made available to support the assessment process. This would require reform of Clause 194 to remove unnecessary filters on public involvement and improve accountability of the prioritisation and assessment process.

3. Turn-around time for assessments

Concerns have been raised previously regarding the time required to resolve nominations under the EPBC Act. The Commonwealth's primary response has been to prioritise nominations using an approach that has generated other problems (see 2.). One outcome of this process has been to reduce the number of nominations that the Commonwealth commits to resolving each year. In 2008, for example, the "finalised priority assessment list" included 20 species and 8 ecological communities nationally, with most of the latter not scheduled for a decision until October 2010. As a result, EPBC listings lag well behind equivalent state listings, while many important ecological communities and key threatening processes are yet to be assessed at Commonwealth level. While the EPBC Act should provide flexibility in the time line to ensure that assessments are scientifically rigorous, additional measures are required to resolve listings more efficiently and provide for accountability in the event of unavoidable delays.

Recommendation: Examine the following options for improving the efficiency of assessments: i) improving the level of specialist expertise on the TSSC (see 4.); ii) enhancing resources available to TSSC (particularly through provision of dedicated technical staff, see 4.); iii) reducing TSSC's reliance on Departmental bureaucracy to progress assessments; iv) more clearly establish TSSC's primary function as the assessment of listing nominations, and reducing its advisory roles in other matters (see 1.); v) improving the transparency and efficiency of prioritisation after public comment is sought (see 2.); and vi) introducing provisions for emergency listings under the EPBC Act..

4. *TSSC and its support staff*

Assessment of nominated species and communities can sometimes be a complex scientific task. Despite this, it is important to avoid errors of detail and logical inconsistencies within listing statements. Such problems have occurred in some recent listings and may affect their legal validity in contested cases. For example, in the Temperate Highland Swamps on Sandstone ecological community, the major listed example swamps do not occur on sandstone. There are a number of other scientific issues concerning the integration of data, circumscription of ecological communities, and assessment of risk that warrant closer attention. To discharge its responsibility effectively, the membership of TSSC needs to be equipped with appropriate expertise in relevant disciplines. Clause 502 of the EPBC Act, which simply requires that the "Minister is to determine in writing the composition of the Committee, including the qualifications of its members" currently does not adequately address these requirements. The TSSC also requires skilled technical support from personnel trained in the evaluation of ecological data and familiar with concepts and methods of conservation risk assessment. This technical support is required in addition to secretariat staff who assist in managing the business and correspondence of the Committee.

Recommendation: That Clause 502 of the EPBC Act should require, as a minimum, the membership of the TSSC to include specialists in: 1) a range of different biotic groups (vertebrates, invertebrates, plants); 2) population biology; 3) community ecology; and 4) ecological risk assessment. The members of TSSC should be selected on the basis of merit (e.g. publication and/or standing in a relevant field) and, ideally, should include a mix of experienced and early-career scientists nominated by independent knowledge brokers (e.g. universities, the Ecological Society of Australia, CSIRO, museums and herbaria). Appointment on the basis of advice from these independent knowledge brokers will strengthen public confidence in the merit-selection process. In addition, technical support should be provided to the TSSC by dedicating three specialist positions, which have relevant qualifications and experience in conservation assessment of animals, plants and ecological communities. To ensure an appropriate level of independence from Departmental business, these positions should report directly to the Committee Chair.

5. *Amending lists of threatened native species, ecological communities and key threatening processes*

As outlined above (see 1.), the schedules of the EPBC Act perform two primary functions. Firstly, they inform the public about the current status of Australia's biodiversity and trends in its status over time. Secondly, they identify species, communities and threats that require action to conserve Australia's biodiversity. Clauses 186(2Ab) and 187(3b) of the EPBC Act provide for the Minister to delete listings if inclusion of the native species, or ecological community key threatening process is not contributing, or will not contribute, to the survival of the native species or ecological community. Similarly Clauses 186(2b) and 187(2b) provide for the Minister to include native species or ecological communities on the basis of the expected effect that listing will have on their survival. However, changes to the lists for these reasons prevent the lists from accurately performing their first function (as a reporting tool) because they confound the assessment of risk with conservation action (see 1.).

Recommendation: That eligibility for respective categories of threat be the sole consideration for amending listings of threatened native species and ecological communities. This would require amendment of the EPBC Act to delete Clauses 186(2b), 186(2Ab), 187(2b) and 187(3b).

6. *Logical inconsistencies in listing criteria.*

The specific criteria for listing species under the EPBC Act, detailed in Regulations of the Act and supporting documentation (e.g. species nomination form; TSSC undated a), are adapted from international best-practice, the IUCN (2001) Red List categories and criteria. The Red List criteria have a strong basis in population theory and are the outcome of an extensive international collaboration of leading experts (Mace & Lande 1991; Mace et al. 2008) and have been exposed to extensive examination and testing in the scientific literature under a wide range of scenarios. The EPBC listing criteria for species deviate in some details from the IUCN Red List criteria, and the rationale for these changes are unclear.

The criteria for listing ecological communities under the EPBC Act (TSSC undated b) were derived from a more limited local consultation process than those for species, as there are currently no internationally agreed criteria for assessing ecological communities. Consequently, there are some logical inconsistencies between the EPBC listing criteria for communities and those for species. For example, the indicative thresholds delimiting different categories of risk for communities entail substantially larger reductions and smaller range sizes than those delimiting equivalent categories of risk for species, even though they purport to represent the same levels of extinction risk (Nicholson et al. in press). As EPBC listing criteria are more stringent for communities than the international standard for species, it would be possible for a particular ecological community to be composed entirely of threatened species, yet not meet any of the criteria for listing as threatened itself.

Recommendation: That the EPBC listing criteria for ecological communities, particularly the indicative thresholds, be reviewed and adjusted as required to attain logical consistency with EPBC listing criteria (and IUCN Red List criteria) for species. This would require amendments to supporting documentation and possibly EPBC Regulations, though is unlikely to require amendment to the EPBC Act, in which the listing criteria are described in a suitably general form.

7. Definition of ecological communities

Assessment of ecological communities poses one of the most complex and challenging scientific tasks in the implementation of the EPBC Act. The EPBC Act defines an ecological community as: "the extent in nature in the Australian jurisdiction of an assemblage of native species that:
(a) inhabits a particular area in nature; and
(b) meets the additional criteria specified in the regulations (if any) made for the purposes of this definition."

Scientific definitions of ecological communities refer to an assemblage of species that occur together in time and space (Begon et al. 2006; Keith in press). On this basis, a listed community may be identified at a particular site, so long as the composition of its assemblage resembles that described in the listing statement. This form of diagnosis also accommodates change; a community may no longer occupy a particular site if the composition of the assemblage no longer resembles that described in the listing statement.

The current practice under the EPBC Act follows an alternative and unusual approach that seeks to list only some states of an ecological community, rather than whole communities. These states are defined by thresholds in proxy variables such as structural features and extent of infestation by introduced species. This approach is described in Guidelines prepared by TSSC (2004). Examples may be seen in listing statements for Littoral rainforest and coastal vine thicket of Eastern Australia, White box - Yellow box Blakely's Red Gum woodland and derived native grassland, and other recent listings. This approach is a significant departure from long-established scientific practice, which identifies communities on the basis of their species composition (see, for example, extensive literature in phytosociology). In contrast, the current approach for identifying and describing ecological communities under the EPBC Act confounds the condition of a community with its defining features (i.e. the composition of the assemblage and its particular area in nature). Consequently, delimitation of listed communities is reliant on arbitrary thresholds in proxy variables (which are often poorly justified), submerging inherent and widely-recognised uncertainties in the identification of ecological communities that demand more explicit treatment. Compared to established approaches based solely on species composition of the assemblage, this introduces a number of unnecessary complexities and serious logical conflicts in the assessment process, the management of listed communities and in enforcement of their protection. Examples include:

i) difficulties in estimating geographic distribution, rates of decline and other status parameters for particular states of a community, which make them difficult to assess with reasonable certainty (cf. whole communities);

ii) transitions between states of a community create uncertainty in the status of particular sites as they move in and out of listed community states, with the result that protection status may be transient over relatively short time scales or in response to short-term management actions;

iii) incorporation of condition parameters into the description of a listed community state may create acute difficulties for enforcement of protection. For example, a prosecution for the destruction of a listed community would have to prove beyond reasonable doubt, not only that the assemblage of species was present on the site, but also that it conformed to particular parameters of tree cover, native groundcover, abundance of introduced species, patch size, etc. Such detailed evidence would rarely be obtainable after illegal disturbance of a stand of a listing community.

As a result, ecological communities listed under the current approach promoted by TSSC and the Department are unlikely to provide effective protection for the biological components and processes that they are designed to conserve.

Recommendation: That the current approach to listing states of ecological communities defined by proxy variables representing condition be abandoned and replaced with an approach that is consistent with established scientific practice, that lists communities in their entirety based on similarity in species composition.

8. Alignment of Commonwealth, State and Territory Listings

While there are several legitimate reasons for differences in listings of species and ecological communities between Commonwealth, State and Territory jurisdictions, some differences arise for artifactual reasons. Time lags (on both sides) are an example of the latter, and could be reduced particularly where the listed species or community is endemic (i.e. restricted) to one state or territory. For example, it may be possible to develop protocols for joint assessments in the event that contemporaneous nominations arise, or abbreviated supplementary assessments where listings have already occurred in one jurisdication and are also relevant for consideration in another jurisdiction. These protocols would need to ensure adequate provision for public input and scientific rigour in assessment.

Other differences arise between jurisdictions as a result of differences in scientific approaches to application of listing criteria (see 6. and 7. for examples). These may occur despite overall uniformity in the general description of listing categories. In some cases there could be legitimate reasons for differences in approach. However, these could be minimised with reference to international best-practice (e.g. IUCN 2001; 2008), and through periodic interaction between the membership of scientific committees from different jurisdictions (e.g. through scientific workshops).

Recommendation: Examine the following options for improving the alignment between Commonwealth, State and Territory listings: i) develop protocols for joint assessments or abbreviated supplementary assessments (see above), while ensuring adequate provision for public input and scientific rigour in listing processes; ii) promote closer commitment in all jurisdictions to scientific approaches and methods adopted internationally as best-practice; and iii) foster scientific interaction through cross-jurisdictional collaborative projects and workshops to develop approaches and methods for assessment of species and ecological communities.

9. Omissions in listings of Key Threatening Processes

A recent review of threatening processes (Auld & Keith in press) revealed omissions in EPBC listings of Key Threatening Processes compared to listings in state jurisdictions. In particular, disturbance regimes (notably certain fire regimes) and introduced plant species were under-represented in current EPBC listings.

Recommendation: That the TSSC consider redressing under-representation of particular types of threat (including disturbance regimes and invasive introduced plants) on the EPBC listings.

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Assessing the threat status of ecological communities

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Abstract

Conservationists are increasingly interested in determining the threat status of ecological communities as a key part of their planning efforts. Such assessments have proved difficult due to conceptual challenges and a lack of generally-accepted criteria. We critically review twelve protocols for assessing the threat status of communities, and identify conceptual and operational issues associated with developing a rigorous, transparent, and universal set of criteria for assessing communities, analogous to the Red List standards for species. We examine how each protocol defines a community and its extinction, and how each applies three over-arching criteria: (1) decline in geographic distribution; (2) restricted geographic distribution; and (3) changes to ecological function. The protocols vary widely in threshold values used to assess declines and distribution size, and the timeframes used to assess declines, leading to inconsistent assessments of threat status. Few of the protocols specify a scale for measuring distribution size, although assessment outcomes are highly sensitive to scale. Protocols that apply different thresholds for species versus communities tend to require greater declines and more restricted distributions for communities than species to be listed in equivalent threat categories. Eleven of the protocols include a reduction in ecological function as a criterion, but almost all assess it qualitatively rather than quantitatively. We argue that criteria should be explicit and repeatable in their concepts, parameters and scale, yet applicable to a broad range of communities, and address synergies between different types of threats. Such criteria should focus on distribution size, declines in distribution, and changes to key ecological functions, with the latter based on workable proxies for assessing the severity, scope, and immediacy of degradation. Threat categories should be delimited by thresholds that are assessed at

standard scales and are logically consistent with the viability of component species and important ecological functions.

Introduction

Conservation professionals and institutions are increasingly concerned with the conservation of ecological communities in addition to individual species. In October 2008, the 4th World Conservation Congress in Barcelona passed a resolution (CGR4.MOT024) to develop a global standard for assessing the status of ecosystems. Developments such as this reflect the realization that an exclusive focus on species-based approaches is unlikely to conserve all components of biodiversity (Franklin 1993). The conservation of communities protects ecological patterns and processes, such as species assemblages, interactions of species with each another and their environment, and higher level biotic and abiotic processes, including disturbance regimes and habitat structure (Franklin 1993; Noss 1996; Cowling & Heijnis 2001). Protecting such processes can ensure the continued provision of ecosystem services (Balmford et al. 2002). The conservation of communities is also assumed to act as a 'coarse filter' or surrogate for species, in particular species that are unknown or poorly understood (Jenkins 1976; Franklin 1993; Noss 1996; Cowling & Heijnis 2001).

The IUCN Red List criteria (2001; 2008) provide a broadly-accepted and widely-used set of criteria for assessing the threat status of species, where species are assigned to categories of threat based on quantitative criteria that reflect varying risks of extinction (Mace et al. 2008). Currently, there is no comparable, widely-accepted assessment protocol for communities, although several have been proposed. There are a number of reasons for developing an analogous protocol for ecological communities: (1) to assess the risk that a given community will be lost; (2) to provide a relative ranking of threat level of communities to assist priority setting; and (3) to act as a proxy for assessing the extinction risk of unknown or poorly-understood species. A consistent protocol would also allow changes in the status of communities to be monitored through time.

Several protocols for assessing the threat status of communities have been proposed by researchers (e.g. Paal 1998; Benson 2006; Rodriguez et al. 2007), governments at national and regional levels (e.g. Blab et al. 1995; EPBC Act 1999; Sattler & Williams 1999), and non-government organizations (NGOs); the latter includes NatureServe's conservation status ranking system (Faber-Langendoen et al. 2007; Master et al. 2007), probably the most widely-used protocol.

We critically review current thinking and available protocols for assessing the threat status of ecological communities, and we identify and discuss key conceptual and operational issues associated with developing a Red-List style protocol for communities. We primarily address protocols for assessing terrestrial communities, because this has been the focus of much of the current literature. A review of methods for defining or delineating communities, while useful, is beyond the scope of this paper (for reviews of classification methods, see Whittaker 1978; Ferrier & Guisan 2006).

We review twelve assessment protocols that include quantitative criteria (Table 1). We begin by examining how each of the protocols defines a community and community extinction. We then focus on the three groups of criteria for assessing threat status that we see as most important, and that emerge most frequently from the literature: (1) decline in geographic distribution; (2) restricted geographic distribution; and (3) changes or disruption to ecological function. We review how these criteria are represented in each of the protocols, and the thresholds applied to designate threat categories. We also examine how the spatial scale of assessment may influence outcomes using a data set from south-eastern Australia. We conclude the paper with recommendations that should facilitate the development of a set of criteria for ecological communities analogous to the IUCN Red List criteria for species.

The assessment protocols reviewed

We searched the scientific and grey literature for protocols that have been proposed, applied, or are in current use for assessing the threat status of communities. We excluded from our review the many protocols that rely wholly on qualitative listing criteria (e.g., Council of the European Communities 1992; Blab et al. 1995) and methods whose aim was broad-scale priority-setting (e.g., Ricketts et al. 1999; Hoekstra et al. 2005). Most of the protocols we reviewed have a rule-set structure similar to the IUCN protocol for species (Table 1), whereby a community is assessed against each criterion individually and assigned to the highest threat category for which any criterion is met (IUCN 2001). Two of the protocols we reviewed apply similar criteria and thresholds (where applicable) to both species and communities: NatureServe's conservation status assessment system; and the NSW TSC Act, which uses the IUCN Red List species criteria to guide the listing of both species and communities. We elaborate on the rationale for congruence or difference in thresholds between species and communities in the *Discussion*.

Defining ecological communities and their extinction

Ecological communities are assemblages of species that occur together in time and space (Begon et al. 2006). Ecosystems are functional systems formed by communities and their environments, with associated transfers and cycles of energy and matter (Whittaker 1975; Franklin et al. 2002; Begon et al. 2006). Some definitions emphasize the interactions between species within a community (e.g., Whittaker 1975), but here we focus on the co-occurrence of species and their area of co-occurrence to define an ecological community (Keith in press). Like species, communities may be classified hierarchically, such that many fine-scale assemblages may be nested within a smaller number of broadly defined units (Gauch & Whittaker 1981). Variation in community properties such as composition, structure and processes is widely recognized as being continuous (Gleason 1926; Austin 1985). For this and other reasons, the properties that distinguish different communities from one another and the boundaries that delineate their distributions are inherently more uncertain than is the case for species, for which a formal hierarchical international system of taxonomy and nomenclature exists (e.g. International Code of Botanical Nomenclature).

The protocols use a variety of definitions (Table 1) for communities and ecosystems, which we treat similarly in this review and refer to generally as ‘communities’. These definitions tend to be rather generalized (Keith in press), allowing flexibility and pragmatism in defining communities for the purposes of conservation planning and government legislation. This reflects the scarcity of comprehensive classification systems at a scale appropriate for listing in most countries or regions, despite progress in developing frameworks for classification (e.g., Grossman et al. 1998; Mucina et al. 2000; Leathwick et al. 2003; Thackway et al. 2007). Indeed, Noss et al. (1995) find the generality of their definition “useful because it allows assessment of loss or degradation of structural, functional, or compositional aspects of ecosystems … at any level of the classification hierarchy and at any spatial scale.” However, the lack of specificity with respect to community structure and composition has important implications when evaluating the degradation, loss of associated structure, species and processes, and ultimately the extinction of a community. Inconsistency in the classification and scale of ecological communities may lead to distortions in the outcomes of assessment, with more communities assessed as being at risk in those areas from which more detailed information is available. Detailed and quantitative vegetation classification of large regions (e.g., Rodwell et al. 2002; Jennings et al. 2004), in combination with accurate map data, therefore provide a basis for comprehensive and consistent assessments. The classifications and maps of ecological communities that generate the most robust and informative generalizations about species distributions are those based on quantitative sampling and analysis of the biota (e.g., Rodwell et al. 2002) and systematic integration of compositional data with spatial data (e.g., Keith & Bedward 1999). The more comprehensive and more evenly stratified the sampling, the more robust and more representative of ecological patterns the resulting classifications and maps will be.

The IUCN Red List assigns species to classes representing different levels of extinction risk (IUCN 2001; Mace et al. 2008). Efforts to develop a similar assessment protocol for communities quickly encounter a major roadblock: community extinction is far more difficult to define than species extinction (“when there is no reasonable doubt that the last individual has died”; IUCN 2008). While communities that have been obliterated by development are readily seen as destroyed, it is much more difficult to determine when a community has become so modified or degraded that it no longer exists (Noss et al. 1995; EPBC Act 1999). Community degradation can be viewed as having at least three interconnected components: (1) change in species composition; (2) change in structure (e.g., the multi-layered structure characteristic of many old-growth forests and coral reefs); and (3) disruption to ecological processes, including disturbance regimes, propagule dispersal and species interactions (see *Changes to Ecological Function*, below; Franklin et al. 2002). Determining threshold values for these components poses some difficult challenges for conservation biologists. How many and which species or how much structure can a site lose before the ecological community is deemed to be gone? The answers to such questions are likely to vary from community to community for ecological reasons. Without some standardization of assessment aided by specific criteria, however, they are also likely to vary from person to person, given the inherent subjectivity and uncertainty in defining community extinction

The difficulty in determining when a community is extinct is acknowledged in most of the protocols we reviewed. Such determinations inevitably rely heavily on the judgment of experts (EPBC Act 1999). Not surprisingly, the definitions of extinction employed by the protocols are vague (Table 1), and, by themselves, offer limited guidance on categorizing a community's risk of extinction in a given time frame. Consequently, most of the protocols rely on symptoms as proxies for extinction risk, which we review below.

The definition of a community and its extinction are two of many sources of uncertainty that pervade the assessment of communities. Others include delineating where one community ends and another begins (spatially, temporally, and in terms of degradation), mapping error, and understanding the key processes that affect viability. The growing literature on dealing with uncertainty in listing threatened species (e.g. Akçakaya et al. 2000; Regan et al. 2002; IUCN 2008) provides a useful starting point for addressing uncertainty in community assessment (Keith in press). Of the protocols we reviewed, only the EPBC Act, NSW TSC Act and NatureServe make explicit mention of how to deal with uncertainty in assessing community status, while some others included a 'Data Deficient' category for communities that have insufficient information for assessment (Austrian Red List; Rodriguez et al. 2007; Raunio et al. 2008).

Criteria represented in reviewed assessment protocols

Ten of the twelve protocols we reviewed explicitly incorporate all three of the major assessment criteria: decline in distribution; restricted distribution; and changes to ecological function (Table 2). Noss et al. (1995) include both community degradation and destruction within a single "decline" criterion. However, this confounds the concept of a reduction in spatial distribution with that of a decline in the structural, functional and compositional features of a community, which are treated separately in other protocols (see *Changes to ecological function*, below). There is considerable variation in how the three major criteria are expressed, and the thresholds, conditions, and the implicit temporal and spatial scales applied.

Decline in geographic distribution

All of the protocols we reviewed include the decline in geographical distribution as a key criterion, and all except one (Paal 1998) use quantitative thresholds (Table 2). Most specify rates of decline in spatial parameters, such as area of occupancy or number of occurrences (see below for definitions), for estimating the magnitude of decline. The protocols differ in the timeframe over which the decline is measured, the thresholds for percent decline (Figure 1), and whether evidence of a continuing or future threat is required as a condition for meeting the criterion (Figure 1; Table 2).

The use of different timeframes across the protocols for assessing decline rendered comparisons difficult. Most protocols include both long-term decline and short-term decline (including recent past, current, and predicted future rates of decline) as separate sub-criteria (Table 2). In most protocols, different threat categories entail different thresholds for percent decline over common timeframes, although some require those declines to be assessed over different timeframes (e.g., Benson 2006; WA TEC). The timeframe for considering long-term declines can be anchored to an historical event such as the industrial revolution or European colonization (e.g., EPBC Act; Austrian Red List; NatureServe; Noss et al. 1995; Rodriguez et al. 2007), a shifting timeframe (e.g., last 30 years in Rodriguez et al. 2007; last 50 years in Raunio et al. 2008), or a biologically-relevant timeframe appropriate to the life cycle and habitat characteristics of component species (NSW TSC Act).

Thresholds for long-term decline in distribution vary widely between each of the protocols (Figure 1). Most protocols do not require evidence of on-going threat for a community to be assigned to a category (Table 2). The NSW TSC Act, following IUCN (2001), specifies two different thresholds for decline in distribution: higher thresholds of decline are applied when the threats are understood, reversible and known to have ceased (Figure 1). By contrast, under the WA TEC and Rodriguez et al. (2007) assessment based on past decline in distribution is contingent on demonstrable ongoing threat of further decline.

Restricted geographic distribution

Seven of the twelve protocols include restricted distribution as a quantitative measure of the threat status of a community (Table 2). Unlike decline, which relies on an estimate of past distribution, this criterion is based solely on the current distribution, whether the community is restricted naturally or due to anthropogenic loss.

Most of the protocols specify the spatial parameters that must be used to estimate the current geographical distribution (Table 2). *Extent of occurrence* (EOO), is the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known or inferred occurrences, including areas that are unsuitable or unoccupied (e.g., cleared), and is typically measured using a minimum convex polygon or an alpha-hull (IUCN 2008). The second parameter, *area of occupancy* (AOO), is the area within the extent of occurrence that is occupied by the community, and thus excludes areas that are unsuitable or unoccupied (IUCN 2008). The *number of occurrences* (NOO) is a third measure of distribution that is applied in four community assessment protocols (Table 2), although the definition of an occurrence varies considerably. The number of occurrences may be correlated with fragmentation (a highly fragmented landscape will contain many small occurrences), potentially rendering it a poor indicator of community viability when used as a measure of distribution size. Consequently, some protocols include fragmentation as a criterion (Table 2), which we consider under *Changes to ecological function* (below).

Threat status of ecological communities

Seven of the protocols measure AOO, with three of these also using EOO (Figure 2, Table 2). Threshold values associated with both AOO and EOO vary greatly between protocols (Figure 2). These differences are greater for the lower threat categories than the higher ones. In four protocols, the criterion relating to distribution size is conditional on evidence of on-going decline or threat (Table 2, Figure 2); the others do not impose this requirement. Communities with restricted distributions may be intrinsically at risk from stochastic events, such as storms, fires or floods, and threats such as development pressure or invasive species, irrespective of whether there is evidence of any recent decline. Such communities may not be listed under protocols in which assessment of distribution size is conditional on ongoing threats, depending on how these threats are interpreted. A more prudent approach may be to omit any requirements for ongoing threat for the distribution size criteria, especially at the higher threat categories.

Influence of spatial scale on estimates of distribution size

Two components of scale are critical to defining, measuring, and assessing the threat status of a community: the thematic scale, representing the hierarchical level at which communities are classified, and the spatial scale at which each community is mapped. Communities that are defined at coarse thematic and spatial scales will generally have more extensive distributions and appear less threatened than communities defined at finer scales (Kirkpatrick 1998), while communities defined at fine spatial and thematic scales provide a more precise and comprehensive representation of biodiversity (Pressey & Logan 1994).

AOO and similar measures of geographic distribution are highly sensitive to the scale of estimation: the coarser the spatial scale, the larger will be the estimated area occupied, with significant implications for assessment against fixed thresholds (Keith et al. 2000; Hartley & Kunin 2003; IUCN 2008). Analyses in the online appendices show the influence of spatial scale on assessment outcomes for 183 ecological communities in southeast New South Wales, Australia (Tozer et al. 2006) that were originally mapped on a 25-metre grid. By calculating the AOO of each community using increasingly large grid cells and allocating it to the appropriate threat category based on thresholds for restricted distribution, we demonstrated that the scale of assessment has a marked impact on the assigned threat status (Appendix).

Communities are typically mapped as grids (rasters) or polygons, both of which are representations of distribution with explicit or implicit scales of resolution and uncertainty (Keith in press). Comparisons of AOO estimates against fixed thresholds are sensitive to the spatial scale of measurement, irrespective of whether the estimates were derived from grids or polygons. The use of grids to scale estimates of AOO has been advocated in species assessment (Hartley & Kunin 2003; IUCN 2008), as their geometrical properties ensure a consistent means of standardization that can be applied to different map scales and types. The spatial scale of polygons can be quantified (for example by the size distribution of polygons or the segment lengths of polygons; IUCN 2008), but resolution may vary across a map in ways that are more difficult to test and define than grids. The development of robust and consistent criteria and valid standardization methods for assessing distributions of communities against fixed thresholds will require a more thorough investigation of the impact of scale on assessment outcomes, based on communities classified and mapped at different thematic and spatial resolutions, using both grid and polygon approaches.

Threat status of ecological communities

Although assessments will be prone to significant biases if the distribution sizes of communities are assessed at different spatial scales (see Appendix), few of the protocols address this problem by specifying a scale for measuring area. NatureServe recommends 2 km grid cells for measuring area of occupancy for species, after IUCN (2008), but "for ecological communities estimates of absolute area are preferred for area of occupancy, given the greater accuracy in mapping stands" (Master et al. 2007); the issues of implicit scale and uncertainty in using raw polygon data to estimate 'absolute area', and the implications of comparing the estimates with fixed thresholds, are not discussed. Assessments under the NSW TSC Act also follow IUCN (2008) guidelines for measuring area. Some of the other protocols recommend a range of mapping scales that are considered appropriate for estimating distributions (e.g. EPBC Act; Benson 2006), but fail to give explicit guidance about the standard resolution necessary to avoid scale-related bias when using such maps to estimate areas of occupancy.

Changes to ecological function

All of the protocols explicitly or implicitly address a reduction in ecological function across the extent of the community (Table 2), with the exception of Walker et al.(2006). Changes to ecological function, including composition, structure and processes, are difficult to quantify in ways that are sufficiently general to apply to a broad range of communities. Consequently, most of the protocols we reviewed used qualitative criteria, with the exception of some protocols that include quantitative thresholds for changes in species composition and fragmentation (below; Table 2). The EPBC Act, NatureServe, Noss et al. (1995) and Benson (2006) quantify thresholds for the proportion of the distribution that is degraded, but the nature and level of degradation that must be considered in such a calculation either remains qualitative or is undefined.

Changes to species composition are generally treated qualitatively (Table 2). Most protocols consider the loss of indigenous species and the degree of invasion by exotic species as key factors. The EPBC Act and Benson (2006) also include as a quantitative criterion the population viability of key native species that "are likely to play a major role in community processes…[and]…whose removal has the potential to precipitate change in community structure or function sufficient to lead to the community's eventual extinction". Examples include keystone species or dominant species that provide structure or much of the biomass of the community. Paal (1998) considers that "if the communities of a particular type are habitats for rare or endangered animals, that type must be accepted as threatened". This approach may create difficulties when threatened species are sparsely distributed across a wide range of communities, which may not be otherwise threatened, or when unthreatened communities provide habitat for species imperiled for other reasons (e.g. overharvesting).

Changes in structure can be divided into vertical and horizontal components (Franklin et al. 2002). Changes in vertical distribution of community biomass or loss of structural elements are considered qualitatively by some protocols (NSW TSC Act; WA TEC; Austrian Red List; Benson 2006; Raunio et al. 2008). Large changes to horizontal structure may result in fragmentation, restricting movement of organisms between habitat patches. Fragmentation is treated qualitatively by many protocols (Table 2), and quantitatively by the EPBC Act, NSW TSC Act and Rodriguez et al. (2007; Table 3). However, most protocols assess the current spatial configuration of patches (e.g. proportion of patches below a given area), rather than the fragmentation process (e.g. decline in median patch size), and therefore fail to account for variation between communities in the 'natural' spatial dispersion of patches. Quantitative measures of fragmentation include typical patch size (e.g. EPBC Act; NSW TSC Act; Table 4), distances between patches, and the proportion of patches under a given size, or combinations of the number, size, and isolation of patches (see Rodriguez et al. 2007; Table 3). However, like area of occupancy, these parameters are sensitive to the resolution of the spatial data from which they are estimated; no explicit guidance is provided on appropriate scales for estimating the fragmentation parameters or how to relate these to natural spatial patterns.

Criteria relating to changes to processes are descriptive, and include species interactions such as pollination or predation (NSW TSC Act; Rodriguez et al. 2007), and abiotic processes such as degradation of soils, changes in nutrient levels or flows (NSW TSC Act; WA TEC; QVM Act; Benson 2006; Raunio et al. 2008), and changes in fire and flood regimes (EPBC Act; NSW TSC Act; WA TEC; Noss et al. 1995; Benson 2006; Raunio et al. 2008).

A further factor assessed in some protocols is the capacity for restoration or recovery, with or without active intervention (Table 2). Rodriguez et al. (2007) argue that inclusion of factors such as recoverability and degree of legal protection are important for priority-setting, but are not directly relevant to risk, and as such should not be included in assessment criteria. Similar thinking underpins the exclusion of such concepts from the IUCN assessment protocol for species (Mace et al. 2008).

Discussion

A framework for assessing the threat status of communities

The IUCN protocol for assessing the threat status of species aims to capture different aspects of threat by using multiple criteria designed to accommodate a wide range of species life histories. Similarly, any community assessment protocol must aim to accommodate different threatening processes and be applicable to communities with a wide range of distributional, structural, and functional characteristics. Under the IUCN protocol, species are assessed against multiple criteria and assigned an overall threat category equal to the criterion with the highest threat category. Criteria may combine two or more measures of risk, such as area of occupancy and decline, where these combinations represent synergies between different sources of threat. This structure allows decisions to be made when data on other criteria are scarce or uncertain, and does not require any assumptions about interactions between criteria. For the same reasons and to maintain consistency with the IUCN Red List, such a decision structure can be readily adapted to assess community status, as has been done by most of the protocols reviewed. Decline in distribution and restricted distribution, based on area of occupancy and extent of occurrence, provide a sound starting point for assessing community status; these data can sometimes be estimated using remote sensing, historical records, or spatial modeling methods.

Given the extent to which natural disturbance regimes have been disrupted and species have been transported across the globe, the degradation (as opposed to outright conversion or destruction) of communities is a very real threat to biodiversity, although it remains difficult to incorporate quantitatively in assessment protocols. One solution is to develop a series of sub-criteria for specific types of degradation that are based in ecological theory and can be incorporated into a framework for assessment. Because every type of degradation or disruption to function for each community cannot be listed as a sub-criterion with quantitative thresholds, these sub-criteria will inevitably be incomplete proxies of function and composition and remain to some extent reliant on expert judgment (see Table 4 for examples from listings under the NSW TSC Act). A useful starting framework for assessing change to ecological function is provided by NatureServe's method for characterizing direct threats to species and communities, based on severity (degree of degradation), immediacy (timeframe), and scope (spatial extent). It demonstrates how semi-quantitative and qualitative criteria can be set in a rigorous and transparent structure, especially when guided by examples. For example, rule sets could identify communities as critically endangered if the decline in function is of high severity and scope, or as endangered if the decline in function is of high severity and at least moderate scope, or of moderate severity and high scope, etc. To reduce linguistic uncertainties, terms such as 'high' or 'moderate' need to be explicitly defined (Akçakaya et al. 2000; Regan et al. 2002) by using quantitative thresholds for scope wherever possible (e.g., 'high' scope could be defined as ≥70% of the distribution; NatureServe). Setting quantitative thresholds for severity is more complex, but may be feasible in many cases. For example, several quantitative parameters already exist for fragmentation (Table 4 from NSW TSC Act listings; EPPC Act; Rodriguez et al., 2007). For the development of criteria and thresholds relating to changes in structure, composition and processes, we can draw on the extensive research into classifying threats to biodiversity and assessing condition (see McCarthy et al.

2004; Gibbons & Freudenberger 2006; Keith & Gorrod 2006; Rouget et al. 2006; Salafsky et al. 2008).

Setting thresholds for decline, distribution and function

We found that there were great discrepancies in the thresholds for restricted distribution and decline in distribution in the protocols we reviewed. These differences in thresholds have major implications for the outcome of assessments, as illustrated in Figure 3: far fewer communities would be eligible for listing as Endangered under the EPBC Act than under the NSW TSC Act simply because those protocols adopt different thresholds for delimiting threat categories. Few of the protocols justify their selection of thresholds other than by referring to non-linear patterns of species loss with reduction in habitat area (e.g. EPBC Act, Walker et al. 2005; Benson 2006; Rodriguez et al. 2007), such as species-area relationships (Rosenzweig 1995) and extinction thresholds associated with fragmentation (e.g. Andrén 1994; Fahrig 2002). While the thresholds may reflect different 'best estimates' for proxies of extinction risk, the underlying reasons for variation in thresholds may be related to differences in implicit spatial scales, as well as socio-political considerations that limit the acceptable number of communities that 'should' be listed.

We also found that protocols that apply different thresholds and criteria for species versus communities (EPBC Act, WA TEC and QVM Act) tended to require greater declines and more restricted distributions for communities than species to be listed in equivalent threat categories. Although the threshold values specified in the EPBC Act differ greatly from their thresholds for species, the threat categories are intended to approximate the same risk of extinction; for example, an Endangered species or community has an extinction risk of at least 20% in the near future (EPBC Reg. 2000a).

Threat status of ecological communities

These findings raise the question: how should thresholds for decline and distribution of communities be defined, and should they be lower, the same, or higher than those for species in order to represent similar levels of extinction risk? The answer depends on the intended role of ecological communities in conservation planning. We address this question based on the key reasons for assessing the threat status of communities that we outlined in the introduction: (1) to assess the risk that communities and associated functions will be lost; (2) to provide a relative ranking of threat level of communities for priority setting; and (3) to act as a proxy for assessing the extinction risk of unknown or poorly-understood species.

Species-based criteria are unlikely to provide an adequate assessment of the natural processes and functions that conservationists hope to protect by protecting ecological communities. Some of these functions, such as fire and other disturbance regimes, require larger areas to maintain community viability and integrity than would be required to maintain viable populations of individual species (Baker 1992); others, such as microbial breakdown of leaf litter, occur at micro scales (Begon et al. 2006). To develop appropriate thresholds for these components of community viability, the size distribution of key biotic and abiotic processes across a wide range of different community types must be examined, and the impacts of decline on these processes need to be carefully analyzed. To be consistent with the philosophy and goals of the IUCN Red list criteria, the threat categories for communities should reflect relative extinction risk based on ecological theory and our best scientific understanding. An alternative approach would be to devise a simple rank order based on arbitrary thresholds. For example, one could rank communities on the basis of metrics such as AOO or EOO, and arbitrarily classify the 5% with the smallest values as

Critically Endangered, the next 10% as Endangered, and the next 15% as Vulnerable. Such a method presupposes knowledge of how such metrics are distributed across communities, and, more importantly, it would skirt the important goal of creating an assessment protocol that reflects our best understanding of extinction risk, notwithstanding the difficulties in defining community extinction.

Ecological communities are sometimes used as surrogates (coarse filters) for undescribed or poorly-known species (Franklin 1993; Noss 1996). If we assume that those poorly-known species have habitat requirements at the same scale at which the community is defined, then lower thresholds for communities would result in a failure to list and protect them to the same level as better-known species. In addition, communities will usually exhibit greater spatial variability than individual species, as their composition varies from place to place, and therefore will require larger areas than species to represent their full diversity, suggesting that area thresholds for communities should be at least as large as those for species.

These considerations outlined above do not obviate the need to consider species thresholds when developing analogous criteria and thresholds for communities; rather, they emphasize that threats to communities and species may have different underlying causes. Above all, we advocate extensive research as the basis of the development of appropriate thresholds, with the goal of creating an assessment scheme that reflects our best understanding of the extinction risk of communities.

Dealing with issues of scale

Identifying appropriate thematic and spatial scales presents a serious challenge when defining and assessing communities. The spatial scale at which a community is described and mapped is not necessarily the same scale at which its distribution size should be estimated for assessment against quantitative thresholds. The scale of description needs to be biologically relevant and will reflect how a community is classified and mapped, while the assessment needs to be based on a consistent spatial resolution to allow equitable comparisons of all communities against the fixed thresholds. Consistency of assessment scale may appear to be a trivial point, but it has non-trivial impacts on assessment outcomes, which are highly sensitive to scale (see appendix, Keith et al. 2000; Hartley & Kunin 2003; IUCN 2008). A valid comparison of area estimates to fixed thresholds is contingent upon transformation of the estimates to a standard spatial scale to ensure consistency in assessing *relative* extinction risk, as recommended by IUCN (2008). Such scaling processes are lacking in the protocols we reviewed, except NatureServe and the NSW TSC Act.

The thematic scale or hierarchical level of community classification also influences the outcome of assessment. Narrowly defined, relatively homogeneous assemblages may meet thresholds more easily than coarsely defined heterogeneous assemblages (Kirkpatrick 1998), even when spatial scales are held constant. While classification scales may be quantified using measures of compositional dissimilarity (Whittaker 1978), these measures are specific to particular sets of sample data and are unlikely to provide a useful general tool for standardizing a thematic scale of assessment. In the absence of such tools, assessors need to be aware that particular protocols have an implicit range of thematic scales of community classification, and that the degree of imperilment of communities defined at finer and coarser scales may be over- or under-estimated respectively.

Threat status of ecological communities

Given the problems of scale, what is the best way to assess the threat status of communities that have been delineated under different thematic and spatial scales? One approach is to use flexible or qualitative thresholds for assessing communities. However, one of the key drivers behind the formalization and quantification of the IUCN criteria was the desire to ensure repeatability, transparency, and consistency across all species (Akçakaya et al. 2000; IUCN 2008; Mace et al. 2008). Setting different thresholds for different types of communities, or those classified or mapped using alternative methods, would create intractable problems (such as the number of sets of thresholds and communities and datasets to which they should apply), particularly for a global set of assessment criteria. Thus, a better approach may be to allow some flexibility in the thematic and spatial scales at which communities are defined, while ensuring a consistent spatial scale for the assessment of their threat status. A limited amount of flexibility in assessing communities across multiple levels of a classification hierarchy allows a parsimonious approach: the broadest possible communities that meet the criteria at a defined spatial scale can be listed, rather than listing many fine-scale communities that occur within the broader one and have similar levels of threat (Figure 4; Keith in press). A parsimonious approach has advantages in producing fewer, broader communities that may be easier to identify compared to large numbers of subtly different assemblages. A smaller number of communities is also simpler to manage in terms of conservation planning and regulatory systems. However, too much flexibility may result in communities so broad that they never meet assessment thresholds or, conversely, so narrow that they always qualify for threatened status. It is therefore important that assessment protocols are explicit about their scales of definition and assessment.

Conclusion

Notwithstanding the many uncertainties and obstacles inherent in developing assessment protocols for communities, in areas where communities have been listed, there has been a surprising acceptance of the process and its inherent uncertainties (for example, in NSW, Preston & Adam 2004b, a; Keith in press). This suggests that it is possible to move beyond the uncertainties and theoretical debates, and to develop rigorous, consistent, and transparent criteria for assessing the status of communities. There may be no way to fully resolve some of the most vexing issues—in particular, the fundamental question of what constitutes community extinction—but it should be possible to make enough progress to produce something comparable in scope, utility, and rigor to the IUCN Red List criteria. The characteristics of such a protocol are likely to include: (1) definitions of key terms and parameters that are explicit, yet carry sufficient generality to enable flexible and wide application of the protocol; (2) numerical thresholds that delimit different categories of threat, with a focus on decline in distribution, current area of the distribution, and decline in ecological function of a community; (3) threshold levels that are logically consistent with viability of component species and ecological functions at landscape scales; (4) criteria that incorporate more than one parameter to address synergies between different types of threats; (5) methods of area estimation that minimize bias due to differences in spatial and thematic scales; and (6) workable and effective proxies for assessing the severity, scope, and immediacy of declines in the ecological functions of communities.

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Tables

Table 1. Protocols for assessing the threat status of communities. *Application* denotes where and how the protocol has been applied (protocols labeled 'research', to our knowledge, have not been implemented by governments or NGOs). *Definition of community* and *Definition of extinction* refer to the definitions of these terms employed in each protocol. For *Decision method & categories* we focus on Critically Endangered CR, Endangered EN, and Vulnerable VU, or their equivalents, given in brackets.

Protocol and key references	***Application***	***Definition of community***	***Definition of extinction***	***Decision method & categories***	***Species analogue***
EPBC Act: Australian Environmental Protection and Biodiversity Act (EPBC Act 1999; EPBC Reg. 2000b)	Key conservation legislation applied by the Australian Federal Government across the country (continent)	**Ecological community**: an assemblage of native species that … inhabits a particular area in nature	An ecological community may be considered effectively extinct when all representatives of the community have undergone an irreversible loss of integrity: when re-establishment of ecological processes, species composition and community structure within the range of variability exhibited by the original community is unlikely within the foreseeable future, even with positive human intervention.	Rule set; CR, EN, VU	IUCN Criteria
NSW TSC Act: New South Wales Threatened Species Conservation Act (TSC Act 1995; TSC Amendment Act 2005)	Key conservation legislation applied by State Government of New South Wales, Australia, for the protection of species and communities	**Ecological community**: an assemblage of species occupying a particular area.	No definition provided	Rule set; CR, EN, (qualitative for VU)	IUCN Criteria
WA TEC : Western Australian List of Threatened Ecological Communities (English & Blyth 1999; DEC 2007)	State government of Western Australia; informal listing that guides protective measures under environmental impact legislation, as there is	**Ecological community**: a naturally occurring biological assemblage that occurs in a particular type of habitat; an assemblage is a defined group of biological entities.	**Presumed Totally Destroyed**: An ecological community which has been adequately searched for but for which no representative occurrences have been located. The community has been found to be totally destroyed or so extensively modified throughout its range that no	Rule set; CR, EN, VU	IUCN Criteria

	currently no legislation for threatened ecological communities		occurrence of it is likely to recover its species composition and/or structure in the foreseeable future.		
QVM Act: Queensland Vegetation Management Act (1999); Sattler & Williams (1999)	Key legislation for the control of land clearance applied by the State Government of Queensland, Australia	**Regional ecosystem***:* a vegetation community in a bioregion that is consistently associated with a particular combination of geology, landform and soil, defined by Sattler and Williams (1999)	No definition provided	Rule set; Endangered (EN), Of Concern (VU)	Based on IUCN, no CR category
Austrian Biotope Red list (Essl et al. 2002)	National government Red List	**Biotope**: Habitat of a regularly reoccurring species community of a particular minimal size specific to the type of biotope and homogenous composition that can be distinguished from the surrounding environment.	**Extinct**: When a biotope can no longer be shown to exist, when loss or destruction has damaged the original biotope beyond recognition	Decision tree; CR, EN, VU	Zulka et al. (2001)
NatureServe conservation status assessment system (Faber-Langendoen et al. 2007; Master et al. 2007)	Developed by NatureServe, applied by NGOs primarily in the Americas, especially the USA & Canada, at global, national and state/province levels	**Ecological Communities** or **Vegetation Types** are assemblages of species and growth forms that co-occur in defined habitats at certain times and that have the potential to interact with each other. They are classified at multiple scales, from formations (biomes) to alliances and associations, based on the International Vegetation Classification (Grossman et al. 1998)	**Eliminated** (ecological communities)—Eliminated throughout its range, with no restoration potential due to extinction of dominant or characteristic species. **Presumed Eliminated** (Historic, ecological communities) —Presumed eliminated throughout its range, with no or virtually no likelihood that it will be rediscovered, but with the potential for restoration, for example, American Chestnut (Forest).	Combinatio n of decision rules & score tallied from all criteria; G1 (CR), G2, (EN), G3 (VU)	Same criteria & thresholds (where applicable) for species
Noss et al. (1995)	Research; once-off assessment of the status of ecosystems in the USA	"The term **ecosystem** is generally used to denote a community of all the species populations that occupy a given area and its nonliving environment… An ecosystem can be a vegetation type, a plant association, a natural community, or a habitat defined by floristics, structure, age, geography, condition, or other ecologically relevant factors."	"Ecosystems can be lost or impoverished in basically two ways. The most obvious kind of loss is quantitative – the conversion of a native prairie to a corn field or to a parking lot. Quantitative losses, in principle, can be measured easily by a decline in areal extent of a discrete ecosystem type (i.e., one that can be mapped). The second kind of loss is qualitative and involves a change or degradation in the structure, function, or composition of an ecosystem. At some level of degradation, an ecosystem ceases to be natural."	Rule set; CR, EN, Threatened (VU)	
Paal (1998)	Research; proposed criteria for assessing the status of	**Plant community** or **vegetation type**: no definition provided.	**Extinct or probably extinct**: Communities that are no longer known to exist in the wild within the territory of the republic after	Rule set; Very threatened	IUCN criteria are used in

	communities in Estonia		repeated searches.	(CR), Threatened (EN), fairly threatened (VU)	Estonia
Benson (2006)	Research; proposed criteria for assessing the communities of NSW	**Plant/vegetation community**: no definition provided, but describes the process of classification used to describe and assess communities (Benson 2006; Benson et al. 2006).	An ecological community is eligible to be included in the **presumed extinct** category if it has been totally destroyed, or so modified throughout its range, that it is unlikely to recover its species composition and/or structure in the very long term (the next 200 years, or within 40 generations of any long-lived species believed to play a major role in sustaining the community, whichever is the longer up to a maximum of 800 years)	Rule set; CR, EN, VU	
Walker et al. (2005; 2006)	Research: once-off assessment of the status of indigenous Land Environments of New Zealand (LENZ, Leathwick et al. 2003); LENZ Levels II (100 units) and IV (500 units) were assessed.	Land Environments of New Zealand (LENZ) is a hierarchical classification of New Zealand's abiotic terrestrial environmental pattern based on 15 environmental climate and landform variables likely to influence species distributions (Leathwick et al. 2003).	No definition provided	Rule set; Acutely Threatened (CR); Chronically Threatened (EN); At Risk (VU)	Townsend et al. (2008)
Rodriguez et al. (2007)	Research; proposed global criteria for assessing community status; case studies with current extents range from ~200 to ~100,000 km^2	**Ecosystem**: "…although we prefer the traditional 'unit of biological organization that encompasses a unique and relatively homogeneous composition of species and abiotic elements and their dynamic processes', we emphasize that a universal definition is not necessary for successful classification… in practice, more simplified and practical definitions (based on delimiting ground-truthed areas with similar spectral properties) may be more readily applicable…"	"Using remotely sensed data, we consider an ecosystem extinct if no intact land cover of the original ecosystem exists. Some components of the ecosystem may still exist, but the way in which all original components were organized no longer does."	Rule set; CR, EN, VU	
Raunio et al. (2008)	Research; proposed criteria for assessing	**Habitat type**: a spatially definable land or aquatic area with characteristic	"The habitat type no longer occurs in the observed area" due to decline in	Decision tree; CR,	

	habitat types in Finland; 381 habitat types and complexes were assessed	environmental conditions and biota which are similar between these areas but differ from areas of other habitat types. The environmental factors include e.g. soil, climate, and topography. The characteristics of the biota include the composition of typical species and their relative abundances.	distribution; or when "an occurrence of a given habitat type is disappeared when it no longer represents that habitat type" due to gradual qualitative change.	EN, VU

Table 2: Criteria used in the assessment protocols. Some include quantitative thresholds while others are qualitative (in brackets).

Criteria & sub-criteria	***EPBC Act***	***NSW TSC Act***	***WA TEC***	***QVM Act***	***Austrian Red list***	***Nature Serve***	***Noss et al. (1995)***	***Paal (1998)***	***Benson (2006)***	***Walker et al (2006)***	***Rodriguez et al. (2007)***	***Raunio et al. (2008)***
Decline in geographic distribution (Figure 1):	✓	✓	✓	✓	✓	✓	✓	(✓)	✓	✓	✓	✓
conditional on continuing threat			✓								✓	
short-term rate of decline	✓	✓	✓		✓	✓			✓		✓	✓
Restricted geographic distribution (Figure 2):	✓	✓	(✓)	✓	(✓)	✓		✓	✓		✓	(✓)
area of occupancy	✓	✓		✓		✓		✓	✓		✓	
extent of occurrence	✓	✓				✓						
number of occurrences		✓				✓		✓			✓	
conditional on continuing threat	✓	✓	✓						✓			
Ecological function, including:	✓	✓	✓	✓	(✓)	✓	(✓)	✓	✓		✓	(✓)
species composition	✓	(✓)	(✓)	(✓)	(✓)	(✓)	(✓)	✓	✓			(✓)
fragmentation	✓	✓	(✓)			(✓)					✓	
capacity for recovery	✓		✓	✓		(✓)			✓			
Risk of extinction	✓								✓			

Table 3. Criterion C: Pattern of fragmentation applied by Rodriguez et al. (2007).

	Risk category		
	CR	***EN***	***VU***
Proportion of extent in fragments of <10 km2 AND 90% fragments >1 km from nearest neighbor	>90%	>70%	>30%
Proportion of extent in fragments of <10 km2 AND 70% fragments >1 km from nearest neighbor		>90%	>70%
Proportion of extent in fragments of <10 km2 AND 30% fragments >1 km from nearest neighbor			>90%

Table 4. Examples of proxy parameters for assessing declines in ecological function in two ecological communities from NSW listed as Endangered under the NSW TSC Act: Coolibah – Black Box Woodland, a flood-prone semi-arid floodplain community; and Cumberland Plain Woodland, a fire-prone temperate grassy woodland community of coastal lowlands west of Sydney. Data extracted from http://www.environment.nsw.gov.au/committee/ListOfScientificCommitteeDeterminations.htm

Process	***Parameter***	***Coolibah – Black Box Woodland***	***Cumberland Plain Woodland***
Fragmentation			
	Number of patches	Increased by 70% during 1998-2004	Increased to 1857 during 1998-2007
	Median size of patches	Declined by 17% to 60 ha during 1998-2004	Declined by 57% to 1.3 ha during 1998-2007
Change in community structure			
	Proportion of distribution affected by change in structure	Tree poisoning and ringbarking over at least 25% of the portion of distribution for which this form of degradation was mapped	Density of old growth trees declined to approx. one per 200 ha since settlement in sampled area
Change in species composition			
	Number of presumed extirpations	-	>30% of mammal fauna extirpated since settlement
	Number of species declining	29 vertebrate species listed as threatened occur within the community	28 vertebrate, one invertebrate & seven plant species listed as threatened occur within the community
	Proportional change in compositional resemblance	-	-
Biological invasion			
	Proportional biomass, abundance or cover of invasive species	Introduced herb, *Phyla canescens* occupies 25-35% of groundcover where present	Introduced shrub, *Olea africana*, covers 10% of the community's distribution at densities detectable on air photos and detected in 43% of sampled sites
	Rate of increase in biomass, abundance or cover	Introduced herb, *Phyla canescens* invaded 8000 ha during 1996-2005	Introduced shrub, *Olea africana*, expanded across c. 1000 ha of woodland since 1970's
Change in disturbance regimes affecting species life histories, resource cycling, etc.			

Change in intensity/magnitude of disturbance	Magnitude of floods with a recurrence interval of two years reduced by 34-61%	-
	Median annual flow reduced by 44% since water regulation	-
Change in frequency of disturbance	Flood frequency reduced by 30% in sampled catchment during 1988-2000	Fire return interval increased to greater than 4-12 years over most of distribution
	Flood duration reduced by 30% in sampled catchment during 1988-2000	-

Figure captions

Figure 1. Thresholds for the percent decline in distribution used to the threat status of communities under ten of the protocols. Because NatureServe assesses community status against the thresholds within a point-and-rule system, NatureServe thresholds are not included in the figure.

Notes: * Protocols require demonstrated on-going threat to qualify under this criterion;

^ QLD VM, WA TEC and the Austrian Red List do not include quantitative thresholds for all threat categories.

Figure 2. Thresholds for the sub-criteria for restricted distribution: (a) area of occupancy; (b) extent of occurrence. Because NatureServe assesses community status against the thresholds within a point-and-rule system, NatureServe thresholds are not included in this figure.

Notes: * protocols require ongoing threat to qualify under this criterion;

^ in the QVM Act, communities qualify under restricted distribution only if also meeting specified decline thresholds;

[a] Rodriguez et al (2007) do not specify a strict area of occupancy thresholds, but rather that the entire geographical distribution is comprised of one (CR), three (EN) or 10 (VU) or fewer fragments of less than 10 km^2; this is a measure of area of occupancy (IUCN 2001) measured at a scale specified by 10 km^2 blocks, though it can also be interpreted as a measure of the number of occurrences (Regan et al. 2004), where an occurrence may be up to 10 km^2.

Figure 3. The percent decline in distribution (x-axis) plotted against current distribution size (measured by the area of occupancy), for 183 ecological communities from southeast NSW (Tozer et al. 2006). AOO was estimated with 2 km grid cells. The pale grey area shows the thresholds for endangered used by the NSW TSC Act, based on the IUCN species thresholds (the threshold for area of occupancy is 500 km^2, and for decline in distribution is 50% where causes of threat are ongoing, poorly understood or irreversible, shown here, or 70% where threats are understood, ceased and reversible), while the thresholds for endangered under the Australian EPBC Act (1999) are shown in dark grey (thresholds of 90% for decline and 100 km^2 for area of occupancy). Far fewer communities would be listed under the EPBC Act than the NSW TSC Act. The points marked with circles are the five communities shown in Figure 5.

Figure 4. A hypothetical classification of ecological assemblages showing three alternative approaches for dealing with thematic scale in assessment. First, at a fixed fine scale of assessment, there are 13 communities, of which eight are threatened due to high rates of decline (black squares). Second, at a fixed coarse scale, there are five broader communities (I-V). Of these, communities II and IV are clearly threatened because all of the finer scale assemblages within them are declining at a rate exceeding the specified threshold (they contain only black squares), while community V is not threatened (contains only assemblage 13). However, the status of communities I and III is uncertain because they contain mixtures of threatened and non-threatened communities. Their

status could therefore be determined by an area-weighted average rate of decline. Finally, under a flexible scale of assessment, communities A, B, C and D are threatened (contain only black squares), while the remaining assemblages are not threatened. Note that communities A and C are identical to fine communities 1 and 9, respectively, while communities B and D are identical to broad communities II and IV, respectively. Flexibility in the scale of assessment will be more limited for assessments based on only distribution size criteria (cf. decline criteria). For example, the broad community IV would only qualify for threatened status under the area criterion if the combined areas of finer communities 11 and 12 did not exceed the area threshold, irrespective of whether the distributions of communities 11 and 12 are individually smaller than the threshold.

Figures

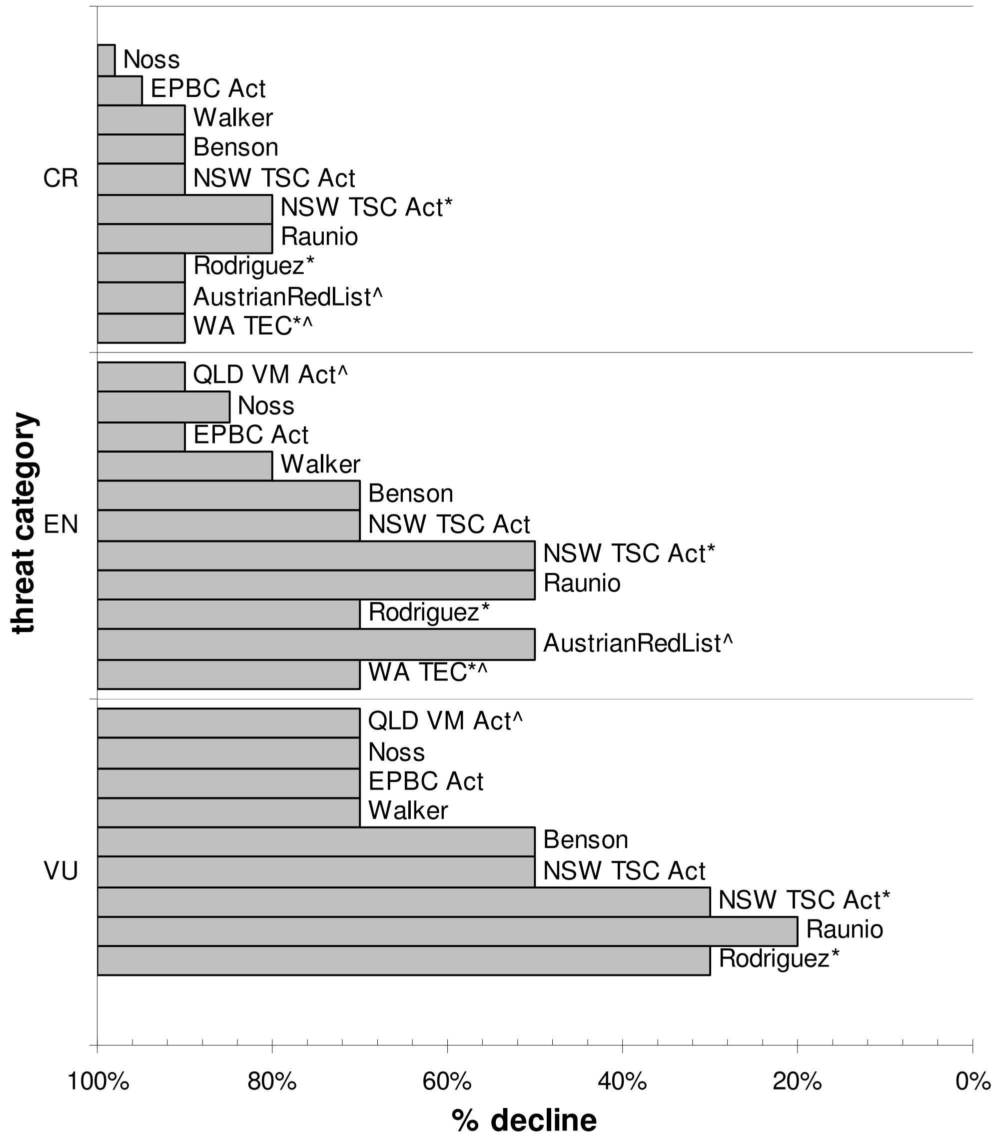


Figure 1.

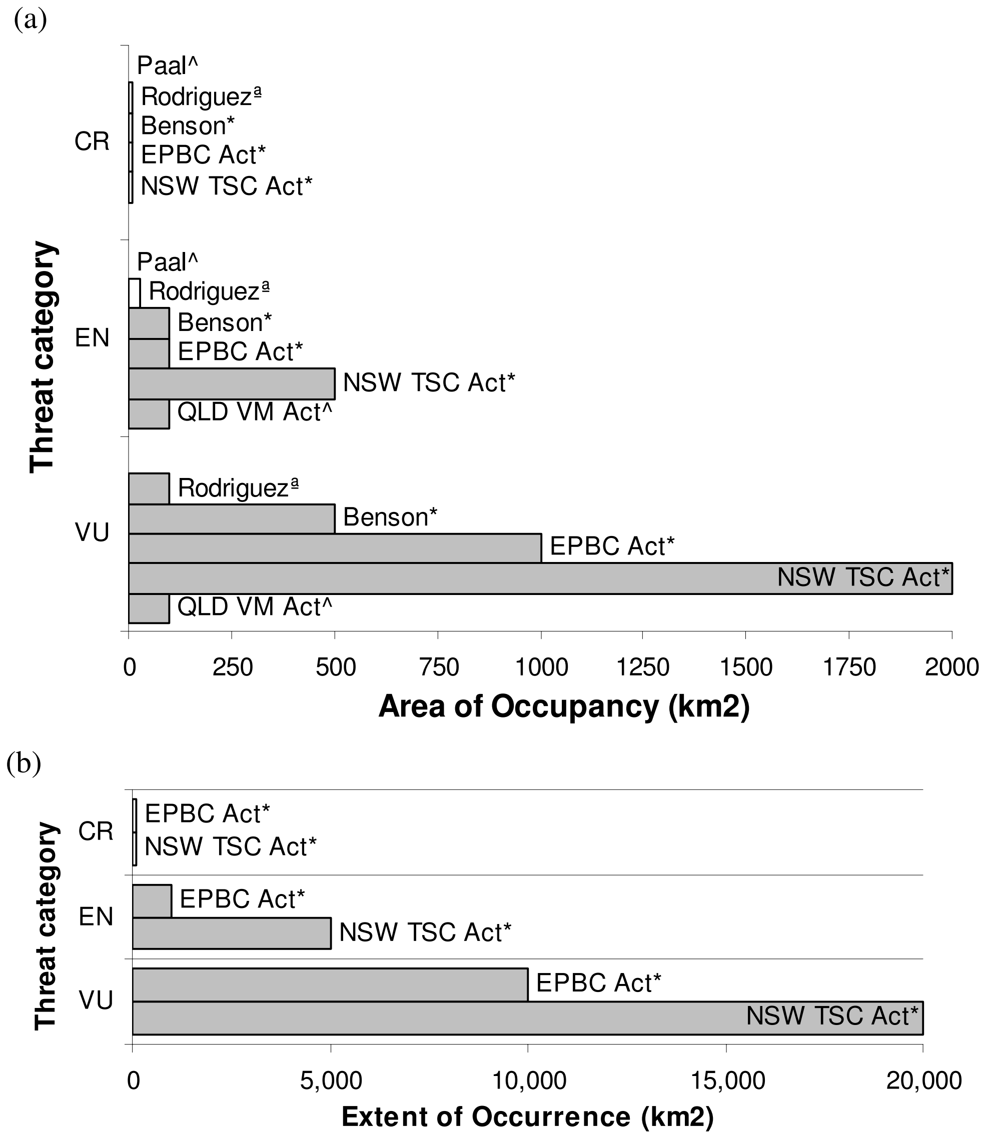

(a)
Paal^
Rodriguez[a]
Benson*
EPBC Act*
NSW TSC Act*
CR
Paal^
Rodriguez[a]
Benson*
EPBC Act*
NSW TSC Act*
QLD VM Act^
EN
Rodriguez[a]
Benson*
EPBC Act*
NSW TSC Act*
QLD VM Act^
VU
Threat category
0
250
500
750
1000
1250
1500
1750
2000
Area of Occupancy (km2)
(b)
EPBC Act*
NSW TSC Act*
CR
EPBC Act*
NSW TSC Act*
EN
EPBC Act*
NSW TSC Act*
VU
Threat category
0
5,000
10,000
15,000
20,000
Extent of Occurrence (km2)

Figure 2.

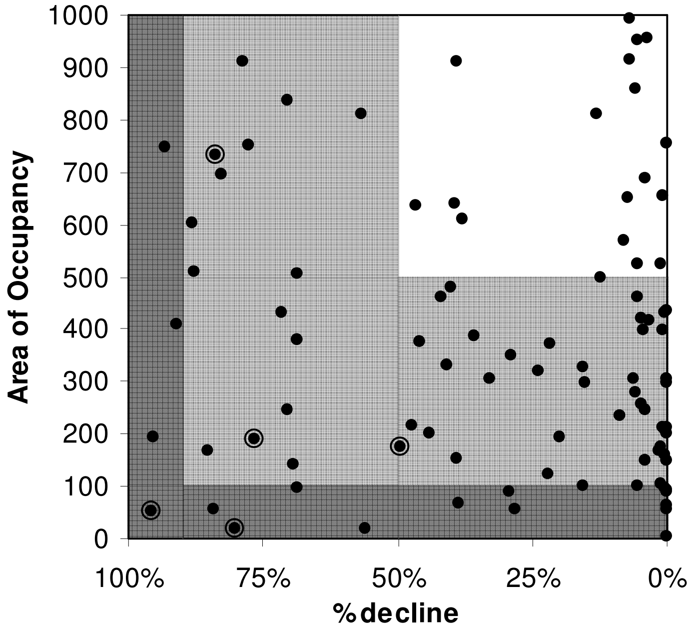
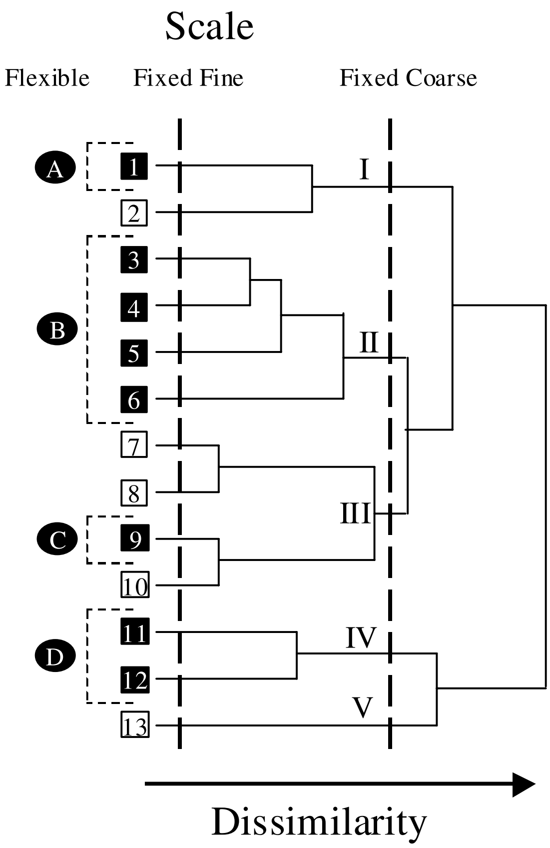

1000
900
800
700
600
500
400
300
200
100
0
Area of Occupancy
100%
75%
50%
25%
0%
% decline

Figure 3.


Scale
Flexible
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A
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1
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I
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III
IV
V
Dissimilarity

Figure 4.

Appendix

Area of occupancy (AOO) and similar measures of geographic distribution are highly sensitive to the scale of estimation (Hartley & Kunin 2003; IUCN 2008). To examine the influence of the spatial scale on assessment outcomes, we calculated AOO at different spatial scales for 183 ecological communities in southeast New South Wales, Australia (Tozer et al. 2006). These communities have been defined by a numerical analysis of quadrat data and mapped at 25m grid cell resolution within a region of 40,000 km^2. We calculated their AOO using a geometric progression of grid cell sizes, from 0.125 km to 16 km grid width, and assigned them to the appropriate threat categories using thresholds from the seven protocols that assess AOO.

For all communities, estimates of AOO increased with grid size, and consequently communities appeared to be less threatened at coarser scales of assessment (Table 5). Examples of scale-area curves for five communities are shown in Figure 5, along with the thresholds for CR and EN used in the NSW TSC Act and the EPBC Act. The communities showed varying sensitivity to scale of assessment, with all spanning at least two threat categories based on AOO. Scale of assessment therefore has a marked impact on the eligibility of a community for different threat categories, as has been found for species (Hartley & Kunin 2003; IUCN 2008).

Table 5. Number of communities meeting the thresholds for area of occupancy (AOO) specified in seven protocols at eight different spatial scales of assessment. Data from (Tozer et al. 2006) for 183 ecological communities in NSW, of which 77 extend beyond

the study region and have larger AOOs than data indicate. The threat categories are Critically Endangered (CR), Endangered (EN), and Vulnerable (VU) or their equivalents (Table 1).

Notes: 1. the QVM Act applies the same area threshold for distribution across threat categories combined with different percent decline thresholds; 2. the thresholds used by NatureServe form part of a point and rule based scoring system, rather than designating a threat category; 3. Paal (1998) has only two threat categories (very threatened, CR, and threatened, EN).

Protocol		***Scale of assessment (dimensions of grid cells in km)***							
		0.125	***0.25***	***0.5***	***1***	***2***	***4***	***8***	***16***
EPBC Act	CR	18	9	5	2	2	0	0	0
	EN	67	55	42	27	15	5	3	0
	VU	88	100	99	96	78	68	37	19
NSW TSC Act	CR	18	9	5	2	2	0	0	0
	EN	132	122	102	85	65	40	19	7
	VU	28	45	68	73	74	66	63	36
QVM Act[1]	EN/VU	87	66	49	30	18	6	4	1
NatureServe[2]	A	3	3	3	3	1	1	1	1
	B	4	3	0	0	2	0	0	0
	C	13	5	4	0	0	0	0	0
	D	16	12	4	3	2	0	0	0
	E	51	43	38	24	13	3	3	0
	F	66	68	61	58	50	16	16	7
Paal (1998)[3]	CR	0	0	0	0	0	0	0	0
	EN	2	1	1	2	0	0	0	0
Benson (2006)	CR	18	9	5	2	2	0	0	0
	EN	67	55	42	27	15	5	3	0
	VU	65	67	60	58	50	35	16	7
Rodriguez et al (2007)	CR	18	9	5	2	2	0	0	0
	EN	25	23	12	5	2	2	0	0
	VU	130	132	129	118	91	71	40	19

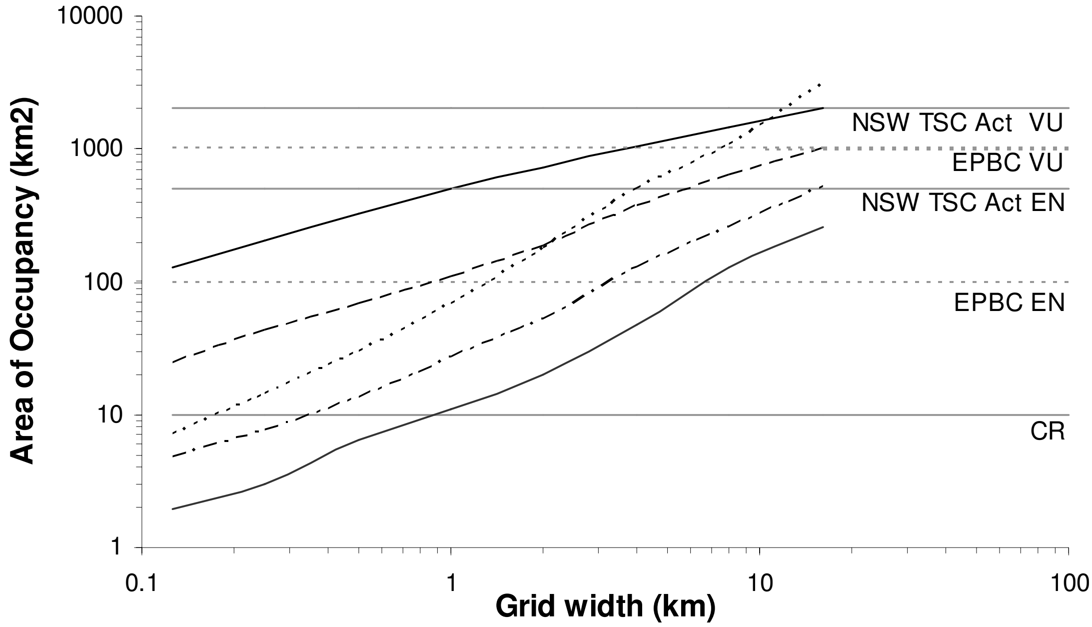

Figure 5. The impact of using different grid-cell sizes (x-axis) to estimate area of occupancy (AOO; y-axis) for five ecological communities listed as threatened in NSW (see Appendix 1; Tozer et al. 2006), and thresholds used to assign threat category under the NSW TSC Act and the EPBC Act; the protocols have the same thresholds for critically endangered (CR), but different thresholds for Endangered (EN) and Vulnerable (VU).

In press: *Ecological Management and Restoration*

The interpretation, assessment and conservation of ecological communities

By David A. Keith

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Summary Ecological communities are assemblages of species that occur together in space and time. Their properties include composition, structure, habitat, distribution, biological interactions and ecosystem functions. The community concept has a central role in conservation planning, and is a key approach for biodiversity conservation above the species level. The relatively recent application of risk assessment and regulatory systems to conservation of ecological communities has highlighted a number of challenges related to intrinsic uncertainties in the definition, diagnosis and assessment of ecological communities. In this review, I aim to elucidate some key conceptual issues essential to the interpretation of communities. Effective description, diagnosis and assessment of communities rests on an understanding of community theory in relation to environmental gradients and ecosystem dynamics. Continuum and discrete models can both contribute to interpretation of communities for conservation. Different sources of uncertainty are inherent in the key properties that characterise communities. While some of these are reducible, remaining uncertainty must be incorporated into assessments and decision-making processes for conservation. Protocols for assessing extinction risks of communities address rates of decline in distribution, size of distribution and rates of decline in ecological functions. Some protocols assess these factors in a manner that may be inconsistent with equivalent methods for assessing species. Communities may be viewed in a framework that distinguishes thematic spatial and temporal scales. These scales influence the outcomes of risk assessment, the benefits and limitations of maps and how well communities perform their function in conservation planning. When applied effectively, ecological communities can be powerful tools for delivering cost-effective outcomes for land use planning and biodiversity conservation.

Key words *biodiversity surrogate, conservation planning, conservation legislation, risk assessment, species assemblages, threatened ecosystems*

Introduction

The conservation of biodiversity is a widely accepted goal, as reflected in international agreements (United Nations Environment Programme 1993). In some countries including Australia, this is supported by national legislation (Commonwealth of Australia 1999), industry strategies (Minerals Council of Australia 2005) and community aspirations (WRI, IUCN & UNEP 1992). The goal is challenging, partly because biodiversity itself is complex – it comprises a very large number of components, which interact at multiple levels of organisation (e.g. genes, species and ecosystems). While species are fundamental units of currency in biodiversity, there is wide agreement that an exclusive focus on species conservation is unable to achieve the broader goal of conserving biodiversity (Noss 1987, 1996; Pressey 1999; Rodriguez *et al*. 2007). This stems partly from the inability of species-level approaches to deal with higher-order ecological processes and functions, and partly from the impracticality of individually accounting for and managing the huge numbers of species in existence, many of which are yet to be described (May 1988).

Ecological communities offer one of the most important approaches for conservation above the species level, alongside umbrella taxa, functional groups of species and threatening processes (Auld & Keith this volume). They provide a conceptual framework for generalisation about biodiversity, its composition, distribution and dynamics, which is essential for conservation (Austin & Margules 1985; Noss 1987; WRI, IUCN & UNEP 1992; Pressey 1999; Mac Nally *et al*. 2002). The community concept has made its way into conservation practice through applications in strategic planning (Master *et al*. 2002), regulatory systems (e.g. Commonwealth of Australia 1999; Republic of South Africa 2004) and management plans for protected areas (NSW Department of Environment and Conservation 2006; South African National Parks 2006). Experience from these applications has shown that conservation of ecological communities poses unique challenges.

In this review, I aim to elucidate some key issues in the interpretation and assessment of ecological communities. I first review how various regulatory instruments define and describe ecological communities with reference to examples from the scientific literature. I discuss how community theory and decision theory underpin sound application of the community concept to conservation, particularly through frameworks for dealing with environmental gradients, ecosystem dynamics and uncertainties from a variety of sources. I provide simple examples to illustrate how uncertainty about communities can be incorporated into conservation planning decisions. Thirdly, I review the approaches for assessing the status of ecological communities, drawing parallels with more developed methods for assessing extinction risks of species. Finally, I outline a conceptual framework for interpreting the scale-dependence of ecological communities and discuss the benefits and limitations of maps for representing their distributions. I focus on generic issues that are common to all applications, but draw primarily from jurisdictions in Australia, where a strong tradition of regional inventory and planning based on communities has emerged in recent decades, and where there are highly developed regulatory systems for conservation of communities as well as species.

Definitions of communities and related terms

Ecological communities are ‘assemblages of species populations that occur together in space and time’ (Begon *et al*. 2006). While Begon *et al*. (2006) simply refer to co-occurrence of species as the defining characteristic of communities, some authors emphasise the interactions that bind species together. Whittaker (1975), for example, defined a community as ‘an assemblage of interacting species, linked by their effects on one another and their responses to the environment’. In a broader sense, communities may be viewed as having six types of properties: composition, structure, habitat, distribution, interactions between their component species and ecological processes and functions (Table 1). The basic scientific definition of a community turns on only two of its six key properties (Begon *et al*. 2006): the composition of species that make up the assemblage; and the distribution of the assemblage in space and time. Any given community may have a characteristic structure or habitat, for example, that may help observers to identify it, but these are not its *defining* properties. This raises a distinguishing feature between communities and ecosystems. The latter represent a higher level of organisation that includes ‘the biological community together with the abiotic environment in which it is set’ (Begon *et al*. 2006). Three of the six properties in Table 1 (composition, distribution and habitat) are therefore defining features of ecosystems.

Both ‘communities’ and ‘ecosystems’ have made their way into regulatory systems for protection of biodiversity in various jurisdictions within Australia and overseas. The legal definitions of these terms deviate from the basic scientific definition to varying degrees (Table 2), although they appear to serve broadly similar regulatory applications. In these applications, the emphasis of community circumscription has been on plant components of the biota. While animals are sometimes recognised as components of the community, they are rarely identified as defining features and conservation of animal communities (*cf*. species) is not a well-developed component of planning systems. Hence, examples of protected communities of animals are rare (e.g. NSW Scientific Committee 1998). None of the legal definitions mention the temporal aspect of species’ co-occurrence (Table 2), presumably because conservation actions are usually concerned with communities that occur at the present time, rather than those of the past or future (e.g. Hunter *et al*. 1988; Hobbs *et al*. 2006).

In some legislative applications, the scientific definitions of communities or ecosystems are loaded with additional terms such as ‘integrated group’, ‘dynamic complex’ and ‘functional unit’ (Table 2). While these terms themselves are undefined in the legislation, they imply an emphasis on species interactions as a defining feature (after Whittaker 1975). Although interactions are an important property of communities (Table 1), they may be somewhat redundant as a defining feature and impose unnecessary difficulties in practical applications of the community concept. For instance, it is difficult to image that a group of species could co-occur without some of them interacting in some way, yet also difficult to define and demonstrate specific interactions in order to meet a regulatory requirement or prove compliance with a legal definition of a community that included reference to interactions. A lack of evidence or knowledge about specific interactions between co-occurring species is not evidence that no interactions exist.

Other legal definitions are less directly linked to the scientific definition (e.g. State of Queensland 1999). The European Union’s Habitat Directive (Council of

European Communities 1992) addresses the protection of 'natural habitat types'. Although defined more broadly in the legislation, listed 'natural habitat types' serve a similar conservation function to 'communities' and 'ecosystems' in other jurisdictions and are described primarily in terms of characteristic plant species that comprise an assemblage (European Commission 2007). In contrast, acts of parliament in the USA (1973) and Canada (2002) define the term 'critical habitat', exclusively in relation to habitats necessary for survival or recovery of listed individual species. Neither the US nor Canadian legislation has any provisions for explicit protection of assemblages of species, whereas the Republic of South Africa (2004) has provisions for protection of listed ecosystems but no direct protection for individual species.

Description of communities: determinations and listing statements

There is no standard scientific protocol to describe ecological communities, although case studies (Keith & Bedward 1999; Mucina & Gledenhuys 2002) and general toolkits provide some guidance on collection of appropriate data and classification approaches (Kent & Corker 1992, Hnatiuk *et al.* 2008). Instead, descriptions tend to be shaped for specific purposes and intended audiences, as well as by constraints imposed by available data, publisher requirements, etc. Consequently, there is great variation in the format and content of community descriptions in scientific literature. Some descriptions having brief short-hand accounts of salient features (e.g. Beard & Webb 1974; Pickard & Norris 1994), while others include detailed commentary (e.g. Beadle 1948), quantitative estimates of species frequency and local abundance, structural and environmental variables (Keith & Bedward 1999), and/or pictorial and map-based information (Davies *et al*. 2002). Some descriptions may also address issues related to the conservation status of communities (e.g. Benson *et al*. 2006). There are few examples that describe all six properties of ecological communities (Table 1). Most descriptions address characteristic species composition, structure and/or habitat. Relatively few address species interactions or other ecological processes and functions, although see comprehensive treatments of Beadle (1948), Keith (2004) and Mucina & Rutherford (2006) for exceptions. The generality of these treatments reflects the patchy coverage of information on interactions and processes at finer scales of classification.

In regulatory applications, such as biodiversity legislation, descriptions of ecological communities may be codified into listing statements or determinations that support regulatory, conservation and recovery operations, as well as legal actions. These listing statements have multiple roles. First, they provide for scientific diagnosis of communities, and thus identify instances where regulatory protocols and conservation actions are triggered. Second, they may be required to articulate a scientific justification for listing of the communities. Third, they are legal documents that support compliance, enforcement and prosecution under laws that protect the listed communities. Fourth, they are a means of alerting the public to the existence of threatened communities, and hence the need to seek specialist advice to confirm diagnosis of the community and to determine how laws and regulations may apply to management of particular land parcels.

The multiple roles of listing statements may create tensions and trade-offs in the content and form of community descriptions that they contain. For example, a simplified expression of key features designed to inform members of the public who may lack formal scientific training might not include sufficient detail to support a rigorous scientific diagnosis of a community. Another trade-off relevant to listing statements involves the flexibility of community descriptions. Flexibility is essential in the description of a community to accommodate the natural variability in its properties (Preston & Adam 2004). Too much flexibility may be seen as precluding identification of a community with sufficient certainty to support regulatory and legal actions, but an overly prescriptive description may fail to identify many examples of the species assemblage that the listing is designed to protect (Regan *et al*. 2002). It is therefore important to recognise uncertainty as an intrinsic characteristic of community description and diagnosis, reduce it where possible, and deal with it explicitly in any decision-making processes. Case law in New South Wales has addressed this practical necessity on several occasions. In rejecting a challenge to the validity of a listing of an Endangered Ecological Community[1], Chief Justice Spigelman of the Supreme Court in NSW stated,

> *The use of the word 'assemblage' does not suggest that either the nomination of species or identification of an area requires a high degree of specificity… The intricacy of all ecological communities means that some indeterminateness is bound to arise from the form of expression used to describe them.*

Applying ecological communities in conservation practice

Applying the community concept to any biodiversity conservation problem poses a number of basic questions. Does an observed assemblage belong to a particular community of interest? Where are the boundaries of this community? What is its conservation status? Where and how can it be protected? How will it be affected by future events, such as particular developments, fire, climate change, etc.? The diagnoses and assessments of communities required to answer these questions are specialist tasks; comparable to diagnoses of medical conditions, legal opinions on contracts, standards for mechanical and building construction, auditing of accounts, etc. In these disciplines, one may be more inclined to trust the advice of experienced professionals than untrained amateurs. However, diagnoses and assessments will usually be based on imperfect information and will therefore entail some risk of error, irrespective of who gives the advice. In this context, a sound understanding of community theory and the sources of uncertainty are essential pre-requisites for robust and accurate interpretations of communities for conservation practice.

Community theory

A strident theoretical debate about the nature and properties of ecological communities took place in scientific forums, primarily between the 1920s and the 1980s (Burrows 1990). In its simplest form, the debate focussed on two opposing paradigms. Clements (1916) proposed a discrete model of the community, drawing an analogy to a superorganism, in which the occurrences of component species were tightly bound together in both space and time, and hence delimited by sharp boundaries. Gleason (1926), on the other hand, proposed a continuum model, in which co-occurrence of species was simply the result of similarities in their

[1] VAW (Kurri Kurri) Pty Ltd v Scientific Committee (Established under s127 of the Threatened Species Conservation Act 1995) [2003] NSWCA 297 (17 October 2003) 9 at [7] per Spigelman CJ

requirements and tolerances to their environment, as well as the outcome of chance events. Based on Gleason's individualistic view, boundaries between communities need not be sharp but are representations of continuous turnover in species composition across landscapes and through time. Austin (1985) illustrated the distinction between the two models in the context of a simple environmental gradient (Fig. 1). While spatial scale may influence how overlapping species distributions are perceived, communities are not necessarily any more discrete at larger or smaller scales (see section on scale dependence below). The alternative models also have different implications for the status of dominant species, which always offer accurate representation of community occurrence under the Clementsian model, but not necessarily under the Gleasonian model (Fig. 1). Under the latter, communities can rarely be identified conclusively based on their dominant species alone.

The modern view of ecological communities is closer to the continuum model than the discrete one, although there is broad recognition that species are not distributed completely independently of one another (Austin & Smith 1989; Burrows 1990; Begon *et al.* 2006). Thus, a given location, by virtue of its physical characteristics, may be expected to support a reasonably predictable assemblage of species; but there is variability in the group of species actually present, vagueness in boundary location and overlapping membership of species between different assemblages (Begon *et al*. 2006). These patterns of variability are not well represented in a discrete system that demands recognition and ready identification of communities as distinct entities.

Despite its limitations, the discrete model nonetheless provides a powerful communication tool for describing and mapping communities, which so far has eluded approaches based on the continuum model (though see Ferrier *et al.* 2007; Schmidtlein *et al*. 2007). This partly explains why the discrete model still dominates much of the thinking that underlies environmental planning and assessment practice in Australia to the virtual exclusion of the continuum model. Another likely reason is that discrete boundaries are more broadly compatible with deterministic zoning approaches to land use planning, which is seen to be messy and uncertain when continuum concepts get in the way. However, a view of communities as largely discrete runs a significant risk of fuelling unrealistic expectations that any stand of vegetation can readily be assigned to the 'correct' pigeon hole, despite widespread recognition of continuum behaviour in both scientific and law literature (Preston & Adam 2004, Begon *et al*. 2006). When inevitably these expectations are not met, confidence in assemblage-based approaches and the planning system can be undermined. New planning and conservation approaches that draw from strengths of both discrete and continuum theories are more likely to achieve their goals and provide a more effective means of communicating complex issues to a broad audience in a sustainable way (Austin & Smith 1989; Burrows 1990; Moilanen *et al*. 2005; Ferrier *et al*. 2007).

Sources of uncertainty

Uncertainty is an inherent and pervasive characteristic of all knowledge. Regan *et al*. (2002) proposed a taxonomy that divided all forms of uncertainty into two groupings: epistemic and linguistic. Epistemic uncertainty encompasses imperfect knowledge about the state of a system – there is a fact of the matter, but it is unknown. The most obvious and measurable sources of epistemic uncertainty arise from extrapolations or interpolations, limitations on sample data and variability of the system over space and time. For example, our knowledge of the species composition of an assemblage is based on only a sample of all occurrences at all times and places, and is therefore subject to both measurement error and natural variation (Regan *et al*. 2002; Elith *et al.* 2003). Knowledge of species composition is also subject to systematic error because some of the component species are less detectable than others. The species composition of a community may appear to be more certain than it actually is unless its true variability is revealed by sampling at multiple times and places (Fig. 2). Subjective judgement comes into play with any interpretation of data and this form of uncertainty is especially influential when data are scarce (Elith *et al*. 2003). For example, judgement may be applied to a limited set of field observations to compile a list of species that characterise a community throughout its occurrence or to identify processes likely to threaten the persistence of a community under given conditions. Expert judgements of this sort are sometimes essential to the description, diagnosis and management of communities, although different experts may make different judgements and many exhibit serial over-confidence about their areas of expertise (Burgman 2005).

A less obvious source of epistemic uncertainty arises from limitations on knowledge about the structure and mechanics of the system itself - model uncertainty. This relates to the definition of a community, its underlying theory and how these concepts apply to particular cases. For example, the discrete and continuum models of communities may imply radically different interpretations as to whether particular species and locations fall within the circumscription of any given community (Fig. 1). Since the concepts underlying models are described in language, they may also be prey to linguistic uncertainty.

Linguistic uncertainty arises from limitations imposed by incomplete, imprecise or inaccurate language and concepts. It may interact with epistemic sources of uncertainty, particularly model uncertainty. An important source of linguistic uncertainty arises when categorical language is used to describe entities that exist along a continuum from one state to another, e.g. wet vs. dry, hot vs. cold (Regan *et al*. 2002). This inevitably creates intermediate cases, which cannot be assigned with certainty to one category or another (i.e. there is no fact of the matter). This source of uncertainty, termed 'vagueness' (Regan *et al*. 2002), is present in both discrete and continuum models of communities, though is more explicitly recognised in the latter (Elith *et al*. 2003). Both Regan *et al*. (2002) and Elith *et al*. (2003) point out that vagueness cannot be eliminated simply by adopting sharper and sharper boundary specifications. The adoption of arbitrary sharp thresholds to delimit communities from one another not only submerges the existence of a continuous reality, but sacrifices generality necessary for valid interpretation of the entities on the continuum. This has important implications for regulatory applications, explored further below.

Some other forms of linguistic uncertainty, such as context-dependence, underspecificity and ambiguity (where one term has more than one meaning), also involve trade-offs between more explicit knowledge and necessary generality about the properties of any given community, as well as wide application of community concepts and ideas (Regan *et al*. 2002). For example, when a community is described only as occurring in coastal Australia, there is no information about which parts of the coast its distribution includes or how far inland the 'coast' might extend. However, this level of underspecificity about its distribution might be appropriate if the community could occur within some coastal areas from which it has not yet been recorded. Some level of underspecificity allows necessary generality to accommodate unobserved cases. Note that there are also issues of vagueness and ambiguity surrounding the interpretation of 'coastal'.

Dealing with uncertainty in decision-making

The conservation of ecological communities requires a range of decisions about whether particular communities warrant protection (e.g. through listing), whether particular occurrences of a community warrant investment of management resources and if so, what actions might produce cost-effective outcomes, whether particular development applications should trigger regulatory processes, whether such developments should proceed and what conditions should be placed on them to minimise impacts. All of these decisions and their outcomes will be influenced by uncertainty, whether it is explicitly recognised or not. There are essentially two complementary ways of dealing with uncertainty in the interpretation of ecological communities. First, the magnitude of uncertainty may be reduced, for example, by measuring characteristics such as composition and distribution more precisely with better sampling methods and more sampling effort. Quantitative descriptive methods such as species fidelity measures (Bruelheide 2000, Tozer 2003) also help reduce uncertainties about community properties. This has benefits, irrespective of whether decision-making is deterministic or risk-based. However, many forms of uncertainty, such as model uncertainty and most linguistic uncertainty, are more difficult to quantify and reduce. Linguistic uncertainty could potentially be reduced by use of more precisely defined terms, but these rely on other terms, which in turn rely on others. As noted above, there also comes a point where terms become defined so tightly that they lose generality and fail to meet their original intent (Regan *et al*. 2002).

A second means of dealing with uncertainty is to explicitly incorporate what we know about it into the decision-making process. A wide range of risk-assessment methods and decision-theory tools have been developed for this purpose (Possingham *et al.* 2000; Ben Haim 2001; Burgman 2005; Moilanen *et al*. 2005). Some of these tools are complex and their uptake into conservation planning practice has been slow. However, the principles underlying them are simple: (i) questions are considered from a probabilistic perspective (quantifying uncertainty) rather than a deterministic one (ignoring uncertainty); and (ii) decisions aim to be robust by reducing the risk of 'bad' outcomes as defined by the planning objectives.

Rather than ask deterministically whether a protected community occurs on a proposed development site, in probabilistic language the initial question becomes, 'how likely is it that the protected community occurs on the development site?' The estimation of likelihood then requires relevant sources of uncertainty to be evaluated by reviewing the evidence of resemblance between all the properties (Table 1) of the observed community and those described in the listing statement or determination. Elith *et al*. (2003) suggest a variety of methods for reducing and quantifying each type of uncertainty. Uncertainty may be expressed as statistical probabilities if quantitative data are available. Otherwise, it will rely on expert elicitation and judgement to express a subjective probabilities or 'degrees of belief' about alternative states of a system or alternative values of a quantity within plausible bounds (Kyburg & Smokler 1964; Burgman 2005). If experts are able to estimate the most likely state of a system or best estimate of a quantity, they should also be able to provide information on the uncertainty associated with their advice (e.g. as degree of belief and/or plausible bounds). However, the latter is seldom sought or given, and hence limits the extent to which conservation decisions can accommodate uncertainty.

To illustrate the role of uncertainty in decision-making, consider a simple example in which a development is proposed on a parcel of land. After field inspection, an expert advises that there is a 40% chance that a threatened ecological community is present on the site and will be significantly affected if the proposed development goes ahead. Alternatively, the expert might simply have advised that the community is unlikely to be present (or not present) because it is more likely to be absent than present. By providing a quantitative estimate of uncertainty, however, the expert allows the decision maker to assess the risks of a bad planning outcome. For example, if development is approved, there is a 40% chance that an example of a threatened community will be destroyed; if approval is refused, there is a 60% chance of forgoing a development opportunity without any conservation benefit to the threatened community. The decision maker's attitude to uncertainty will determine how these risks are weighed up. Akcakaya *et al*. (2000) proposed that attitudes to uncertainty in conservation applications could be viewed as a continuum from risk-averse (i.e. precautionary) to risk-tolerant (i.e. evidentiary). Decision makers with highly precautionary attitudes to risk would be unwilling to grant development consent, even if there was a small likelihood that the protected community is present on site. Conversely, decision makers with highly evidentiary attitudes would be unwilling to refuse development consent unless there was almost no doubt about the presence of the community on site. This trade-off is analogous to the risks of making Type I and Type II errors in statistical inference (Taylor & Gerodette 1993). In law, the 'standard of evidence' expresses a similar concept (Simon & Mahon 1971).

An appropriate attitude to uncertainty should depend on the context of the decision, which may vary considerably between applications. When interpreting the results of experiments, for example, the frequentist statistical convention is to reject a null hypothesis (e.g. the threatened community is not present on site) only if there is less than 5% chance ($P<0.05$) that the result of an experiment could have been obtained by chance (Zar 1984). This very evidentiary attitude equates to a requirement for 95% certainty and has been heavily criticised on a number of grounds (e.g. Taylor & Gerodette 1993; Fidler *et al*. 2004). The convention in criminal law has a slightly less evidentiary attitude to uncertainty, whereby guilt must be proven 'beyond reasonable doubt'. Quantitative estimates of certainty that meet this requirement vary from 74-93%, depending on the crime, the people making the judgement and the proposed punishment under a guilty verdict (Simon & Mahon 1971; Abbott & Batt 1999). In civil law cases, however, questions must be settled on the basis of the 'balance of probabilities' or 'preponderance of evidence', implying a likelihood threshold of 50%[2]. For environmental decision-making, the precautionary principle has been widely adopted, at least in part because of the consequences of Type II errors (concluding that there is no effect when there actually is), especially

[2] Lord Denning in Miller v. Minister of Pensions (1947) 2 A11 ER 372.

when the statistical power of the investigation is low (Taylor & Gerodette 1993). The precautionary principle states that *where there are threats of serious or irreversible environmental damage, lack of full scientific certainty should not be used as a reason for postponing measures to prevent environmental degradation* (United Nations Environment Programme 1993). For Red Listing of threatened species, assessors adopt a ‘precautionary but realistic’ attitude to uncertainty (IUCN 2001), implying a certainty requirement somewhat less than 50% (*cf*. balance of evidence). If severe environmental consequences are involved, a more extreme precautionary attitude to uncertainty might be warranted, for example, triggering protective actions even when there is less than 10% risk of those consequences eventuating.

Figure 3 illustrates simple scenarios in which these different attitudes to uncertainty are applied to decision-making based on the risk of a bad planning outcome (e.g. the destruction of a threatened community at a development site). It shows that conservation decisions may sometimes depend on the decision-maker’s attitude to uncertainty. It is therefore important to be clear about the attitude to uncertainty adopted or the standard of evidence required and how it may be justified by the context of the decision. To a large extent, this will be guided by the consequences of getting it wrong and more precautionary attitudes to uncertainty will be warranted where important biodiversity assets could be lost.

Assessing the status of communities

Extinction of communities

Processes for assessing the conservation status of ecological communities share several common features with those for assessing the status of individual species. One global assessment protocol (Faber-Langendoen *et al*. 2007) uses identical criteria for assessing communities and species. The concept of extinction is common to assessing the status of both species and communities. Legislation protecting ecological communities in the Commonwealth of Australia (1999) and the State of NSW (1995) both refer to risk of extinction in defining different categories of threat, but neither offer a particularly workable definition of community extinction intended for direct application in assessments. Rodriguez *et al*. (2007) contrasted two extreme definitions of ecosystem extinction and recommended middle ground. The main difficulty is defining the endpoint where no examples of a community remain in existence. Assessments of species extinction are plagued by sampling questions about how much search effort is required before a diagnosis of extinction is justified (Keith & Burgman 2004; IUCN 2008). For communities, ‘extinction’ usually involves a transformation of community properties, which may be viewed as a replacement of the original community by one or more different communities. This creates a need to decide when an assemblage of species occupying an area no longer fits the description of the original community. Interpretation of community extinction therefore involves uncertainties involving categorisation of intermediate cases (vagueness), which cannot be resolved merely by specifying finer and finer arbitrary boundary conditions. Existing methods for dealing with uncertainty (Burgman 2005) could help to deal with these problems, but their uptake in routine assessments has been slow.

Risk criteria

Although defining the end point of extinction has intrinsic benefits, an explicit enunciation of the symptoms of extinction processes is likely to be more useful for assessing the conservation status of communities (Mace *et al.* 2008). Consequently, the most explicit protocols for assessing the status of communities rely on a number of indicators or symptoms of extinction risk. Nicholson *et al*. (in review) classified these into three groups of criteria: rate of decline in distribution; current size of distribution; and decline in ecological functions. The first two of these have analogues in Caughley's (1994) declining population and small population paradigms, respectively, and the third relates to higher-order community properties that involve multiple species and/or interactions between species and their environment.

While most current assessment protocols include representation of all three groups of criteria, there is some variation in how they are expressed. For example, some protocols assess reductions in distribution over a single time frame (e.g. Sattler & Williams 1999), some use two or more different time frames (e.g. Rodriguez *et al*. 2007), while others use flexible time frames scaled to the temporal dynamics of component species or the habitat (e.g. State of NSW 2005). Assessment protocols exhibit greatest variation in their treatment of criteria addressing ecological functions (Nicholson *et al*. in review). Some protocols exclude such factors from the assessment process (State of Queensland 1999), while others have several criteria that address decline in different ecological processes and functions (e.g. Commonwealth of Australia 2000). These criteria mostly have no analogues in species assessment criteria and remain an aspect of community assessment that continues to develop.

Assessment protocols

Most assessment protocols employ a decision-rule structure in which a community qualifies for a particular threat category if it meets any one of several conditions (Keith 1998). The conditions may specify either qualitative states (e.g. severe fragmentation) or thresholds in quantitative parameters (e.g. rates of decline or range size) or some combination of both. Faber-Langendoen *et al*. (2007) use an alternative approach in which assessments are based on a set of risk factors, which are combined in a complex algorithm based mainly on scores. Estimates of both qualitative and quantitative parameters are subject to a range of uncertainties. These uncertainties may be incorporated into the assessment process using a range of methods as discussed above (Burgman *et al*. 1999; Akcakaya *et al*. 2000; Regan *et al*. 2002). Again, key principles involve the estimation of uncertainty bounds and specifying an attitude to uncertainty or level of risk tolerance (Fig. 4). Robust assessments of risk therefore require information about both the best estimate of a parameter and the uncertainty around it in the form of bounds or a distribution (Fig. 4), rather than point estimates alone.

Nicholson *et al.* (in review) noted considerable variation among assessment protocols in thresholds for both rates of decline and range size that delineate different categories of threat. In general, thresholds adopted for decline and range size in communities were markedly more stringent than those for analogous parameters in accepted protocols for species assessment (IUCN 2001). Assessment methods adopted for listing of threatened species and communities under Australian legislation (Commonwealth of Australia 1999) provide an example. The indicative thresholds of decline for ecological communities are 95% for Critically Endangered (*cf*. 80% for species), 90% for Endangered (*cf*. 70% for species) and 70% for Vulnerable (*cf*. 30% for species) (Threatened Species Scientific Committee, undated a, b). Higher thresholds apply for species (90%, 70% and 50%, respectively, Threatened Species Scientific Committee, undated b) where the causes of decline are clearly reversible, understood and ceased (after IUCN 2001). Despite these differences, the categories of threat for assessing communities purport to delimit equivalent levels of extinction risk as those for species (at least 50% in the immediate future for Critically Endangered, at least 20% in the near future for Endangered, and at least 10% in the medium-term future for Vulnerable). Communities therefore require higher rates of decline and smaller ranges than species if they are to qualify for listing in equivalent categories of threat. Other methods (e.g. State of Queensland 1999; English and Blyth 1999; Benson 2006) exhibit a similar tendency, insofar as their thresholds are markedly more stringent than those in species assessment criteria adopted by IUCN (2001). Only NatureServe (Faber-Langendoen *et al*. 2007) and State of NSW (2005) employ identical thresholds for communities and species.

The protocols that employ more restrictive assessment thresholds for communities than those for species do not articulate a rationale for doing so. It is unclear why a community should decline further and be restricted to a smaller distribution than an individual species to be at similar risk of extinction. Arguably, the relationship should be reversed, or at least entail equivalent thresholds, if protection of ecological communities is to fulfil its role in conservation of poorly known or rare species, genetic variability and processes that operate above the species level. As Nicholson *et al.* (in review) point out, the use of more restrictive risk assessment practices for communities than species seems inconsistent with the fundamental roles of ecological communities in biodiversity conservation (Austin & Margules 1985; Noss 1987, 1996; Rodriguez *et al*. 2007).

Scale-dependence of ecological communities

Ecological communities are scale-dependent representations of biodiversity – they can be defined at any level of resolution (Begon *et al*. 2006). Scale comprises three components: thematic, spatial and temporal. Thematic scale (or classification scale) describes the level of resemblance at which a given community is distinguished from others. Classification of communities may thus be viewed hierarchically, whereby broadly defined assemblages may comprise a number of related and more narrowly defined communities (Fig. 5). To illustrate this property, Preston & Adam (2004) drew an analogy between communities and Russian Dolls, whereby each doll contains a set of smaller dolls. Thematic scale may be quantified by a range of numerical resemblance metrics (Faith *et al*. 1987) or less explicitly, by the number of communities defined within a given area. Such measures are context-dependent; they have meaning relative to the samples within a given data set, but no absolute meaning that might support universal statements about how broadly or narrowly particular communities may be defined.

Spatial scale describes the resolution at which the distribution of a community is mapped, and can be quantified using a range of spatial parameters including scale ratios, bar scales, pixel sizes or grain, minimum polygon sizes, etc. Unlike measures of thematic scale, these are absolute and permit comparison across different communities, irrespective of whether they were defined using the same or different data sets. Temporal scale concerns the time intervals that distinguish variation within and between communities (e.g. Fig. 2). These are rarely specified, although most conservation applications implicitly assume a contemporary interval encompassing an unspecified length of time into the recent past and near future. More importantly, temporal scale influences the degree to which the 'successional' stages or states of species assemblages are delineated as separate communities or lumped together.

The three components of scale are often correlated. Broadly defined assemblages are typically mapped at broad spatial scales and may encompass temporal variation over long time scales, while narrowly defined assemblages are typically mapped at fine spatial scales and some of these may be quite transient expressions in a landscape. However there are many exceptions to this generalisation and, conceptually, the three components of scale are independent.

The scale at which communities are defined influences their performance in conservation planning applications. Pressey & Bedward (1991), for example, found that finer scale map units represented the distributions of individual species more effectively than coarse-scale units. However, trade-offs arise because data are rarely adequate to support fine-scale classifications with a reasonable level of accuracy across large regions. Fine-scale classifications may also sacrifice some parsimony if they involve large numbers of communities.

Spatial scale also has a well-known effect on estimates of the area occupied by species and communities (IUCN 2001, 2008; Hartley & Kunin 2003; Nicholson *et al*. in review). When mapped individually, the area occupied by a community (or species) is larger when estimated at coarser spatial scales than at finer scales (Fig. 6). Thematic scale is likely to have a similar influence. Consequently, variation in scale may have important implications for assessing the status of communities, whenever this involves estimates of distribution size (Nicholson *et al*. in review). Put simply, the coarser the scale at which distributions of individual communities are estimated, the more likely they will exceed area thresholds that define levels of threat, and *vice versa* (Fig. 7). To limit inconsistencies in such assessments, IUCN (2008) recommend a standard scale for assessing species' distributions that is appropriate to the fixed thresholds defining different categories of threat. None of the existing protocols for assessing communities deal with scale in a similarly explicit manner, except those that follow the IUCN species assessment guidelines (Faber-Langendoen *et al*. 2007; State of NSW 2005).

Maps of ecological communities

Maps showing the distribution of ecological communities are often seen as an effective means of clarifying uncertainties in regulatory applications designed to protect them[3]. Some regulatory applications (e.g. State of NSW 1985; State of Queensland 1999) take a prescriptive view, whereby maps with 'sharp' boundaries delineating protected communities are part of a regulatory instrument and may only be updated subject to a Ministerial approval process. On the face of it, such an approach provides great certainty to regulators and developers, and yet it fails to recognise that maps and their boundaries also have intrinsic uncertainties (Elith *et al*.

[3] Hornsby Shire Council v Vitone Developments Pty Ltd [2003] NSWLEC 272 at 114

2003). It also fails to recognise that distributions of communities may change through time (natural variability) and that knowledge about their distributions may also change. Consequently, a prescriptive approach carries some risk that incorrectly mapped areas may be unduly protected or conversely, some unmapped occurrences of a listed community may escape regulatory protection. The process for remedy of such anomalies may be cumbersome, tardy and, at worst, open to abuse.

Courts have determined that maps are not required to support legal protection of ecological communities in NSW (Preston & Adam 2004). In relation to the NSW Threatened Species Conservation Act 1995, Chief Justice Spigelman held that

> *To satisfy the requirement of certainty to an appropriate standard, the terms of the Scientific Committee's final determination must enable a citizen to decide whether a specific location falls within it. This does not necessitate the enumeration of the minimum number of species that must be found together to constitute the community, nor the provision of maps indicating where the community may be found.* [4]

Determinations of threatened communities in NSW identify bioregions in which the community occurs and cite relevant studies (often with maps) that help describe the distribution of the community in more detail. This approach allows use of maps to inform interpretation but does not fix them into the regulatory instrument. This additional flexibility readily accommodates corrections and updates and, where published scientific literature is used as reference material, also benefits from improved certainty associated with the peer review process. Legal tests of the existence of a community at particular locations are likely to apply, irrespective of whether maps are used in flexible or prescriptive ways.

Despite their limitations, maps can make useful contributions to conservation decisions about communities. Uncertainty in maps can be incorporated into the decision-making process in a variety of ways, some of which require quantification (Elith *et al*. 2003), while others require only rudimentary awareness of relevant issues. Very few existing maps include information on accuracy or uncertainty (Keith & Simpson 2008). Even where this information is lacking, a general understanding of uncertainties in mapped community boundaries can inform decisions in ways that reduce risks of bad outcomes. For example, planning boundaries can incorporate buffers that accommodate uncertainty in the locations of mapped boundaries of protected communities and simultaneously address local land management issues such as urban runoff and catchment integrity (Fig. 8). Both of these considerations support the location of a planning boundary some distance outside the currently mapped boundary of the community, rather than aligning the planning boundary with the community boundary (Fig. 8).

Conclusion

Ecological communities are important conceptual tools for conservation of biodiversity above the species level. The concept has a long history in ecology and a well-established role in conservation planning; but its application in regulatory systems and risk assessment protocols is relatively recent. Conceptual complexities

[4] VAW (Kurri Kurri) Pty Ltd v Scientific Committee (Established under s127 of the Threatened Species Conservation Act 1995) [2003] NSWCA 297 (17 October 2003) 9 at [9] per Spigelman CJ

pose significant challenges for the valid and effective application of communities to conservation problems. Misapplications of the concept, for example, through failure to identify relevant community properties, failure to minimise and accommodate intrinsic uncertainties in decisions and failure to address communities at scales appropriate to a given problem, may result in substantial risks of poor conservation or planning outcomes. Satisfactory resolution to these challenges will usually require specialist advice, analogous to that provided in medical, engineering and accounting professions. However, the challenges are worth persevering with. When applied effectively, the concept of ecological communities can be a powerful tool for delivering cost-effective outcomes for land use planning and biodiversity conservation.

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Table 1. Six key properties of ecological communities. Defining properties are underlined.

Property	Description
<u>Composition</u>	Species present in the assemblage and their relative abundances (including dominance)
Structure	Vertical & horizontal arrangement of component organisms in the assemblage
Habitat	Environmental conditions (niche) in which the assemblage occurs
<u>Distribution</u>	Geographic area and time period within which the assemblage occurs
Biological interactions	Processes by which component organisms affect others within the community, including competition, predation, pathogenicity, mutualisms (e.g. pollination, herbivory)
Other ecological processes and functions	Processes, in addition to biological interactions, by which component organisms may affect their environment and vice versa including the cycling of water, nutrients & carbon, dispersal & recruitment of organisms, ‘successional’ change in relation to disturbance regimes (fires, floods, storms), etc.

Table 2. Definitions of ecological communities and ecosystems applied in regulatory applications within various jurisdictions, compared to an ecological text.

Term	Source	Definition	Notes
Community	Begon *et al*. (2006)	An assemblage of species populations that occur together in space and time	Scientific definition
Ecological community	Commonwealth of Australia (1999)	The extent in nature in the Australian jurisdiction of an assemblage of native species that: (a) inhabits a particular area in nature; and (b) meets the additional criteria specified in the regulations (if any) made for the purposes of this definition	‘Nature' and 'indigenous' are undefined. 'Native species' are defined as indigenous to Australian territory. No 'additional criteria' (b) are currently specified in Regulations.
Ecological community	Republic of South Africa (2004)	An integrated group of species inhabiting a given area	‘Integrated' is undefined.
Ecological community	State of New South Wales (1995)	An assemblage of species occurring in a particular area	
Community	State of Victoria (1988)	A type of assemblage which is or which is wholly or substantially made up of taxa of flora or fauna existing together in the wild	‘Substantially made up of' and 'the wild' are undefined.
Ecosystem	Begon *et al*. (2006)	The biological community together with the abiotic environment in which it is set	Scientific definition
Ecosystem	Commonwealth of Australia (1999)	A dynamic complex of plant, animal and micro-organism communities and their non-living environment interacting as a functional unit	‘Dynamic complex' and 'functional unit' are undefined.
Ecosystem	Republic of South Africa (2004)	A dynamic complex of animal, plant and micro-organism communities and their non-living environment interacting as a functional unit	‘Dynamic complex' and 'functional unit' are undefined.
Regional ecosystem	State of Queensland (1999)	A vegetation community in a bioregion that is consistently associated with a particular combination of geology, soil and landform	‘Vegetation community' and 'consistently associated with' are undefined.

Term	Source	Definition	Notes
Natural habitat [type]	European Union (1992)	Terrestrial or aquatic areas distinguished by geographic, abiotic and biotic features, whether entirely natural or semi-natural	‘Features', 'natural' and ‘semi-natural' are undefined.
Critical habitat	United States of America (1973)	Defined only with reference to listed species	Legislation confers only species-level protection
Critical habitat	Canada (2003)	Defined only with reference to listed species	Legislation confers only species-level protection

Figure 1. Alternative models of ecological communities showing variation in abundance of component species along a simple environmental gradient (after Austin 1985). (a) Organismal discrete model (Clements 1916). (b) Individualistic continuum model (Gleason 1936).

Figure 2. Multi-dimensional scaling ordination showing variation in species composition of four ecological communities (A-D) in the Nowra area, south-eastern Australia (data from Simpson *et al*. 2008). Lines show trajectories joining a temporal sequence of samples from the same quadrats that were surveyed at intervals over a five-year period after a fire. The temporal variation along each trajectory and spatial variation between sites within communities encompass sampling error, natural variation and elements of systematic bias related to variation in detectability between species. Despite the uncertainty attributable to these sources, the non-overlapping ellipses suggest that the four communities exhibit distinctive species composition. Increased sampling in space and time may reveal a blurring of these distinctions, however, consistent with the continuum model.

Figure 3. A risk assessment framework incorporating uncertainty showing the effect of varied 'attitudes to risk' or 'standards of proof' on decisions. A-D represent four potential sites for development. Bars represent the distribution of uncertainty in the risk of environmental harm if development proceeds. In this example, uncertainty for all sites is of equal magnitude (bar length) and follows a uniform distribution (each level of risk between the upper and low bounds has an equal chance of being the 'true' risk of environmental harm). Note that uncertainty could be represented by any fuzzy number or statistical distribution (see Fig. 4). Broken vertical lines (I-III) represent different decision thresholds: development is approved if risk of harm falls to the left of the line and consent is refused if risk of harm falls to the right of the line.

The three decision thresholds imply different levels of tolerance to risk of harm ranging from I (risk-averse) through III (risk-tolerant). Numbers on each bar represent different attitudes to uncertainty or standards of proof that could be adopted in making a decision about development consent on each site (see text): 1- null hypothesis testing, 2- beyond reasonable doubt, 3- balance of evidence, 4- precautionary but realistic, 5- extreme precautionary. If decisions were made using a relatively risk-averse threshold (I), development consent would be refused at sites A, B and C, irrespective of the attitude to risk because the bounds of uncertainty are to the right of threshold I. Development could only be approved at site D, and then only under the most evidentiary attitude to risk or highest standard of proof (1- null hypothesis testing). If decisions were made using a risk-tolerant threshold (III), development would always be approved at sites C and D, and would only be refused at site B under an extreme precautionary attitude to risk (5) and only refused at site A under a precautionary but realistic attitude (4 or 5). If decisions were made using an intermediate level of risk tolerance (threshold II), then the planning outcome for all sites depends on the decision-makers' attitude to uncertainty. For example, under a 'beyond reasonable doubt' attitude, development would be approved at all sites except A. Under a 'balance of evidence' attitude, development consent would be refused for sites A and B, and approved for sites C and D.

Figure 4. Examples of different patterns of uncertainty in range-size estimates for three communities illustrated by contrasting distributions: A- normal; B- uniform; C-

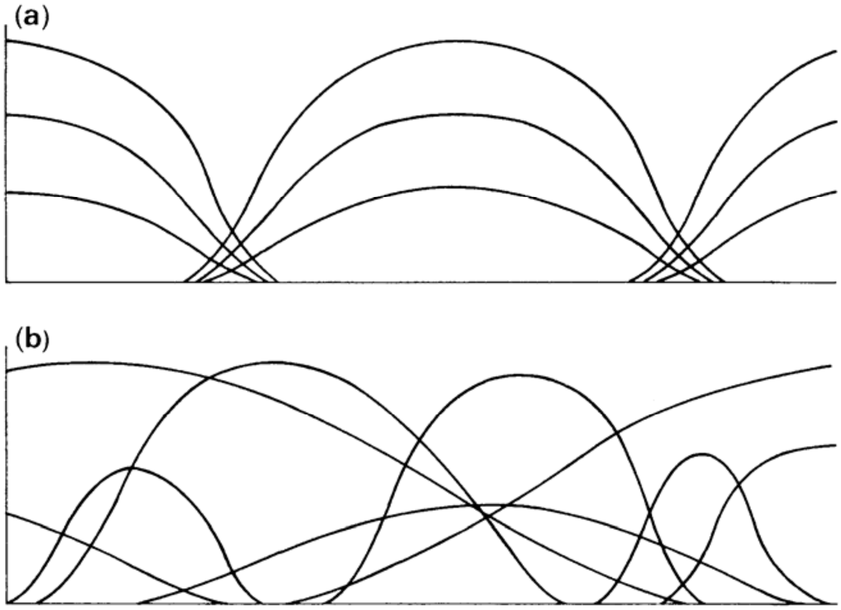
lognormal. The vertical line represents an assessment threshold that defines a category of threat. There is a 50% chance that communities A and B fall below the range-size threshold, compared to 65% for community C, which is therefore more likely to be eligible for listing as threatened.

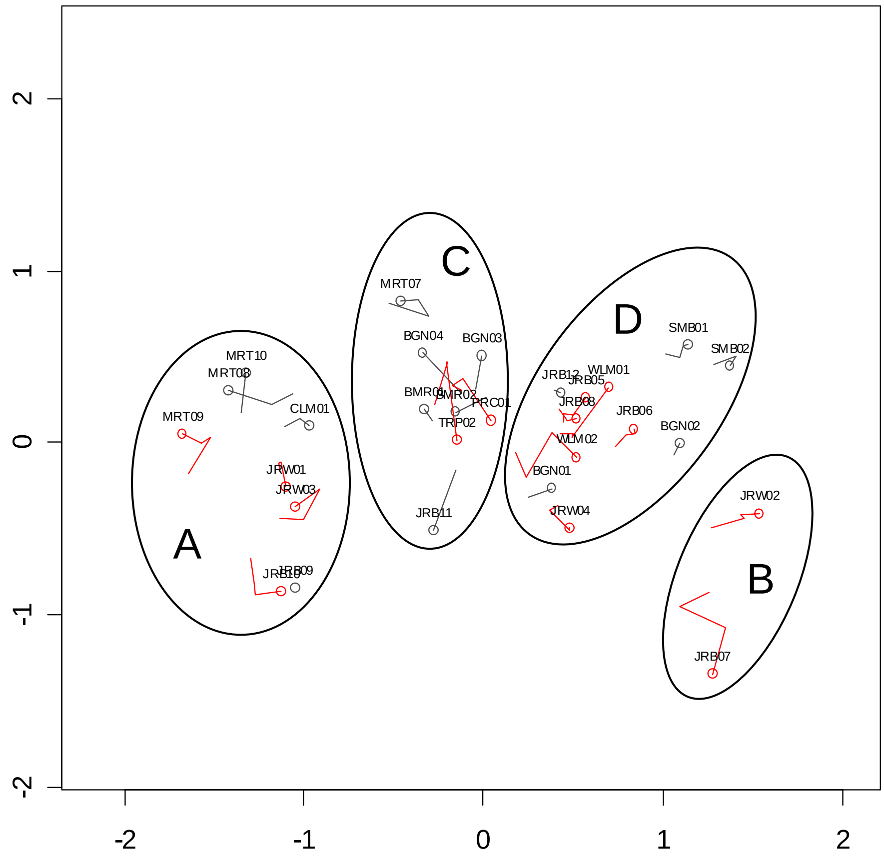
Figure 5. Two different representations of thematic scale in hierarchical classifications of ecological communities. (a) A Venn diagram showing 14 fine-scale communities nested within each of seven meso-scale communities nested within each of three coarse-scale communities. (b) A dendrogram showing communities 13 recognised at fine scale (low dissimilarity or high level of resemblance), 10 communities at meso-scale (intermediate dissimilarity) and five communities recognised at coarse scale (high dissimilarity).

Figure 6. Areas occupied by ecological communities (and species) may be estimated at a variety of scales, illustrated most simply by grid cells of varying size (after Gaston 1995). Coarser scales of measurement (a) yield larger estimates of area occupied than finer scales (b).

Figure 7. Area of occupancy estimated for four different ecological communities in south-east NSW using a geometric progression of grid sizes to represent varying scales of measurement (data from Tozer *et al*. 2005). The different communities have different scale-area relationships due to their different patterns of dispersion in the landscape, though all have positive slopes (i.e. estimates increase with increasing coarseness of scale). Whether the areas occupied by the communities are smaller than thresholds that define categories of threat (thin horizontal lines) depends on the scale at which their areas are measured. Using 2 × 2 km grids (vertical broken line), the scale recommended for assessment by IUCN (2006), three of the communities are eligible for Endangered status and one is outside the threshold for Vulnerable.

Figure 8. Vegetation map boundaries (thin white lines) delineating a stand of Blue Mountains Swamp, a Vulnerable Ecological Community in NSW, and a hypothetical planning boundary designating an environmental protection zone located with regard to uncertainty in the swamp boundary and protection of its catchment.


(a)
(b)


A
B
C
D
MRT07
BGN04
BGN03
MRT10
MRT08
MRT09
CLM01
BMR01
BMR02
PRC01
TRP02
JRB12
WLM01
JRB05
JRB08
JRB06
BGN02
SMB01
SMB02
WLM02
BGN01
JRW01
JRW03
JRW04
JRW02
JRB11
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-1
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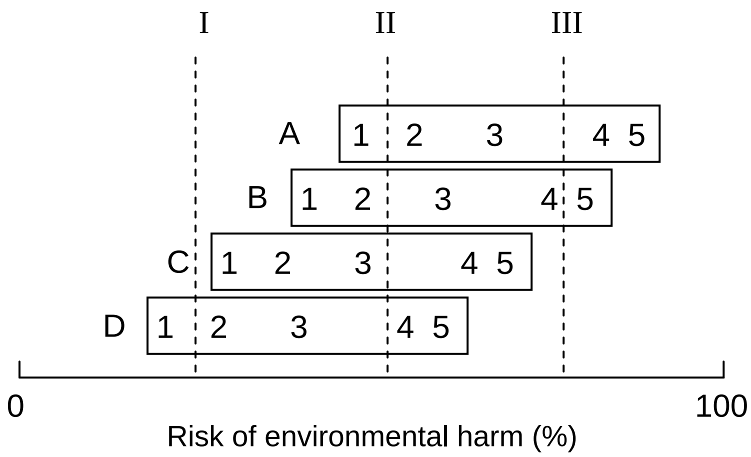
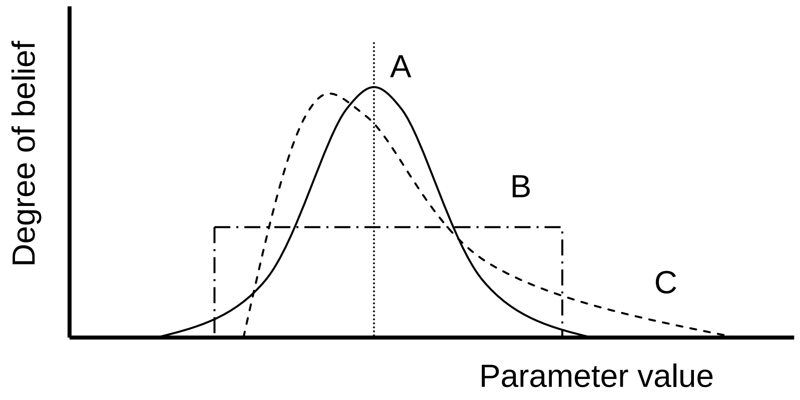
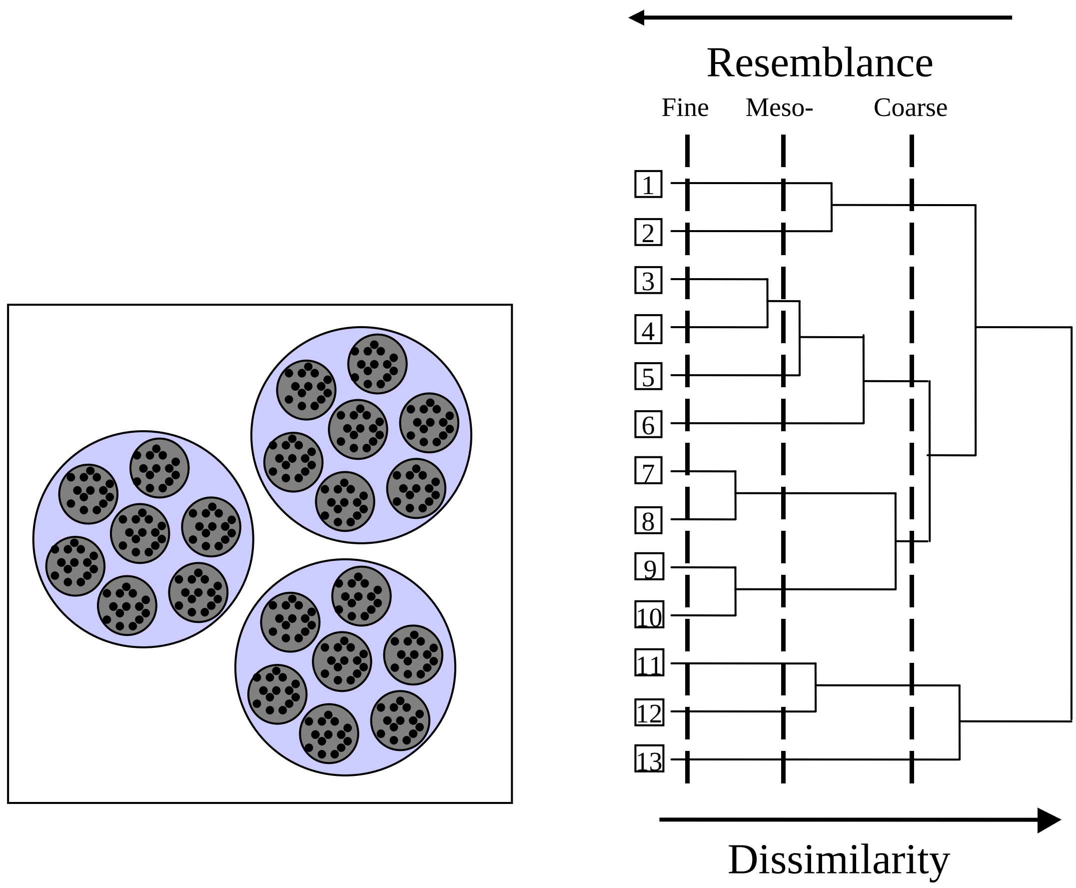
1
2


I
II
III
A
1 2 3 4 5
B
1 2 3 4 5
C
1 2 3 4 5
D
1 2 3 4 5
0
100
Risk of environmental harm (%)


Degree of belief
A
B
C
Parameter value


Resemblance
Fine
Meso-
Coarse
1
2
3
4
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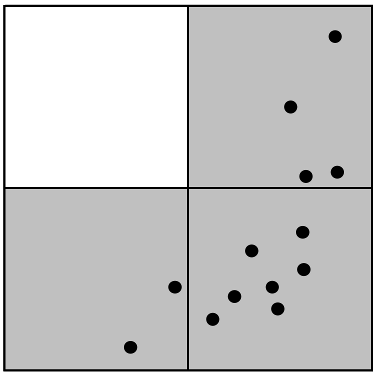
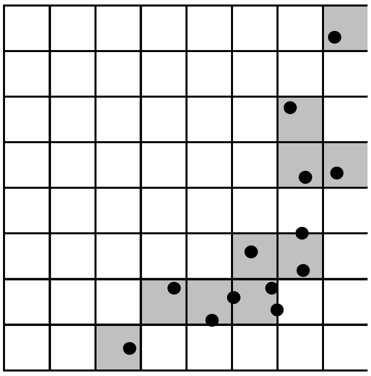
6
7
8
9
10
11
12
13
Dissimilarity

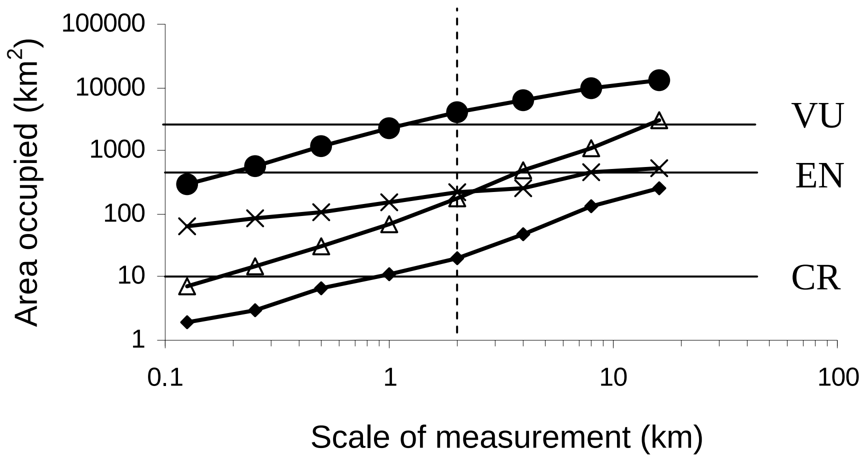
(a) (b)



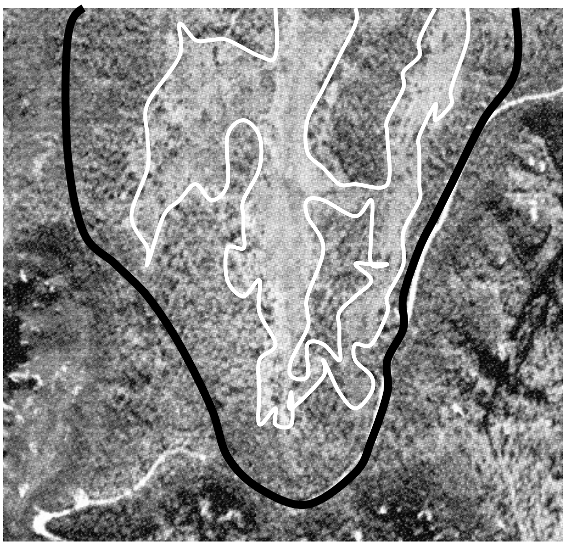
(a)



(b)



100000
10000
1000
100
10
1
Area occupied (km2)
VU
EN
CR
0.1
1
10
100
Scale of measurement (km)



Dealing with threats: integrating science and management

By Tony D. Auld and David A. Keith

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Summary Threats to biodiversity are pervasive and diverse. The management of these threats is the major focus of conservation biology. We briefly reviewed the taxonomy of direct threats to biodiversity as a means of examining legislated planning actions and strategies for managing threatening processes in eastern Australia and propose a simple classification of five general types of threat. These include the major threats of destruction and fragmentation of habitat and global climate change; threats that relate directly to changes in disturbance regimes (e.g. fire and water); threats related to reduced functionality of biological interactions or life cycle processes (e.g. due to invasions of exotic species); and, over-exploitation affecting specific groups of plant and animal species. We applied this 5-level classification to the listings of Key Threatening Processes under legislation in three Australian jurisdictions (Commonwealth, New South Wales, Victoria). A small core group of threats are currently common to all three jurisdictions, including clearing of native vegetation, climate change and particular alien predators and diseases. The bulk of listings reflect differences in emphasis between jurisdictions, with the Commonwealth having not listed threats related to weeds or disturbance regimes.

To manage all types of threats we need a clear understanding of cause and effect combined with adaptive management strategies for amelioration. We use selected case studies to show how a process-based understanding of threats can be incorporated into conservation management. In fire-prone ecosystems in NSW, for example, threats posed by increased fire frequency have been addressed in fire planning by prescribing management thresholds, derived from knowledge of the limits set by seed bank dynamics in relation to species persistence. Monitoring of species after fire allows re-evaluation of recommended intervals. In arid and semi-arid systems, grazing by introduced herbivores has disrupted the recruitment of new individuals into plant populations, resulting in long-term decline of a number of perennial plant species across extensive proportions of their ranges. In conjunction with long-term monitoring of recruitment patterns at selected locations, control of introduced herbivores is being used to address this threat. Finally, strategies for dealing with threats of a changing climate (including managing the capacity of species to disperse across the landscape) must include strategies for minimising habitat loss and on-site management of existing threats.

Key words clearing, climate change, disturbance regimes, exotic species, fire, life history processes, key threatening processes

Introduction

Biodiversity conservation is a key objective for many land management agencies. Effective conservation measures that promote the persistence of species, communities and ecosystems must deal with those factors (threats) that directly or indirectly lead to an increased risk of decline in species, populations or ecosystem functions or to unwanted changes in species assemblages. The diagnosis and management of the causes of these declines is a central concern of biodiversity conservation (Caughley 1994).

An important property of threatening processes is that they operate in ways that affect multiple species and ecosystem processes more or less simultaneously. Conservation actions that mitigate a particular threat are therefore likely to produce broad benefits in the form of reduced risks to multiple species. Threatening processes are therefore an important conceptual tool for conservation above the species level, including ecological communities and functional classifications of species (Keith this issue).

Dealing effectively with threats requires a taxonomy that enables us to order ideas and information into a framework useful to guide policy and management actions. This applies to direct threats relevant to the local or regional context of biodiversity conservation (Salafsky *et al.* 2002, 2008; IUCN 2006); while, at the same time, there will be a number of indirect threats and opportunities (Salafsky *et al.* 2002, 2008) that will impact on the ability of land managers to deal with the direct threats.

In Australia, threatened species legislation provides a framework for identifying and co-ordinating management of threats to biodiversity at a broad scale. The legislated listings of Key Threatening Processes are not necessarily complete, as many potential threats are yet to be assessed, but as this legislation has been in operation for over a decade the listings are representative of the major direct threats. The co-ordination of management actions occurs through a number of mechanisms: formal Threat Abatement Plans (e.g. NPWS 2001; DEC 2006); Statements of Intent (DECC 2008); Action Statements (e.g. DSE 2003); or, other conservation initiatives addressing the direct threat or the influence of indirect threats and management opportunities.

In this paper, we briefly review the taxonomy of threatening processes and propose a classification of five general types of threat. We review the current listings of Key Threatening Processes in three Australian jurisdictions to explore differences and commonalities in approach and emphasis. We then examine several examples to assess the success of a range of conservation actions at specific locations that attempt to minimise the impacts of direct threats to biodiversity from local to landscape scales.

Classification of Threatening Processes

One of the earliest taxonomies of threat was proposed by Diamond (1989), who coined the term ‘evil quartet’ to identify habitat destruction and degradation, over-exploitation, species introductions, and extinction cascades as the major causes of extinction in the modern era. Salafsky *et al.* (2002) proposed an expanded hierarchical classification based on three broad classes (ecosystem elimination, ecosystem degradation, and species decline and elimination) and dealing with specific issues such as pollution and transport infrastructure. More recently, Salafsky *et al.* (2008) suggested a hierarchical classification of threats (under 11 themes), conservation actions and stresses (attributes of the ecology of species and communities that are affected by threats). We propose a simple classification of five major groupings based on cause and effect (Table 1). Inevitably, interactions and synergies between different threatening processes blur the distinctions between these broad classes, but they provide a useful framework for assessing policy and management approaches. We applied this classification to the listings of Key Threatening Processes under legislation in three Australian jurisdictions: the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act); the NSW *Threatened Species Conservation Act 1995* (TSC Act) and NSW *Fisheries Management Act 1994* (Fisheries Act) and the Victorian *Flora and Fauna Guarantee Act 1988* (FFG Act).

Overall, some 40-50 Key or Potentially Threatening Processes are currently listed in each of NSW and Victoria, whereas 17 are listed under the EPBC Act. In each case, more than half of the listings relate to terrestrial habitats. Threats to freshwater habitats appear under-represented at the Commonwealth level (one of 17 listings), compared to NSW and Victoria (nine listings each), while threats affecting marine habitats account for 3-6 listings in each jurisdiction. Surprisingly few threatening processes are common to all three jurisdictions (Fig. 1), including anthropogenic climate change, three feral animals (foxes, rabbits, cats) and two diseases (*Phytophthora cinnamomi* and Chytrid fungus). Both NSW and Victoria have a number of unique listings (Fig. 1). NSW shares more than half of its other listed threats with the Commonwealth or Victoria, while these latter jurisdictions share only one threat that is not listed in NSW (Fig. 1).

Across all jurisdictions, listing of Key or Potentially Threatening Processes does not simply reflect those factors affecting currently listed threatened species, but also identifies broadscale threats that affect widespread or common taxa. Both the *EPBC* and *TSC* Acts state that a Key Threatening Process ‘could cause species, populations or ecological communities that are not threatened to become threatened’. The *FFG* Act states a similar but more general principal. This recognises the need to control threats across the landscape to prevent further species declines. Some of the listings in each jurisdiction are proactive, seeking to identify future threats before their impacts cause significant losses of biodiversity. Examples include anthropogenic climate change and a number of exotic species that are yet to establish, but have been shown to be problematic elsewhere (e.g. for NSW the *TSC* Act lists Red Imported Fire Ant, and Large Earth Bumblebee). Such listings promote preventative actions to reduce the risk of realisation of a threat in an area, rather than simply managing control after establishment.

Habitat destruction and degradation

In Australia, the impacts of habitat loss operate at various spatial scales and are skewed towards landscapes with flat terrain in humid temperate and subtropical climates, the most suitable environments for agricultural production and urban development (e.g. Pressey *et al*. 2000). At broad scales, habitat loss refers to clearing of native vegetation, but finer scale examples include the loss of habitat elements such as hollow-bearing trees, coarse woody debris or bushrock. Habitat loss increases extinction risks by reducing the area of habitat available for plant and animal populations. Less obviously, it may also reduce the suitability and connectivity of remaining habitat patches. Habitat degradation (caused for example by erosion, sedimentation, salinization, eutrophication, pollution, overgrazing, weed invasion etc.) may reduce the number of individual plants and animals that a given area is able to support (Maclean *et al*. 2003) and may also reduce rates of survival and reproduction. Habitat fragmentation limits the scope of interactions between and within species and reduces species movements across the landscape, interrupting gene flow, recolonization and rescue effects (Hobbs & Yates 2003; Young *et al*. 1996; Young & Clarke 2000). The scale and severity of habitat loss and degradation is widely documented (e.g. Cogger *et al*. 2003). It is foremost among Diamond's (1989) 'evil quartet' and is listed as Key Threatening Processes in all three Australian jurisdictions that we examined (Table 1).

Global climate change

Anthropogenic climate change is now 'virtually certain' (IPCC 2007) and will impose major impacts on species and ecosystems (Hughes 2003; Thomas *et al.* 2004; Dunlop & Brown 2008), although many of the potential impacts on ecological processes remain poorly understood. The mechanisms by which climate change could affect biodiversity include: (i) rapid shifts in suitable bioclimates relative to dispersal rates; (ii) changed growth rates and competitive relationships resulting from changes in atmospheric and oceanic CO_2 concentrations (in combination with changed temperature and moisture regimes); (iii) loss or change of habitats resulting from ice-melt and sea level rise; (iv) changes in life-history phenologies leading to desynchronization of dispersal events, decoupling of mutualisms and other interactions; (v) exposure to new competitors, predators and pathogens able to expand their distributions under new climatic regimes; and (vi) changes to disturbance regimes and frequency of extreme weather events with consequent disruptions to life cycles of plants and animals. The interactions and synergies between climate change and other threatening processes are likely to be stronger and less predictable than those between other groups of threats. Unlike Diamond's (1989) historically focussed 'evil quartet', climate change is an emerging threat, whose outcomes are only just beginning to be realised (Hughes 2000). As well, climate change tends to be overlooked as a tangible threat to currently listed threatened species (Westoby & Burgman 2006; Burgman *et al*. 2007). Nevertheless, all three Australian jurisdictions list anthropogenic climate change as a Key Threatening Process (Table 1).

Dysfunction of biological interactions

Processes that change interactions between species can threaten their persistence in two ways. First, the decline or loss of a key partner in multi-species relationships may lead to declines in dependent species. Loss of key predators (Berger *et al*. 2001; Johnson *et al*. 2007), hosts (Koh *et al*. 2004), prey (Ferrer & Negro 2004; Wiedenmann *et al*. 2008) or mutualisms such as mycorrhizae, pollinators and dispersal agents (Nunez-Iturri *et al*. 2008), are all examples that may exacerbate extinction cascades (Diamond 1989). These effects are likely to be most evident where organisms have highly specific dependencies on one another (Koh *et al*. 2004) or in systems governed by top-down regulation (Johnson *et al*. 2007). Second, invasions of alien species (weeds, pests and pathogens) may generate new interactions that threaten indigenous species and ecosystems. There are many examples where introduced competitors (Gentle & Duggan 1998), herbivores (Auld 1995a, b), predators (Dickman 1996) and diseases (Cahill *et al*. 2008) have lead to significant declines in native biodiversity, and they are also identified in Diamond's (1989) 'evil quartet'. A notable characteristic of Australian listings is that they currently overlook the first group of interactions; no example of disruption to interactions among native species is listed as a Key Threatening Process (Table 1). In contrast, all three jurisdictions list a broad range and large number of specific interactions involving alien species, although alien pollinators and plant competitors are both conspicuously absent from the *EPBC* listings (Table 1), perhaps reflecting a biased emphasis on vertebrate pests and diseases.

Changes to disturbance regimes

Disturbance regimes play an important role in the life histories of many species. Changes to these disturbance regimes (the frequency, intensity, seasonality and type of events) and their spatial patterns may interrupt life cycles or reduce the availability of food, shelter and breeding sites, resulting in species declines and extinctions (Whelan *et al*. 2002; Nebel *et al*. 2008). Alterations to components of both fire and water regimes have been listed as key threats in both NSW and Victoria, but are absent from the Commonwealth listings (Table 1). While not explicitly identified in Diamond's (1989) 'evil quartet', Salafsky *et al*. (2002) identify changes to disturbance regimes as a direct threat to biodiversity, and Salafsky *et al.* (2008) use natural systems modifications to encompass changes to disturbance regimes.

Over-exploitation of native species

Declines and extinctions of over-exploited species are a well-documented element of the 'evil quartet' (Bucher 1992; Bowen-Jones & Pendry 1999; Clapham *et al*. 1999). The impacts of over-exploitation may directly reduce survival or fecundity of the target species (e.g. Lamont *et al*. 2001) or indirectly affect other species, for example through bycatch of fish or shark-netting (Krogh & Reid 1996) or logging impacts on non-harvested taxa (Ough & Murphy 2004). In Australia, the most conspicuous examples of terrestrial over-exploitation are historical (King 1999; Hrdina & Gordon 2004). The small number of current listings in each jurisdiction are relatively species-specific and recognise threats to marine species and terrestrial orchids (Table 1).

Case studies

Management of direct threats for biodiversity conservation requires a clear understanding of how and where each threat operates, the nature of any interactions between different threats, and how each threat may be changing. Diagnosis of cause and effect is crucial to the effective management of threatening processes (Caughley 1994). Once this is done, adaptive management strategies can be developed to ameliorate the relevant threats. Such strategies may range from specific on-ground control actions (e.g. in the case of exotic species) to legislative planning and policy tools (e.g. in water management). In some cases, the ameliorative action will need to be experimental as knowledge of the likely response and sensitivity of the biota to the control treatment may be limited (e.g. herbicide spraying of weeds). Adaptive strategies promote such experimentation to spread and reduce risk of failure across alternative actions (Lindenmeyer & Burgman 2005). Finally, responses to any management actions must be monitored to inform adjustment and development of threat amelioration measures accordingly over time. All this must be done in the context of limited resources, necessitating a prioritisation of actions or target species, while minimising the impact of threats across local and landscape scales. For any one type of threat, there may be several different but complementary strategies applied at a range of scales. Below, we examine examples of threat impacts resulting from changes to disturbance regimes and dysfunction of biological interactions, while also suggesting key issues that need to be considered for addressing the impacts of climate change on biodiversity.

Changes to disturbance regimes

Disturbances, such as fires, floods, storms, landslips and tree falls, have key roles in the survival, recruitment and dispersal of many plants and animals. The timing, magnitude and frequency of these disturbance events may be altered by humans, with consequences for the persistence of species with disturbance-dependent life history processes.

Fire regimes have crucial roles in the life cycles of a large portion of the Australian flora (Whelan *et al*. 2002). Fires govern the survival of standing plants and may stimulate the release or germination of seeds and facilitate establishment of seedlings by liberating resources and reducing competition from standing vegetation. Diagnosis and management of fire as a threat to biodiversity requires an understanding of how variation in the frequency, intensity and seasonality of fires, as well as their size and patchiness, interacts with the life cycles of individual species. Some fire regimes may promote the persistence of a species by providing the essential stimuli for population turnover, while other fire regimes may be detrimental to persistence of the same species by interrupting its key life cycle processes. One well documented example is that of the endangered plant *Grevillea caleyi* (Proteaceae) from the Sydney Basin of south-eastern Australia. In the Sydney Basin, fires are widespread across much of the landscape. The region has high levels of species richness, including many endemic species and many species with recruitment cued to fire (Keith 1996; Auld & Ooi 2008). The distribution of *G. caleyi* is restricted to a few hectares in the urban/bushland interface of the northern suburbs of Sydney. Fire plays a key role in the life history of *G. caleyi* (Fig. 2) and its habitat (the endangered Duffys Forest ecological community). *Grevillea caleyi* is killed by fire and relies on germination from a soil seed bank for post-fire persistence. The abundance of post-fire seedlings will depend on the size of the soil seedbank, which is influenced by the size of the remnant habitat area and fire frequency (Auld & Scott 2004). Post-fire seedlings reach maturity at 2.5-5 years after fire, but replenishment of the soil seed bank is slow because fecundity is low and there are high levels of seed predation by native mammals (Fig. 2, Auld & Scott 1996; Auld & Denham 2001, Llorens 2004). Repeated high-frequency fires are likely to lead to population extinctions as the soil seed bank becomes exhausted (Regan *et al.* 2003). Fire management for conservation of the species is therefore governed by a minimum fire-free interval that is derived from data on the time it takes for the soil seed bank to be replenished (Fig. 2). A recommended minimum interval of 8-12 years is needed to allow the soil seed bank to be replenished after any one fire (Scott *et al.* 1995; DEC 2004). Populations of *G. caleyi* are monitored post-fire over sequential fires (Auld & Scott 2004) to inform any need to modify these minimum fire-free interval prescriptions.

The fire regime at any one location will be influenced by the landscape context (extent and pattern of habitat fragmentation, proximity to human settlements and sources of ignitions), fire mitigation activities designed to protect human life and property, climate and future climate changes, and local factors such as topography, soil and litter moisture and weed invasion. An urban landscape context constrains options for managing fire regimes for conservation of *G. caleyi*, because its location, within asset protection and strategic fire management zones (NSW Rural Fire Service 2006), create an imperative to manage fire for the protection of human life and property. Other contextual factors, such as loss and fragmentation of habitat (Auld & Scott 1996), sources of unplanned ignitions, weeds, disturbance and dumping of rubbish (DEC 2004), compound the threat posed to the species by frequent fires.

Fire management issues affecting *G. caleyi* are representative of those affecting very large numbers of plant species whose persistence depends on the interaction between fire frequency and seed bank accumulation (Keith 1996; Auld & Ooi 2008). High fire frequency has been listed in New South Wales and Victoria as a key threatening process (Table 1). In NSW, while no threat abatement plan has been prepared, the threat is being addressed broadly through strategic fire management systems established under the NSW *Bushfires Act* and the NSW *National Parks and Wildlife Act*. The bushfire planning process requires local fire management plans to identify different zones for asset protection and conservation and specifies environmental assessment protocols to resolve any conflicts that may arise between biodiversity conservation and protection of life and property (NSW Rural Fire Service 2006). A key element of the bushfire planning process is the identification of management thresholds, including minimum fire-free intervals, to maintain local biodiversity (NSW Rural Fire Service 2006). Such thresholds are set for all habitats and each listed threatened species in the area, based on either species-specific data or inferences from other relevant knowledge (Keith *et al*. 2002). Fire management plans undergo regular review, which provides opportunities to update fire regime thresholds and other management strategies based on new knowledge emerging from further research and monitoring of populations of key species.

Some important outstanding questions in relation to fire management remain. For *G. caleyi*, adult plants die and the soil seed bank appears to decay when fire-free intervals exceed about 20-25 years (Fig. 2, DEC 2004). Hence, the upper bounds on fire frequency also need to be addressed in fire planning and are not well understood for the majority of species. In addition, the resilience of the species to frequent fire depends on how much of the seed bank emerges after any one fire. Seed banks of species that hold their seed in the canopy are exhausted after a single fire. This also seems to be the case for some species with soil seed banks, while others retain a residual seed bank (Auld & Denham 2006; Auld *et al.* 2007). This variation is poorly understood, but may be due to variation in germination cues between taxa, spatial variation in the heat, smoke and ash produced in any single fire, and variation in the timing and magnitude of post-fire rainfall. Much more data on these patterns and responses are needed to formulate effective management strategies to mitigate threats posed by particular fire regimes. More work is also needed on the impacts of other components of the fire regime (severity, season and patchiness) and their interactions with fire frequency, so that these components can also be incorporated into fire management for conservation.

Dysfunction of biological interactions: exotic species impacts

Weeds, pathogens and feral animals are major threats to the long-term conservation of biodiversity (Soule 1990; Lonsdale 1999). These exotic species generally act as competitors, predators/herbivores or diseases in natural systems (Table 1), but may also alter disturbance regimes (Mack & D'Antonio 1998). In doing so they may directly eliminate individuals (either as adults or juveniles) or may alter habitat and other biological processes.

In much of arid and semi-arid (??) Australia, there has not been extensive clearing of habitat, unlike the more coastal regions. Instead, arid landscapes have undergone a number of fundamental modifications related to invasions of alien species that have had long term impacts on the biodiversity. Australia has the largest mammal extinction rates of any continent (Morton 1990), much of which has been attributed to introductions of feral predators including the Cat and the Red Fox (///)(Dickman 1996). Since the 1850s, introduced herbivores (sheep, rabbits and goats) have eliminated or greatly reduced recruitment in a range of palatable plant species with life spans varying from 30-50 to over 200 years (Crisp & Lange 1976; Crisp 1978; Chesterfield & Parsons 1985; Lange & Graham 1983; Auld 1993, 1995a, b; Woodell 1990; Tiver & Andrew 1997). Examples include species of *Acacia*, *Casuarina*, *Hakea*, *Myoporum* and *Callitris*. Typically, the current size and/or age distributions of these plant populations in these parts of Australia are heavily skewed, with no to few juvenile or young adult plants present (Fig. 3). While emergence of new seedlings or vegetative suckers is widespread and common in many of the affected taxa, albeit episodic (see Auld 1993, 1995a, b), these recruits do not persist under the current grazing regime. Due to the longevity of the remaining adult plants, none of these species has yet gone globally extinct, but many populations have been lost and others have undergone major declines, altering the structure of the habitats and landscapes of which they are a part. The key remaining questions concern the development and implementation of a management strategy to initiate and sustain plant recruitment in this landscape. This must be initiated before the irreversible collapse of these perennial shrubs and trees, and the communities for which they are critical components, occurs.

Threat abatement plans have been initiated to address the threats from a number of feral animals and weeds. The NSW threat abatement plans for the Red Fox (NPWS 2001) and Bitou Bush (*Chrysanthemoides monilifera* ssp. *rotundata*) and Boneseed (*Chrysanthemoides monilifera* ssp. *monilifera*) (DEC 2006) involve extensive poison baiting and weed control programs targeted, respectively, to reduce Red Fox populations and weed infestations where predation and competition from these species threatens the persistence of threatened species. The control programs, combined with detailed monitoring, are likely to benefit a wide range of species.

While there is currently no threat abatement plan for the European Rabbit (*Oryctolagus cuniculus*), biological control methods have been under development for some decades due to the significance of rabbits as agricultural pests. The abundance of rabbits was greatly reduced in arid and semi-arid Australia by the accidental release of Rabbit Calicivirus Disease (RCD) in 1995 (Pech & Hood 1998). This provided an opportunity for conservation managers to reduce the habitat available to this exotic species by ripping or filling in warrens in order to minimise recovery of rabbit populations. At the same time it allowed an examination of whether or not successful recruitment of perennial plants could occur while rabbit numbers were low. Long-term monitoring and treatment plots have been established to provide guidance on whether RCD impacts on rabbit abundance in combination with warren destruction are sufficient to alleviate the impact of the rabbit as a threat to plant recruitment (Denham & Auld 2004). This work suggests that there has been some recruitment over large areas where warrens have been destroyed, but that this recruitment has been limited by a long-term drought. If rabbit numbers can be kept low until substantial rains occur, recruitment of perennial shrubs could be more widespread. Further management action is also needed to ensure the survival of any such recruits leading up to the next drought, as it is at that time that young plants will be exposed to heavy grazing pressure, particularly from rabbits, goats and sheep. This highlights the need for both long-term (decadal) management action and monitoring of impacts in systems where episodic rainfall is an important driver of population processes.

Biodiversity and a changing climate – a broad scale threat

Climate change is the most pervasive, least understood and least predictable of threatening processes. Its impacts on biodiversity are likely to be many and varied (Hughes 2003). The responses of species and ecosystems will depend on complex processes and interactions (e.g. Keith *et al.* 2008); including physiological tolerance, population and dispersal dynamics, landscape dynamics, species interactions and existing stressors such as habitat loss. Climate change will exacerbate a number of existing threats, for example, by altering the frequency and intensity of fire (Cary *et al.* 2006) and flood events (Roshier *et al*. 2001). Other impacts will be novel, for example CO_2 impacts on plants and their relationships with competitors and herbivores, temperature impacts on animal physiology (Kearney & Porter 2004) or seed germination patterns.

Species may have some capacity to tolerate some degree of change in climate and adapt *in situ* to the changes and /or move in response to the changes. The degree of phenotypic, behavioural or genotypic variation within species and the capacity to cope with changing interactions (specialists to generalists) is very poorly known. What is better known is that many taxa will not have the capacity to disperse at the rate required to remain in a comparable climatic zone to the one they currently occupy (Midgley *et al.* 2006; Morin *et al.* 2008).

Unlike other threatening processes, the development of strategies for abating threats to biodiversity from climate change has only just begun (Dunlop & Brown 2008). A lack of knowledge poses a fundamental impediment to development of an effective abatement strategy. We suggest that there are several key elements to improving understanding of the risks posed by climate change to biodiversity along with three integrated steps for possible adaptive management in response to climate change (Fig. 4). Firstly, a risk assessment framework is needed that integrates habitat and bioclimatic data with demographic, physiological and genetic processes. This would improve our capacity to predict likely climate change impacts and hence to identify those species or ecological communities most at risk beyond the more predictable ones such as those on mountain tops and littoral zones. Secondly, an improved understanding is needed of how climate change will interact with disturbance regimes that drive ecological systems and how it will affect existing threats to these systems. Models predicting changes in bushfire risk (Hennessy *et al.* 2005) and consequent changes in fire regime (Cary *et al.* 2006) provide a clear example of how disturbance regimes may change with climate. Thirdly, there will be some novel effects of a changing climate on biota, including impacts of higher CO_2 levels, physiological heat impacts on plants and animals, increasing frequencies of extreme weather events and impacts on life history process that are directly controlled by temperature and/or seasonality of rainfall (e.g. reproductive phenology, seed germination). The consequences of these changes urgently need examination in complex field environments.

Protection of biodiversity will involve the integration of three key conservation elements (Fig. 4). Eliminating or at the least minimising further loss of remnant habitats will reduce synergistic pressures on species exposed to multiple threats, as well as providing refugia or stepping stones for dispersal to sites that may become more suitable habitat in future. Management of existing threats in remnant habitats will enhance the capacity of species to persist *in situ* and form the base from which any migration events may occur. This will include management of weeds, pathogens and pest animals and disturbance regimes. Finally, we need to manage the capacity and opportunity of species to move across the landscape (should they be able to) or potentially assist those that cannot persist at their current locations and are unable to migrate sufficiently rapidly without intervention. This will require a greater understanding and protection of keystone dispersal agents, such as the Emu (*Dromaius novaehollandiae*) (Calviño-Cancela *et al.* 2006, 2008) and flying foxes (Shilton *et al.* 1999; McConkey & Drake 2006). It will also require an understanding of the elements needed for landscape connectivity for different types of taxa and the development and field testing of techniques for restoration (repair or reconstruction) of ecological communities. In line with this, the need and capacity for translocation of both species and ecological communities also requires consideration. Of all threatening processes, climate change is the most challenging for which to develop adaptive management strategies for abatement. Innovative approaches will be needed to spread risks, monitor responses and assess alternative management options.

Conclusions

The diagnosis and management of threatening processes is a key approach to biodiversity conservation above the species level. Scientific research plays an important role in both the diagnosis of threatening processes that affect biodiversity and the design of management strategies to mitigate their impacts. An understanding of how threats affect species persistence (e.g. through disruption of life cycles, competition, predation and habitat change) is fundamental to dealing with those threats effectively. Although many challenges and uncertainties remain, developing the knowledge base will allow a suite of amelioration measures to be developed progressively in an adaptive management framework. Implementation of threat abatement actions will require ongoing resources and the broad-scale benefits that may result should be viewed as a synergism between management of threats and the management of threatened species or ecological communities.

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Table 1. Taxonomy of threats listed under legislation at the Commonwealth (EPBC Act), NSW (TSC and Fisheries Acts) and Victoria (FFG Act). Numbers are the key threatening process listed under each group.

Threat Grouping	EPBC Act	TSC & Fisheries Acts	FFG Act
Habitat Loss and Degradation			
Loss	1	5	1
Degradation	1	5	13
Climate Change	1	1	1
Changes to disturbance regimes			
Fire	0	1	2
Water	0	1	2
Dysfunction of biological interactions			
Competition	0	5	4
Herbivory	2	3	2
Predation	6	7	2
Competition & Predation	0	1	2
Herbivory & Predation	1	1	0
Pathogens	3	3	4
Pollination	0	2	2
Over exploitation			
Direct	0	1	1
By-catch	2	2	1

Captions for Figures

Figure 1. Commonality of listing of key threatening processes across three Australian jurisdictions. Australia (EPBC Act), NSW (TSC and Fisheries Acts) and Victoria (FFG Act).
Filled – unique to each jurisdiction;
open – common to all jurisdictions;
vertical lines - common to Commonwealth and NSW;
hatching –common to Commonwealth and Victoria;
sloped lines – common to NSW and Victoria

Figure 2. Fire impacts on the life history of the endangered shrub *Grevillea caleyi*.

Figure 3. Stylised example of three possible classes of current size distributions in perennial plants (e.g. *Acacia, Alectryon, Callitris, Casuarina, Hakea, Myoporum*) from arid and semi arid Australia to illustrate the effect of loss of recruitment under exotic grazing pressure.
(a) successful ongoing recruitment is occurring;
(b) no successful ongoing recruitment, only large mature plants present;
(c) no successful ongoing recruitment, but evidence of episodic recruitment of juveniles that do not progress to adulthood.

Figure 4. Key elements for dealing with an understanding of impacts and mitigation of impacts for biodiversity under a changing climate.

Fig. 1

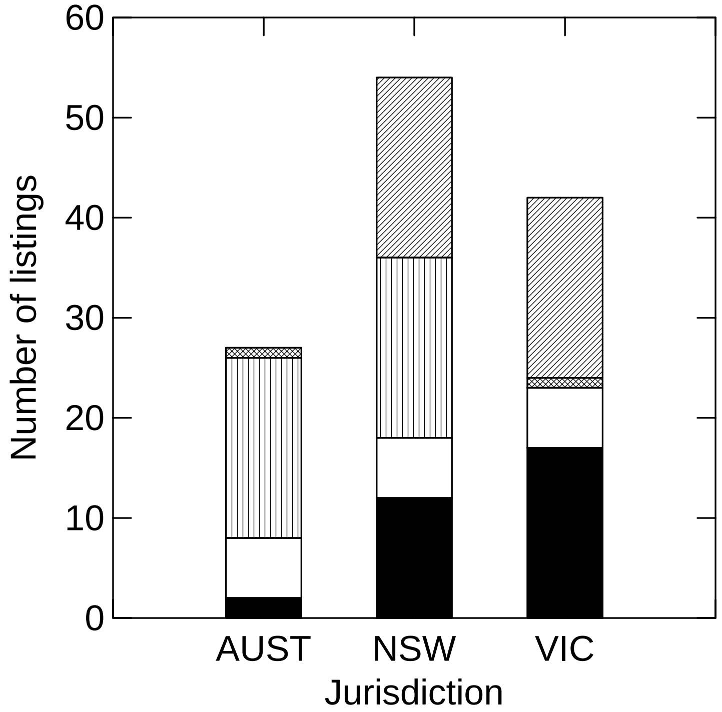

60
50
40
30
20
10
0
Number of listings
AUST
NSW
VIC
Jurisdiction

Fig. 2

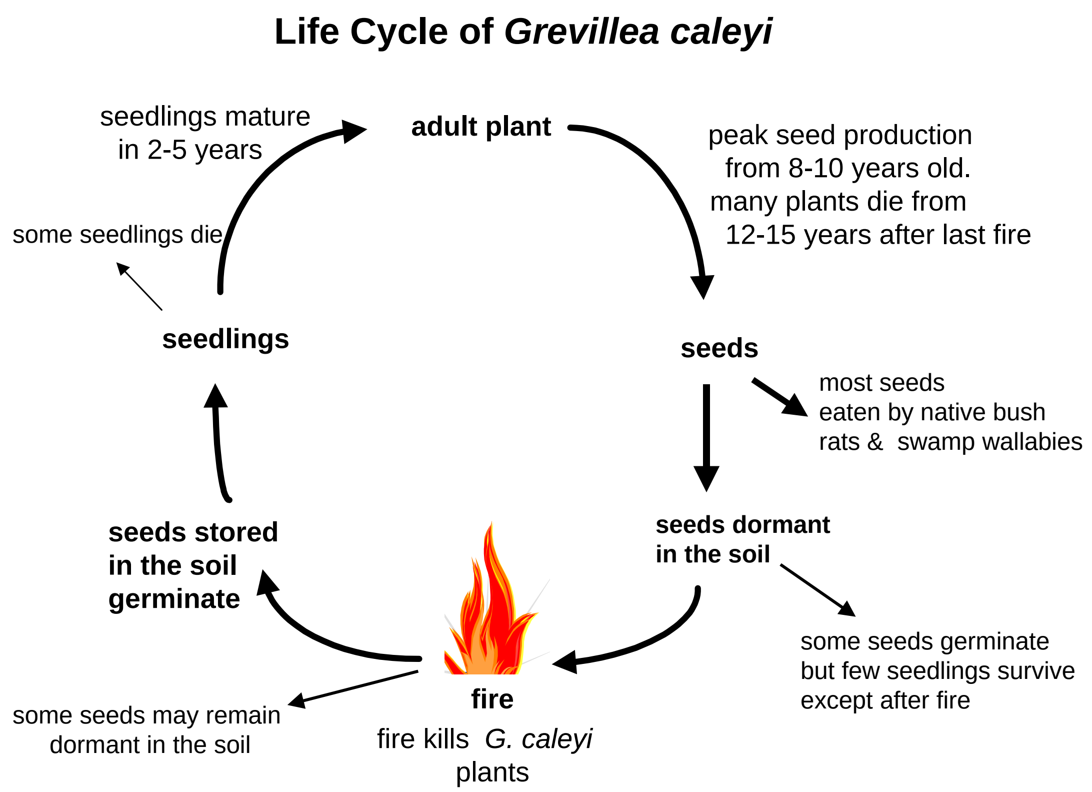

Life Cycle of *Grevillea caleyi*
seedlings mature in 2-5 years
adult plant
peak seed production from 8-10 years old. many plants die from 12-15 years after last fire
some seedlings die
seedlings
seeds
most seeds eaten by native bush rats & swamp wallabies
seeds dormant in the soil
some seeds germinate but few seedlings survive except after fire
seeds stored in the soil germinate
fire
fire kills *G. caleyi* plants
some seeds may remain dormant in the soil

Fig. 3

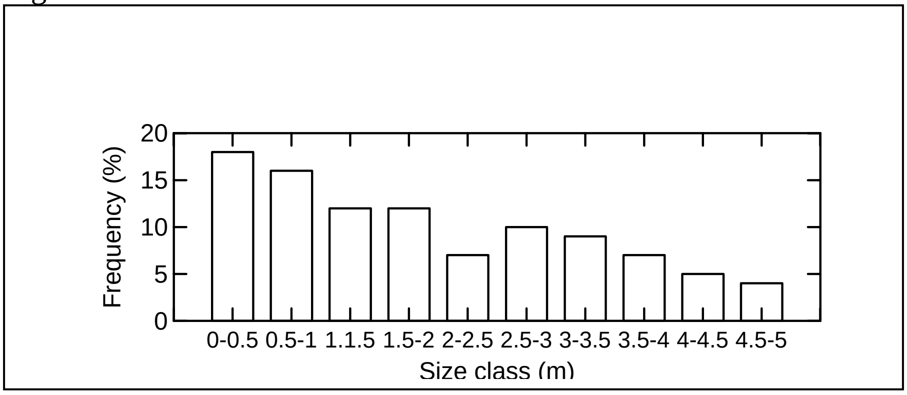
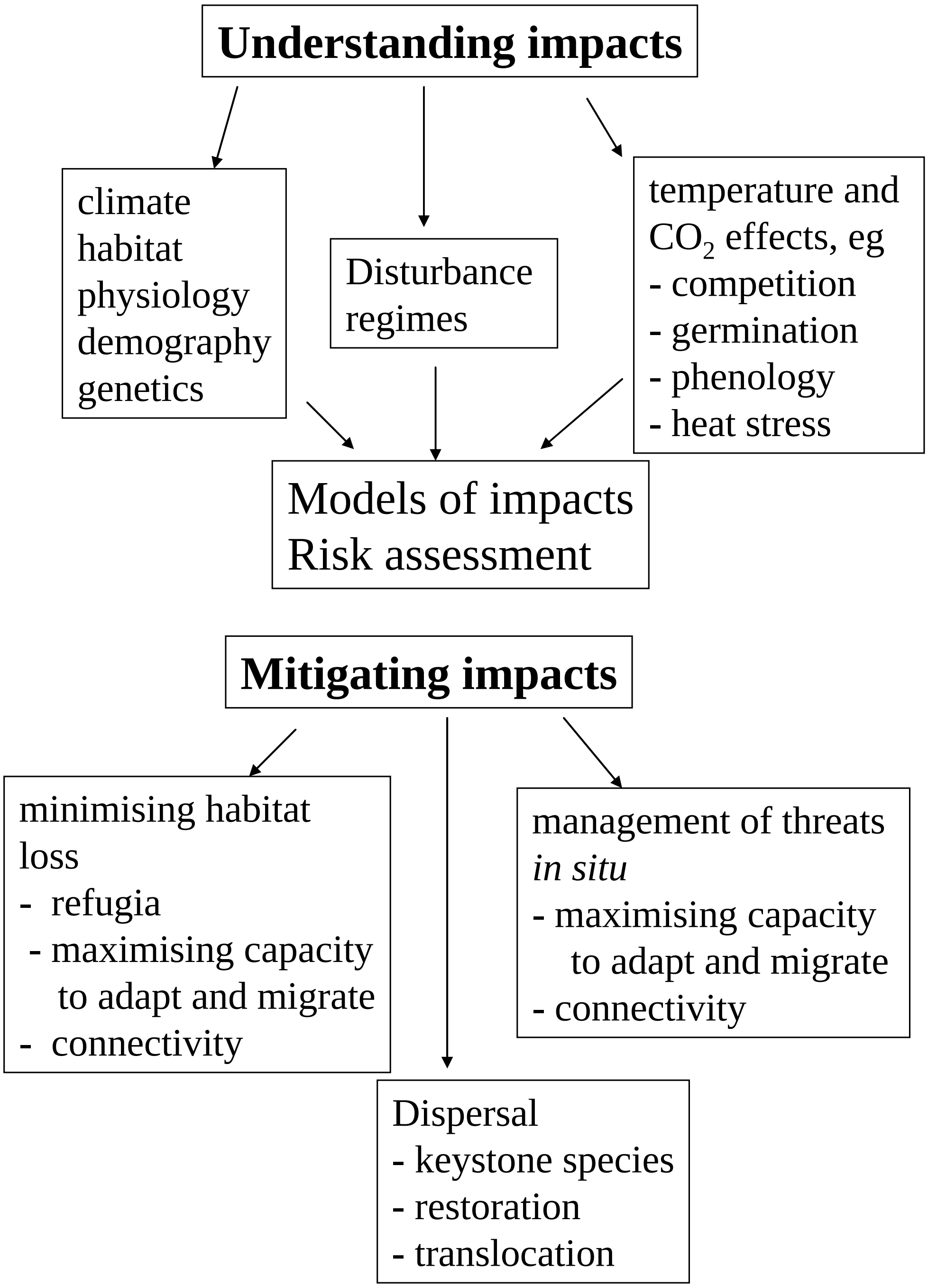

20
15
10
5
0
Frequency (%)
0-0.5 0.5-1 1.1.5 1.5-2 2-2.5 2.5-3 3-3.5 3.5-4 4-4.5 4.5-5
Size class (m)

Fig. 4


Understanding impacts
climate
habitat
physiology
demography
genetics
Disturbance regimes
temperature and CO_2 effects, eg
- competition
- germination
- phenology
- heat stress
Models of impacts
Risk assessment
Mitigating impacts
minimising habitat loss
- refugia
- maximising capacity to adapt and migrate
- connectivity
management of threats *in situ*
- maximising capacity to adapt and migrate
- connectivity
Dispersal
- keystone species
- restoration
- translocation