Guidelines for interpreting listing criteria for species, populations and ecological communities under the NSW Threatened Species Conservation Act

Version 1.4
NSW Scientific Committee, November 2014

The NSW Scientific Committee has prepared these guidelines to assist interpretation of the concepts and terms in the listing criteria given in the TSC Regulation 2010. They should always be used in conjunction with the TSC Act 1995 and the TSC Regulation (listing criteria) 2010. If cases arise where advice given in the guidelines is in apparent conflict with the Act or the Regulation, the Act and Regulation will apply.

1 Introduction

The Threatened Species Conservation Regulation (listing criteria) 2002 introduced explicit criteria to guide listing decisions for threatened species, populations and ecological communities under the *NSW Threatened Species Conservation (TSC) Act 1995*, and was replaced by the Threatened Species Conservation Regulation 2010. To be eligible for listing as threatened, a species, population or ecological community must, in the opinion of the Scientific Committee, meet one or more of the relevant Clauses specified in the Regulation.

Many aspects of the assessment and listing process under the TSC Act are modelled on the IUCN Red List categories and criteria (IUCN 2001). Moreover, the listing criteria and the definitions of terms in the TSC Regulation 2010 closely follow the wording of criteria and definitions in IUCN (2001). These provide an explicit, objective and widely understood framework that represents international best practice for classifying species according to their extinction risk. The Red List criteria are a product of extensive consultation with a large community of international scientists and have undergone a long history of research, development and testing (Mace & Lande 1991, Mace *et al.* 2008). They are applied worldwide by an extensive network of specialists across all taxonomic groups excluding micro-organisms. Interpretation of the Red List criteria is supported by scientific advice from an international Standards and Petitions Working Group, which publishes and regularly updates detailed guidelines to assist application of the criteria across the full range of biological taxa (IUCN 2014).

The categories and criteria for listing species under the TSC Act have a very close relationship with those developed for the IUCN Red List. The three categories of threat under the TSC Act (Critically Endangered, Endangered and Vulnerable) mirror those used in the IUCN Red List for threatened species (IUCN 2001). The listing criteria and terms defined in the TSC Regulation 2010 are also based closely on those developed by IUCN (2001). In addition, the IUCN Red List criteria for species have been adapted to assess the eligibility of populations and ecological communities for listing under the TSC Act. The criteria for listing ecological communities in the TSC Regulation 2010 is comparable with the Red List Criteria for ecosystems (Keith *et al.* 2013). By adopting similar listing criteria, NSW benefits from this substantial intellectual capital associated with the IUCN Red List, ensuring world’s best-practice assessments of species, populations and ecological communities potentially at risk of extinction in NSW. The close parallels between listing criteria for the TSC Act and the IUCN Red List also ensure a high degree of compatibility between listings in NSW and those on the global Red List.

The NSW Scientific Committee has prepared these guidelines to assist interpretation of the concepts and terms in the listing criteria given in the TSC Regulation 2010. They draw extensively from relevant material in the IUCN Red List Guidelines (IUCN 2014). The TSC listing guidelines address species, populations and ecological communities in separate sections, although cross-references are given where concepts and terms are common to these different entities. Where possible, examples are included to illustrate points of interpretation. The Committee intends to update these guidelines periodically to include additional examples and address new questions of interpretation as they arise. These guidelines should always be used in conjunction with the TSC Act 1995 and the TSC Regulation (listing criteria) 2010. If cases arise where advice given in the guidelines is in apparent conflict with the Act or the Regulation, the Act and Regulation will apply.

2 Assessments of Species

Nominations of species for listing as threatened under the TSC Act 1994 must be assessed under Clauses 6-10 of the TSC Regulation 2010. A species is eligible for listing if it meets any one of these clauses.

2.1 Clause 6 – reduction in population size

Clause 6 is based on IUCN (2001) criterion A. The basis for Clause 6 is the declining population paradigm (Caughley 1994): a declining population is more likely to become extinct than one that is stable or increasing (see also Mace & Lande 1991, Keith 1998, Mace *et al.* 2008, IUCN 2014). Species that have undergone large reductions or are likely to undergo large reductions in the future are likely to be at greater risk of extinction than those that have undergone or are likely to undergo smaller reductions. To be eligible for listing under Clause 6, species that have undergone a sufficiently large reduction within the relevant past time frame need not exhibit evidence of a continuing decline.

2.1.1 Reduction in population size

Reductions in population size refer to a decrease in the total number of individuals of the species in NSW over a specified time frame. Not all populations of a species may be changing at the same rate or in the same direction. To assess the overall reduction in the total population of a species, trends in local populations must be weighted according to their relative size and averaged. Thus, a species may not meet the criteria for reduction even though there is a very large reduction in one population, so long as the largest populations of the species are stable or increasing. Conversely, a species may meet the criteria for reduction if its largest population has undergone a large reduction, even though all other populations are stable or increasing. Where trends in all component populations of a species have not been estimated, a representative sample may be used to estimate any overall reduction in the total species population.

2.1.2 Measures of reduction

Clause 6 indicates that reductions in population size may be assessed in different ways according to Clause 4. Under Subclause 4a, an **index of abundance appropriate to the taxon** may include a range of direct or indirect measures including direct counts or estimates of all types of individuals, or direct counts of individuals belonging to particular life stages (e.g. mature individuals). Other indices that may be appropriate include projective cover of foliage or canopies, biomass, frequency of collections, observations or captures, harvest volumes, range size, area of suitable habitat, etc. Ideally, assessments of reduction should justify the choice of an appropriate index of abundance.

Under Subclause 4b, a reduction in population size may be based on **geographic distribution, habitat quality or diversity** or **genetic diversity**. **Geographic distribution** is defined under Clause 23 of the TSC Regulation 2010 (see sections 2.2.1-2.2.2). **Habitat quality** (also known as habitat suitability) refers to the environmental conditions that govern a species' rates of survival, growth and reproduction, and hence the ability of its populations to persist. A reduction in habitat quality will therefore usually be associated with an increase in extinction risks due to a decline in survival, growth and/or reproduction. Where knowledge of the relationship between these demographic rates and the environment is limited, habitat quality may be inferred from variation in the abundance of the species across different environments. This type of inference requires careful evaluation because it assumes that populations will be in approximate equilibrium with their environment. **Habitat diversity** refers to the range of environmental conditions and resources that an organism is able to exploit to sustain its survival, growth and reproduction. A reduction in the range of available conditions or resources (e.g. resulting from loss of particular food sources or decline in types of nesting sites available) is likely to reduce the ability of a population to persist either as relics in habitat refuges or through behavioural adaptation to avoid adverse conditions and processes. **Genetic diversity** refers to the level of heritable (genetic) variation represented in the variety of alleles and genotypes within a species or population (Frankham *et al.* 2002). Genetic diversity may be structure between and within populations of a species and is usually measured by the frequencies of genotypes and alleles, the proportion of polymorphic loci, the observed and expected heterozygosity or the allelic diversity (Toro & Caballero 2005). A reduction in the genetic diversity may reduce the fitness of a species to persist in its present environment and reduce its evolutionary potential to adapt to environmental change.

2.1.3 Magnitude of reductions

To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, a species must have undergone or be projected to undergo **very large** (Subclause 6a)**, large** (Subclause 6b) **or moderately large** (Subclause 6c) **reductions** in population size. The corresponding listing criteria in IUCN (2001) provide indicative guidance for quantitative interpretation of these terms (Table 1). Past and projected reductions in population size may be interpreted under Subclauses 6a, 6b and 6c using more stringent numerical thresholds of criterion A1 (IUCN 2001) if the causes of reduction are clearly reversible AND understood AND ceased. If any of these conditions do not apply, the standard thresholds in criteria A2, A3 and A4 are appropriate (IUCN 2001).

Table 1. Corresponding thresholds for reductions in population size for the TSC Regulation 2010 and the IUCN (2001) Red List criteria .

Category of threat	Requirement under Clause 6 of TSC Regulation 2010	Thresholds for reduction under criteria A2, A3 and A4 of IUCN (2001)	Thresholds for reduction under criterion A1 of IUCN (2001)
Critically Endangered	very large	≥80%	≥90%
Endangered	large	≥50%	≥70%
Vulnerable	moderately large	≥30%	≥50%

2.1.4 Time frames for assessing reductions

Reductions in population size must be assessed over a **time frame appropriate to the life cycle and habitat characteristics of the taxon**. Based on IUCN (2001), a time frame appropriate to the life cycle is three generation lengths or 10 years, whichever is the longer. Generation length is defined by IUCN (2001, 2014) (see Box 1). In most cases, habitat characteristics will not alter the appropriate time frame determined from generation length. In exceptional circumstances, where an appropriate time frame for assessing reductions cannot be inferred from generation length, turnover in habitat may be used as a proxy for generation length. Usually, this will only be possible in taxa that have a direct life-history dependence on cyclical habitat dynamics (e.g. freshwater amphibians inhabiting ephemeral desert streams).

Box 1. Generation length

Definition of Generation Length (after IUCN 2001)

Generation length is the average age of parents of the current cohort (i.e. newborn individuals in the population). Generation length therefore reflects the turnover rate of breeding individuals in a population. Generation length is greater than the age at first breeding and less than the age of the oldest breeding individual, except in taxa that breed only once. Where generation length varies under threat, such as the exploitation of fishes, the more natural, i.e. pre-disturbance, generation length should be used.

Methods for estimating and inferring generation length (after IUCN 2014)

In general, time-based measures in the criteria are scaled for the different rates at which taxa survive and reproduce, and generation length is used to provide this scaling. The current definition of generation length has been widely misunderstood, and there are difficulties when dealing with very long-lived taxa, with taxa having age-related variation in fecundity and mortality, with variation in generation length under harvesting, with environmental changes and variation between the sexes. Some of the different acceptable methods for estimating generation length are included here.

It is also appropriate to extrapolate information such as a generation length from closely related well-known taxa and to apply it to lesser-known and potentially threatened taxa.

Formally, there are several definitions of generation length, including the one given above; mean age at which a cohort of newborns produce offspring; age at which 50% total reproductive output is achieved; mean age of parents in a population at the stable age distribution; and time required for the population to increase by the replacement rate. All of these definitions of generation length require age- and sex-specific information on survival and fecundity, and are best calculated from a life table (e.g., option 1 below). Depending on the taxon concerned, other methods may provide a good approximation (e.g., options 2 and 3). Care should be taken to avoid estimates that may bias the generation length estimate in a non-precautionary way, usually by under-estimating it. Generation length may be estimated in a number of ways:

1. the average age of parents in the population, based on the equation

$$G = \Sigma\, x l_x m_x / \Sigma\, l_x m_x$$

where the summations are from age (x) 0 to the last age of reproduction; m_x is (proportional to) the fecundity at age x; and l_x is survivorship up to age x (i.e., $l_x = S_0 \cdot S_1 \cdots S_{x-1}$ where S is annual survival rate, and l_0=1 by definition).

This formula is implemented in an associated spreadsheet file (Generation length.xls, see IUCN 2014). To use this formula, follow the instructions in the file, noting the exact definitions of the parameters required.
2. 1/adult mortality + age of first reproduction. This approximation is useful if annual mortality after the age of first reproduction is well known, and if mortality and fecundity do not change with age after the age of first reproduction (i.e., there is no senescence). Many species exhibit senescence, with mortality increasing and fecundity decreasing with age; for these species, this formula will overestimate generation length (in such cases, use the spreadsheet mentioned above). For age of first reproduction, use the age at which individuals first produce offspring in the wild (which may be later than when they are biologically capable of breeding), averaged over all individuals or all females. If first reproduction typically occurs by 12 months, use 0, not 1; if it occurs between 12 and 24 months, use 1, etc.
3. age of first reproduction + *z* * (length of the reproductive period), where *z* is usually <0.5, depending on survivorship and the relative fecundity of young vs. old individuals in the population. For age of first reproduction, see (2) above. This approximation is useful when ages of first and last reproduction are the only available data, but finding the correct value of *z* may be tricky. In general, for a given length of reproductive period, *z* is lower for higher mortality during reproductive years and it is higher for relative fecundity skewed towards older age classes. To see how generation length is affected by deviation from these assumptions, you can use the spreadsheet mentioned above. Note that the length of the reproductive period depends on longevity in the wild, which is not a well-defined demographic parameter because its estimate often depends very sensitively on sample size.
4. for partially clonal taxa, generation length should be averaged over asexually and sexually reproducing individuals in the population, weighted according to their relative frequency.
5. for plants with seed banks, use juvenile period + either the half-life of seeds in the seed bank or the median time to germination, whichever is known more precisely. Seed bank half-lives commonly range between <1 and 10 years. If using the spreadsheet for such species, enter seed bank as one or several separate age classes, depending on the mean residence time in the seed bank.

Options 2 and 3 are still appropriate if the interbirth interval is more than one year; a more precise calculation can be made in this case by using the spreadsheet (see above), and for each age class averaging fecundity over all individuals (or females) in that age class (regardless of whether they actually reproduced at that age). The turnover rate mentioned in the definition is not directly related to the interbirth interval; it reflects the average time it takes one group of breeding individuals to be replaced by its progeny.

It is not necessary to calculate an average or typical generation length if some subpopulations of the taxon differ in terms of generation length. Instead, use each subpopulation's generation length to calculate the reduction over the appropriate number of generations, and then calculate the overall population reduction (for criterion A) or overall estimated continuing decline (for criterion C1) using a weighted average of the reductions calculated for each subpopulation, where the weight is the size of the subpopulation 3 generations ago

2.1.5 Types of evidence for reduction

Both Clause 6 of the TSC Regulation 2010 and criterion A of IUCN (2001) refer to different types of direct and indirect evidence for reductions in population size. To qualify for listing, reductions of the above magnitude must be **estimated, projected, inferred or reasonably suspected**. IUCN (2014) provides explicit definitions for these terms (Box 2).

Box 2. Types of evidence (after IUCN (2014) Section 3.1)

Observed: information that is directly based on well-documented observations of all known individuals in the population.

Estimated: information that is based on calculations that may include statistical assumptions about sampling, or biological assumptions about the relationship between an observed variable (e.g. an index of abundance) to the variable of interest (e.g. number of mature individuals). These assumptions should be stated and justified in the documentation. Estimation may also involve interpolation in time to calculate the variable of interest for a particular time step (e.g. a 10-year reduction based on observations or estimations of population size 5 and 15 years ago). For examples, see discussion under criterion A (section 4.5 of IUCN 2014).

Projected: same as "estimated", but the variable of interest is extrapolated in time towards the future. Projected variables require a discussion of the method of extrapolation (e.g. justification of the statistical assumptions or the population model used) as well as the extrapolation of current or potential threats into the future, including their rates of change.

Inferred: information that is based on indirect evidence, on variables that are indirectly related to the variable of interest, but in the same general type of units (e.g. number of individuals or area or number of subpopulations). Examples include population reduction (A1d) inferred from a change in catch statistics, continuing decline in number of mature individuals (C2) inferred from trade estimates, or continuing decline in area of occupancy (B1b(ii,iii), B2b(ii,iii)) inferred from rate of habitat loss. Inferred values rely on more assumptions than estimated values. For example, inferring reduction from catch statistics not only requires statistical assumptions (e.g. random sampling) and biological assumptions (about the relationship of the harvested section of the population to the total population), but also assumptions about trends in effort, efficiency, and spatial and temporal distribution of the harvest in relation to the population. Inference may also involve extrapolating an observed or estimated quantity from known subpopulations to calculate the same quantity for other subpopulations. Whether there are enough data to make such an inference will depend on how large the known subpopulations are as a proportion of the whole population, and the applicability of the threats and trends observed in the known subpopulations to the rest of the taxon. The method of extrapolating to unknown subpopulations depends on the criteria and on the type of data available for the known subpopulations. Further guidelines are given under specific criteria (e.g. see section 5.8 (IUCN 2014) for extrapolating population reduction for criterion A assessments).

Suspected: information that is based on circumstantial evidence, or on variables in different types of units, for example, % population reduction based on decline in habitat quality (A1c) or on incidence of a disease (A1e). For example, evidence of qualitative habitat loss can be used to ***infer*** that there is a qualitative (continuing) decline, whereas evidence of the amount of habitat loss can be used to ***suspect*** a population reduction at a particular rate. In general, a suspected population reduction can be based on any factor related to population abundance or distribution, including the effects of (or dependence on) other taxa, so long as the relevance of these factors can be reasonably supported.

2.1.6 Inferring changes in population size from changes in geographic distribution

A reduction in population size may be based on a decline in geographic distribution (Clause 6 in conjunction with Subclause 4b). The assumptions made about the relationship between habitat loss and population reduction have an important effect on the outcome of an assessment. In particular, the simplest assumption, that the relationship is linear, is not often true and may lead to over- or under-listing. IUCN (2014) gives the following examples to illustrate this. The population of a bird species may not be reduced by 50% if 50% of its habitat is lost (perhaps because it will colonise new habitats). Or, reduction may happen mostly in lower-density areas, leading to a faster decline in range than in population size. Conversely, if reductions occur predominantly in high-density areas, population reduction will be faster than can be deducted from range contraction (decrease in EOO) (Rodríguez 2002). Similarly, the population of a hollow-dependent mammal may be reduced by more than 50% if 50% of its habitat is lost due to logging in productive breeding sites that removes many suitable tree hollows.

In all cases, an understanding of the taxon and its relationship to its habitat, and the threats facing the habitat is central to sensible use of inference and projection in making the most appropriate assumptions about habitat loss and subsequent population reduction. These assumptions should be justified and documented.

IUCN (2014) notes that available population data may sometimes contradict habitat data (e.g. habitat seems to be declining in quality, but population numbers are stable). This can occur because: (1) one set of data is uncertain, biased, or dated, or (2) the population has a lagged response to loss of habitat (likely if generation time is long). In the first case, the assessors must use their judgement to decide which data are more certain. The implications of a possible lagged response in abundance to loss of habitat should, however, be considered when evaluating criterion 6 in relation whether the species “is likely to undergo” a reduction of particular magnitude over the relevant future time frame. For example, if population reduction in the last 3 generations is 30% based on abundance data, which are adequate to determine trends, then the species should be listed as VU, even if habitat loss in the same period was 60%. However, if a lagged response in abundance to loss of habitat (i.e. the impact of habitat loss at present leads to a future reduction in the number of mature individuals) is likely, then the population may be expected to decline further in the future (even if habitat loss has stopped). In this case, listing as EN should be considered if the 60% loss of habitat is inferred to lead to a 60% reduction in the population within the next 3 generations.

2.1.7 Calculating reductions in population size

IUCN (2014) sections 4.5 and 5 provide extensive advice on the interpretation and calculation of reductions in population size.

For taxa with more than one population or area of occurrence, reductions should be averaged across all populations and areas or across a sample of all populations and areas. Across the entire range of a species, some populations may be increasing, some may be declining and others may be stable. In such situations, the change should be weighted by the size of the population; for example, declines in large populations will outweigh increases in small populations. Box 3 shows an example calculation.

Box 3. Protocol for estimating population reduction (after IUCN (2014) section 5.8)

For species with multiple populations or occurrences, it is recommended that the available data on past reduction be presented in a table that lists all known populations, occurrences or parts of the range, and gives at least two of the following three values for each subpopulation:

1. the estimated abundance at a point in time close to the required base line for estimating population reduction (e.g. 3 generations ago), and the year of this estimate;
2. the most recent estimated abundance and its year;
3. suspected or inferred reduction (in %) over the last 3 generations.

If there are estimates of abundance for years other than those reported in (1) or (2), these should also be reported in separate columns of the same table. Any qualitative information about past trends for each population should be summarised in a separate column, as well as quantities calculated based on the presented data (see examples in IUCN 2014, section 5.8).

There are three important requirements:

a) The values should be based on estimates or indices of the number of mature individuals. If the values are based on indices, a note should be included that explains how the index values are expected to relate to the number of mature individuals, and what assumptions are necessary for this relationship to hold.

b) The populations or occurrences should be non-overlapping. This does not mean that there is no or infrequent dispersal among populations. The point of this requirement is to avoid double-counting as much as possible. ‘Occurrences’ are any type of non-overlapping subunits of the species, such as parts of the species’ range

c) Together, the populations or occurrences should include all of the species within NSW. If this is not possible, a “population” named *Remainder* should include an estimate of the total number of mature individuals not included in the listed populations. This estimate, like others, can be uncertain (see below).

In many cases, there will be uncertainty, because the abundances are not known precisely, are in different units for different populations, or are available only from one or few populations. These cases are discussed below in a section on *Dealing with uncertainty.*

IUCN (2014) gives several examples of calculating population reductions under different scenarios of available data. One example for a species with three occurrences (Pacific, Atlantic & Indian) and a generation length of 20 years is reproduced below. The assessment date was 2001 (i.e. for these examples, the “present” is 2001 and "three generations ago" is 1941).

Population	**Past**	**Present**	**Notes**
Pacific	10 000 (1930s)	7 000 (1995)	most of the decline in the last 20 yr
Atlantic	8 000 (1975)		believed to have been stable
Indian	10 000 (1961)	4 000 (1981)	

In this case, the “past” and “present” population estimates are not from the same year for all populations. Thus, it is necessary to make projections in order to estimate reduction for each population in the same time period. There are several types of projection. For example it is necessary to project the population from the “past” census (in the 1930s) to 1941 (3 generations ago) as well as from the most recent census (in 1995) to the present.

Any information about past trends can be valuable in making such projections (as in the “Notes” in the example). For instance, given that most of the decline in the Pacific subpopulation has occurred in recent years, the estimate in the 1930s can be assumed to also represent the population in 1941 (3 generations ago). However, in this case, it is necessary to make a projection from the most recent estimate (in 1995) to 2001. If the estimated decline from 10000 to 7000 occurred in 20 years, then assuming a constant rate of decline during this period, annual rate of decline can be calculated as 1.77% [$1-(7000/10000)^{(1/20)}$], giving a projected decline of about 10.1% in the 6 years from the last census (in 1995) to 2001, and a projected 2001 population of 6290 ($=7000*(7000/10000)^{(6/20)}$). This means a 3-generation decline of 37% (10000 to 6290).

When there is no evidence that the rate of decline is changing, exponential decline can be assumed. For example, for the “Indian Ocean” subpopulation, the 20-year reduction from 1961 to 1981 is 60% per generation; corresponding to 4.48% per year [$-0.0448=(4000/10000)^{(1/20)}-1$]. Thus, 3-generation decline can be estimated as 93.6% [$-0.936=(4000/10000)^{(60/20)}-1$]. Another way to calculate the 3-generation decline is based on annual rate of change, which is 0.9552 (1-4.48%). Thus, 60-year population change is $0.9552^{60}=0.064$; i.e. only 6.4% of the population will remain after 60 years, which is a 93.6% decline]. The population size 3 generations ago can thus be estimated as 25000 [=10000/(1-0.6)], and the current population as 1600 [=4000*(4000/10000)].

It is important to note that the assumption of the pattern of decline can make an important difference to the estimated reduction, and that exponential decline is not the only possible assumption. See the discussion in section 5 (*Dealing with uncertainty*).

The “Atlantic” subpopulation has been stable, so a reduction of 0% is assumed. Combining the three estimates, the weighted average of reduction for the taxon is estimated as 63% [(-0.37*10+0*8-0.936*25)/43].

When such projections are used in estimating the overall reduction, the projected declines and projected subpopulation sizes should be given in different columns of the table than those that are used for the data (see completed table below).

Pop.	**Past**	**Present**	**Notes**	**Population 3 gen. ago (est.)**	**Current population (est.)**	**Estimated 3-generation reduction**
Pacific	10000 (1930s)	7000 (1995)	Most of decline in the last 20yr	10000	6290	37.1%
Atlantic	8000 (1975)		Believed to have been stable	8000	8000	0%
Indian	10000 (1961)	4000 (1981)	-	25000	1600	93.6%
Overall				43000	15890	63.0%

As illustrated in Box 3, available data on population reductions may not correspond to the "time frame appropriate to the life cycle and habitat characteristics of the taxon" over which reductions must be assessed against the listing criteria. Interpolation or extrapolation may be required where the data are available for a longer or shorter period than the required time frame for assessing population reductions. In both cases, the best approach is to fit a regression model to the available data and use the appropriate time interval (e.g. between the present year and three generations lengths prior) on the fitted line to calculate the reduction. Fitting a model in this way helps to eliminate some of the variability in the data that may be attributable to natural fluctuations, and which should not be included when estimating population reductions (IUCN 2014). Interpolation or extrapolation will require assumptions about the data and the trend that should be justified with reference to the life history and/or habitat characteristics of the taxon, and the processes driving its decline (e.g. pattern of exploitation, habitat loss, disease spread, disturbance events, etc.). For example, depending on the shape of the data, a linear or exponential regression model may be fitted. Assumptions about the rate of decline remaining constant, increasing or decreasing relative to the observed interval must be justified, especially where population reduction is estimated over long generation times from data over shorter time frames.

2.2 Clause 7 – size of geographic distribution and other conditions

Clause 7 is based on IUCN (2001) criterion B. The basis for Clause 7 is the level of exposure of a species to spatially correlated threatening processes (Mace & Lande 1991, Keith 1998, Mace *et al.* 2008, IUCN 2014). The larger a species' distribution, the more its risk of exposure to threats will be spread across different locations. Conversely, species that have restricted geographic distributions, will have fewer opportunities for persistence because it is more likely that a single threatening process or event will adversely affect the entire species. Clause 7 is indirectly related to the small population paradigm (Caughley 1994), as certain measures of geographic distribution may be proxies for population size (Gaston 1992, Keith 1998). To be eligible for listing under Clause 7, a species must have a sufficiently restricted distribution AND meet additional conditions specified in Subclauses 7d OR 7e. These additional conditions are discussed under section 2.4. This section discusses the interpretation and measurement of geographic distribution.

2.2.1 Geographic distribution

The **geographic distribution** of a species is defined in Clause 23(1) of the TSC Regulation 2010 as 'the area or areas in which a species [or ecological community] occurs, excluding cases of vagrancy.' Geographic distribution may be assessed in a number of different ways, including the extent of occurrence, area of occupancy and area of suitable habitat.

2.2.2 Measures of geographic distribution

Under Clause 7, the geographic distribution of a species may be assessed by estimating the extent of occurrence, the area of occupancy or the area of suitable habitat. Each of these terms is defined in Clause 23(2).

(a) **Extent of occurrence** (EOO) is defined in Clause 23(2a) as the area of the total geographic range that includes all extant populations of the species. Its application in Clause 7 follows criterion B1 in IUCN (2001). Extent of occurrence can often be measured by a minimum convex polygon or convex hull (the smallest polygon in which no internal angle exceeds 180 degrees and which contains all the sites of occurrence). IUCN (2001) states that EOO may exclude discontinuities or disjunctions within the overall distributions of taxa (e.g. large areas of obviously unsuitable habitat). However, the consequences of excluding discontinuities vary, depending on whether the estimate of EOO is to be used for assessing the total distribution in Clause 7, or whether it is to be used for estimating or inferring reductions (Clause 6) or continuing declines (Subclauses 7d and 8d in conjunction with Subclause 4b). Box 4 summarises guidance from IUCN (2014) on how to estimate EOO under these different criteria.

***Box 4. Estimating Extent of Occurrence*.** The following considerations apply to EOO as a measure of geographic distribution size in Clauses 7 and 18 of the TSC Regulation 2010. In relation to criterion B IUCN (2014) states that "exclusion of areas forming discontinuities or disjunctions from estimates of EOO is discouraged except in extreme circumstances because disjunctions and outlying occurrences accurately reflect the extent to which a large range size reduces the chance that the entire population of the taxon will be affected by a single threatening process. The risks are spread by the existence of outlying or disjunct occurrences irrespective of whether the EOO encompasses significant areas of unsuitable habitat (IUCN 2014). Inappropriate exclusions of discontinuities or disjunctions within the overall distribution of a taxon will underestimate EOO and consequently will underestimate the degree to which risk is spread spatially for the taxon" (IUCN 2014).
The following considerations apply to changes in EOO as a measure of reduction of population size in Clause 6, reduction of geographic distribution in Clause 17, and as an indicator of continuing decline in geographic distribution size in Subclauses 7d and 18d in conjunction with Subclause 4b of the TSC Regulation 2010.

Effects of outlying occurrences on estimates of EOO based on minimum convex polygons (also known as convex hulls) and their sensitivity to sampling effort makes them less suitable as a method for comparing two or more temporal estimates of EOO for assessing reductions or continuing declines. If outliers are detected at one time and not another, this could result in erroneous inferences about reductions or increases. Therefore a method such as the α-hull (a generalisation of a convex hull) is recommended for assessing reductions of continuing declines in EOO because it substantially reduces the biases that may result from the spatial arrangement of habitat (Burgman & Fox 2003). The α-hull provides a more repeatable description of the external shape of a species' range by breaking it into several discrete patches when it spans uninhabited regions. Simulations show that the estimate of area and trend in area converges on the correct value as sample size increases unless other errors are large. Kernel estimators may be used for the same purpose but their application is more complex. IUCN (2014) and Burgman & Fox (2003) provide guidance on the calculation of α-hulls.

In the case of migratory species, EOO should be based on the minimum of the breeding or non-breeding (wintering) areas, but not both, because such species are dependent on both areas, and the bulk of the population is found in only one of these areas at any time.

To ensure consistency with the definition of Area of Occupancy (AOO), if EOO is less than AOO, EOO should be changed to make it equal to AOO.

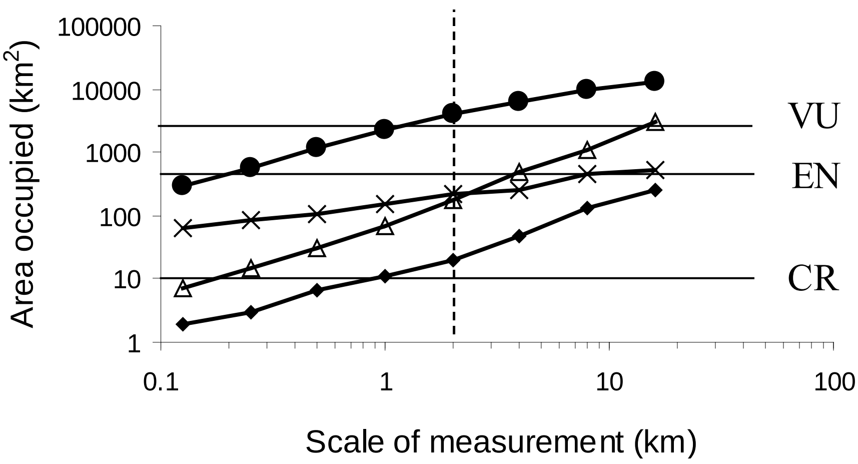
(b) **Area of occupancy** (AOO) is defined in Clause 23(2b) as the area within the total range (and hence within EOO) that is currently occupied by the species. It. excludes unsuitable and unoccupied habitat. Its application in Clause 7 follows criterion B2 in IUCN (2001). In some cases, (e.g. irreplaceable colonial nesting sites, crucial feeding sites for migratory taxa) the area of occupancy is the smallest area essential at any stage to the survival of existing populations of a taxon (IUCN 2001). IUCN (2014) explains the rationale underpinning AOO as follows: "Suppose two species have the same EOO, but different values for AOO, perhaps because one has more specialised habitat requirements. For example, two species may be distributed across the same desert (hence EOO is the same), but one is wide ranging throughout (large AOO) while the other is restricted to oases (small AOO). The species with the smaller AOO may have a higher risk of extinction because threats to its restricted habitat (e.g. degradation of oases) are likely to reduce its habitat more rapidly to an area that cannot support a viable population. The species with the smaller AOO is also likely to have a smaller population size than the one with a larger AOO, and hence is likely to have higher extinction risks for that reason" (IUCN 2014). Estimates of AOO are highly sensitive to scale of measurement. The scale-dependence of AOO and recommendations for a method of estimation are discussed in Box 5.

Box 5. Estimating Area of Occupancy (after IUCN 2014 sections 4.10.1 – 4.10.3)

Both IUCN (2001) and TSC Regulation 2010 (Clause 23(3)) acknowledge that AOO should be estimated at a scale appropriate to the biology of the species, nature of threats and available data. IUCN (2014) recommends that this be done by using a standard scale based on 2 x 2 km grid cells, unless an alternative scale is justified.

Problems of scale

Classifications based on the area of occupancy (AOO) may be complicated by problems of spatial scale. There is a logical conflict between having fixed range thresholds and the necessity of measuring range at different scales for different taxa. “The finer the scale at which the distributions or habitats of taxa are mapped, the smaller the area will be that they are found to occupy, and the less likely it will be that range estimates … exceed the thresholds specified in the criteria. Mapping at finer spatial scales reveals more areas in which the taxon is unrecorded. Conversely, coarse-scale mapping reveals fewer unoccupied areas, resulting in range estimates that are more likely to exceed the thresholds for the threatened categories. The choice of scale at which AOO is estimated may thus, itself, influence the outcome of Red List assessments and could be a source of inconsistency and bias.” (IUCN 2001). These effects are illustrated in the graph below (from Keith 2009 and Nicholson *et al.* 2009), which shows how the scale at which AOO is measured may influence whether or not different species meet the AOO thresholds for different categories of threat. The broken vertical line shows the standard scale recommended for assessment of AOO by IUCN (2014). At this scale, three of the species are within the AOO threshold for Endangered and one is outside the AOO thresholds for all three categories of threat.



To reduce scale-related bias, some estimates of AOO may require standardisation to an appropriate reference scale. Below, a simple method of estimating AOO is described, an appropriate reference scale is recommended, and a method of standardisation is described for cases where the available data are not at the reference scale.

Methods for estimating AOO

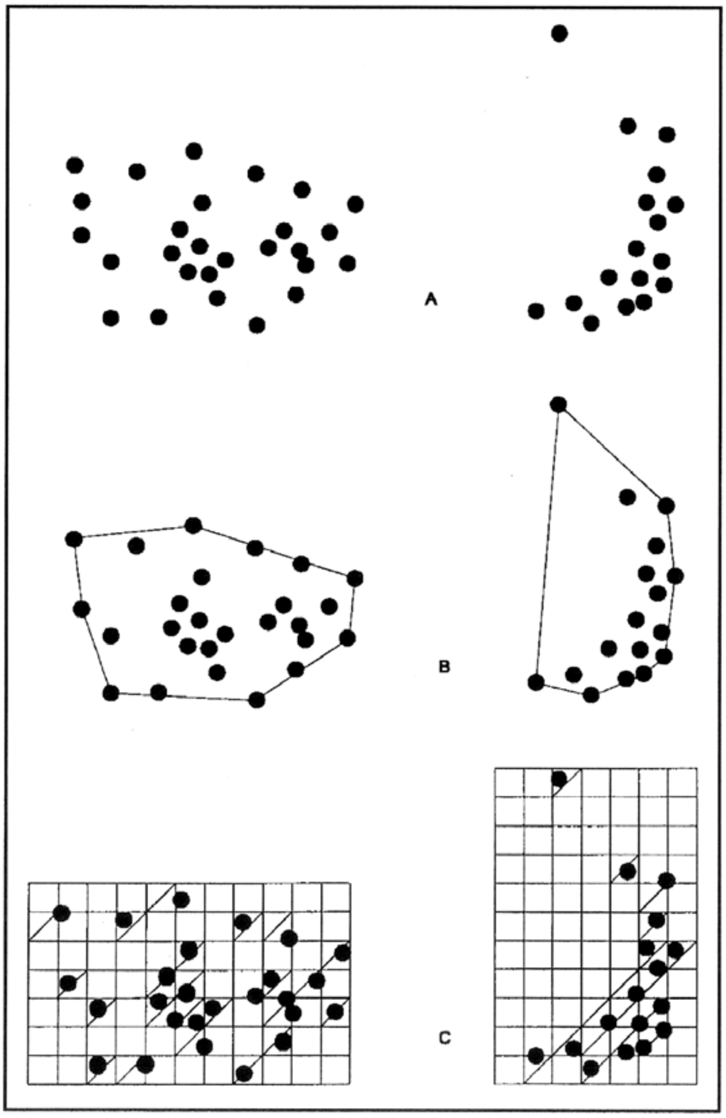
There are several ways of estimating AOO, but for the purpose of these guidelines we assume estimates have been obtained by counting the number of occupied cells in a uniform grid that covers the entire range of a species (see Figure 2.2.2), and then tallying the total area of all occupied cells:

AOO = no. occupied cells × area of an individual cell (equation 5.1)

The 'scale' of AOO estimates can then be represented by the area of an individual cell in the grid (or alternatively the length of a cell). There are other ways of representing AOO, for example, by mapping and calculating the area of polygons that contain all occupied habitat. The scale of such estimates may be represented by the area of the smallest mapped polygon (or the length of the shortest polygon segment), but these alternatives are not recommended. If different grid locations or origins (reference points of the grid) result in different AOO estimates, the minimum estimate should be used.

Figure 2.2.2. Two examples of the distinction between extent of occurrence and area of occupancy.
(A) is the spatial distribution of known, inferred or projected sites of present occurrence.
(B) shows one possible boundary to the extent of occurrence, which is the measured area within this boundary.
(C) shows one measure of area of occupancy which can be achieved by the sum of the occupied grid squares.

(taken from IUCN (2001)



The appropriate scale

It is impossible to provide any strict but general rules for mapping taxa or habitats; the most appropriate scale will depend on the taxon in question, and the origin and comprehensiveness of the distribution data. However, in many cases a grid size of 2 km (a cell area of 4 km^2) is an appropriate scale. Scales of 3.2 km grid size or coarser (larger) are inappropriate because they do not allow any taxa to be listed as Critically Endangered (CR); even species that occur within a single grid will have an area that exceeds 10 km^2, the AOO threshold for Critically Endangered under criterion B of IUCN (2001). Scales of 1 km grid size or smaller tend to list more taxa at higher threat categories than these categories imply. For most cases, a scale of 4 km^2 cells as the reference scale is recommended by (IUCN 2014). If an estimate was made at a different scale, especially if data at different scales were used in assessing species in the same taxonomic group, this may result in inconsistencies and bias. In any case, the scale for AOO should not be based on EOO (or other measures of range area), because AOO and EOO measure different factors affecting extinction risk.

Estimates of AOO may be standardized by applying a scale-correction factor. Scale-area relationships (e.g,. Figure. 4.3 of IUCN (2014)) provide important guidance for such standardization. It is not possible to give a single scale-correction factor that is suitable for all cases because different taxa have different scale-area relationships. Furthermore, a suitable correction factor needs to take into account a reference scale (e.g. 2 km grid size) that is appropriate to the area of occupancy thresholds in criterion B. The example below shows how estimates of AOO made at fine and coarse scales may be scaled up and down, respectively, to the reference scale to obtain an estimate that may be assessed against the AOO thresholds in Criterion B.

Standardising the scale of an AOO estimate

Where the available data are at inappropriate scales for estimating AOO (e.g. point ocurrences or large polygons), it will be necessary to obtain an estimate at the reference scale represented by a 2 km grid. This may done cartographically or in a Geographic Information System by simply intersecting the mapped occurrences with a 2 km grid and then summing the area of occupied grid cells by applying equation 5.1. This method is suitable when the occurrence data are available at a relatively fine scale and need to be scaled up to a 2 x 2 km grid. This will be the case for most species and ecological communities to be assessed in New South Wales, where data are available at finer resolution than for many other parts of the world. In other cases, it may be necessary to calculate a scale-area curve, as shown in the above graph, and interpolate or extrapolate an estimate of AOO at the reference scale. This can be done mathematically by calculating a scale correction factor (C) from the slope of a scale-area curve as follows:

$$C = (\log_{10}(AOO_2/AOO_1)/\log_{10}(Ag_2/Ag_1)) \qquad \text{(equation 5.2)}$$

Where AOO_1 is the estimated area occupied from grids of area Ag_1, a size close to, but smaller than the reference scale, and AOO_2 is the estimated area occupied from grids of area Ag_2, a size close to, but larger than the reference scale. An estimate of AOO_R at the reference scale, Ag_R, may thus be calculated by rearranging equation 5.2 as follows:

$$AOO_R = AOO_1 * 10^{C*\log(Ag_R / Ag_1)} \qquad \text{(equation 5.3)}$$

Worked examples of calculations are given in section 4.10.5 of IUCN (2014).

(c) Area of suitable habitat is defined in Clause 23(2c) as ‘the area within the total range that includes occupied and unoccupied suitable habitat, but excludes unsuitable habitat.’ Maps of suitable habitat may be derived from interpretation of remote imagery and/or analyses of spatial environmental data using simple combinations of GIS data layers, or by more formal statistical habitat models (e.g. generalised linear and additive models, decision trees, Bayesian models, regression trees, etc.). Habitat maps can provide a basis for estimating AOO and EOO and, if maps are available for different points in time, rates of change can be estimated (IUCN 2014). They cannot be used directly to estimate a taxon’s AOO because they often map an area that is larger than the occupied habitat (i.e. they also map areas of suitable habitat that may presently be unoccupied). However, they may be a useful means of estimating AOO indirectly, for which IUCN stipulates three conditions that must be met.

i) Maps must be justified as accurate representations of the habitat requirements of the species and validated by a means that is independent of the data used to construct them.
ii) The mapped area of *suitable* habitat must be interpreted to produce an estimate of the area of *occupied* habitat.
iii) The estimated area of occupied habitat derived from the map must be scaled to the grid size that is appropriate for AOO of the species.

Habitat maps can vary widely in quality and accuracy (condition i). A map may not be an accurate representation of habitat if key variables are omitted from the underlying model. For example, a map would over-estimate the habitat of a forest-dependent montane species if it identified all forest areas as suitable habitat, irrespective of altitude. The spatial resolution of habitat resources also affects how well maps can represent suitable habitat. For example, specialised nest sites for birds, such as a particular configuration of undergrowth or trees with hollows of a particular size, do not lend themselves to mapping at coarse scales. Application of habitat maps to the assessment of species for listing under the TSC Act, should therefore be subject to an appraisal of mapping limitations, which should lead to an understanding of whether the maps over-estimate or under-estimate the area of suitable habitat.

Habitat maps may accurately reflect the distribution of suitable habitat, but only a fraction of suitable habitat may be occupied (condition ii). Therefore the area of suitable habitat may be an upper bound of the possible AOO although, depending on the proportion of suitable habitat actually occupied, it could be substantially larger than any plausible upper bound of AOO. Low habitat occupancy may result because other factors are limiting – such as availability of prey, impacts of predators, competitors or disturbance, dispersal limitations, etc. In such cases, the area of mapped habitat could be substantially larger than AOO and will therefore need to be adjusted (using an estimate of the proportion of habitat occupied) to produce a valid estimate of AOO. This may be done by random sampling of suitable habitat grid cells, which would require multiple iterations to obtain a stable mean value of AOO (IUCN 2014).

Habitat maps are produced at a resolution determined by the input data layers (satellite images, digital elevation models, climate surfaces, etc.). Often these will be at finer scales than those required to estimate AOO (condition iii), and consequently scaling up will be required (see Box 5).

In those cases where AOO is less than the area of suitable habitat, the population may be declining within the habitat, but the habitat may show no indication of change (Rodriguez 2002; IUCN 2014). Hence estimates of population reduction (under Clause 6) could be both inaccurate and non-precautionary.

However, if a decline in mapped habitat area is observed (and the map is a reasonable representation of suitable habitat – condition i), then the population is likely to be declining at least at that rate. This is a robust generalisation because even the loss of unoccupied habitat can reduce population viability (Levins 1970; Hanski & Gilpin 1997; Beissinger & McCulloch 2002). Thus, if estimates of AOO are not available, then the observed decline in mapped habitat area can be used to invoke "continuing decline" in Subclauses 7d and 8d, and the rate of such decline can be used as a basis for calculating a lower bound for population reduction under Clause 6.

2.2.3 Size of geographic distribution

To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, a species must have a **geographic distribution** that is estimated or inferred* to be **very highly restricted** (Subclause 7a)**, highly restricted** (Subclause 7b) **or moderately restricted** (Subclause 7c), in addition to meeting other particular conditions (Subclauses 7d or 7e). The corresponding listing criteria in IUCN (2001) provide indicative guidance for quantitative interpretation of these terms. (Table 2).

Table 2. Corresponding thresholds for size of geographic distribution size for the TSC Regulation 2010 and the IUCN (2001) Red List criteria .

Category of threat	Requirement under Clause 7 of TSC Regulation 2010	Thresholds for Extent of Occurrence under criterion B1 of IUCN (2001)	Thresholds for Area of Occupancy under criterion B2 of IUCN (2001)
Critically Endangered	very highly restricted	≤100 km^2	≤10 km^2
Endangered	highly restricted	≤5000 km^2	≤500 km^2
Vulnerable	moderately restricted	≤20000 km^2	≤2000 km^2

2.3 Clauses 8 & 9 – number of mature individuals

Clauses 8 and 9 are based on IUCN (2001) criteria C and D. The basis for Clauses 8 and 9 is the small population paradigm (Caughley 1994): a small population is more likely to become extinct than a large one (see also Mace & Lande 1991, Keith 1998, Mace *et al.* 2008, IUCN 2014). For the purposes of assessing Clauses 8 and 9, the sizes of species' populations are assessed by estimating or inferring the number of mature individuals.

2.3.1 Mature individuals

Clause 22(1) (after IUCN 2001) defines **mature individuals** as 'individuals in the wild known, estimated or inferred to be capable of producing viable offspring'. The total number of mature individuals excludes individuals that are too young (juvenile), too old (senescent), too moribund (for example, diseased) or otherwise unable to produce viable offspring (for example, due to low population density).

Clause 22(2) (after IUCN 2001) provides guidance on the interpretation of mature individuals in the following special cases:

(a) In populations with biased sex ratios, it is appropriate to use a lower value for the total number of mature individuals in a way that takes this into account.
(b) In populations that fluctuate (see Section 2.4.2.4), the number of mature individuals will refer to a minimum number of individuals that are present

* See Box 2 for definitions of 'estimated' and 'inferred'

most of the time (in a time span appropriate to the life cycle and habitat characteristics of the species), and will thus usually be much less than the mean number present.

(c) In clonal organisms, reproducing units may be regarded as mature individuals, so long as they survive independently of one another. However, if clonally reproduced individuals are more limited in viability or dispersal ability than sexually reproduced individuals, the total number of mature individuals may be reduced accordingly to take this into account.

(d) For species in which individuals have synchronous dormant life stages, the number of mature individuals should be assessed during, or projected for, a time when mature individuals are available for breeding.

(e) Re-introduced individuals must have produced viable offspring (after the individuals were re-introduced) before they are counted as mature individuals.

(f) Captive, cultivated, or artificially maintained individuals cannot be counted as mature individuals.

2.3.2 Number of mature individuals

To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, under Clause 8, the estimated* total number of mature individuals of a species must be **very low** (Subclause 8a)**, low** (Subclause 8b) **or moderately low** (Subclause 8c) in addition to meeting other particular conditions (either subclause 8d OR 8e, in conjunction with Clause 4). To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, under Clause 9, the estimated* total number of mature individuals of a species must be **extremely low** (Subclause 9a)**, very low** (Subclause 9b) **or low** (Subclause 9c) and no additional conditions are required. The corresponding listing criteria in IUCN (2001) provide indicative guidance for quantitative interpretation of these terms (Table 3).

Table 3. Corresponding thresholds for number of mature individuals for the TSC Regulation 2010 and the IUCN (2001) Red List criteria.

Category of threat	Requirement under Clause 8 of TSC Regulation 2010	Thresholds for number of mature individuals under criterion C of IUCN (2001)	Requirement under Clause 9 of TSC Regulation 2010	Thresholds for number of mature individuals under criterion D of IUCN (2001)
Critically Endangered	very low	Fewer than 250 mature individuals	extremely low	Fewer than 50 mature individuals
Endangered	low	Fewer than 2500 mature individuals	very low	Fewer than 250 mature individuals
Vulnerable	moderately low	Fewer than 10,000 mature individuals	low	Fewer than 1000 mature individuals

* See Box 2 for definitions of ‘estimated’ and ‘inferred’

2.4 Subclauses 7d-e & 8d-e – continuing decline, fragmentation, concentration and fluctuation

To be eligible for listing under Clause 7, a species must have a geographic distribution that is estimated or inferred to be restricted to particular degrees (Subclauses 7a-c). Similarly, eligibility for listing under Clause 8, requires a very low, low or moderately low number of mature individuals. However, for both clauses, a species must meet at least one of two further requirements (Subclauses 7d OR 7e in conjunction with Clause 4, and Subclauses 8d OR 8e in conjunction with Clause 4, respectively) in order to be eligible for listing. The first of these (Subclauses 7d and 8d) refers to a projected or continuing decline in specified species parameters (section 2.4.1). The second (Subclauses 7e and 8e) refers to a combination of conditions including severe fragmentation, the number of populations or locations of the species, and extreme fluctuations (section 2.4.2).

2.4.1 <u>Projected or continuing decline</u>

To meet Subclauses 7d and 8d, a **projected or continuing decline** must be observed, estimated or inferred* in the key indicators defined in Clause 4:

(a) an index of abundance appropriate to the taxon (see section 2.1.2), or
(b) geographic distribution (see section 2.2.2), habitat quality or diversity (see section 2.1.2), or genetic diversity (see section 2.1.2).

"A continuing decline is a **recent, current or projected future decline** (which may be smooth, irregular or sporadic) which is **liable to continue unless remedial measures are taken**. Fluctuations (see section 2.4.2.4) will not normally count as continuing declines, but an observed decline should not be considered as a fluctuation unless there is evidence for this." (IUCN 2001)

Note that a continuing decline is not possible without a 'reduction' (which must be assessed under Clause 6), but a reduction is possible without a continuing decline: if a reduction has 'ceased' (Clause 6), there cannot be a continuing decline.

To invoke a continuing decline under Subclause 7d, and distinguish it from a fluctuation, a downward trend in a population size must be liable to continue and be non-trivial in magnitude relative to the total population of the species. For example, a decline in one population of a species or in part of a species' range would not be evidence of a 'continuing decline' in the total species population if it was compensated by increases in other populations or other parts of the species' range. Similarly, loss of a few individuals from a total population that contains millions of individuals could not easily be distinguished from fluctuations or sampling errors, and therefore constitutes weak evidence of a continuing decline. Interpreting whether population changes are non-trivial requires an overall assessment of the total population analogous to calculations of population reductions shown in Box 3. In other words, continuing declines should be interpreted from overall trends determined by proportionately weighted consideration of trends in individual populations based on their relative size. This consideration need not be quantitative, as required for reductions (Box 3). However, where observations or estimates are unavailable on trends in large populations, inferences drawn about such trends must be justified, for

* See Box 2 for definitions of 'observed', 'estimated' and 'inferred'

example, by considering whether those populations are likely to be affected by similar threats to those for which trends are known.

2.4.2 Fragmentation, concentration and fluctuation

To be eligible for listing under Subclauses 7e and 8e, at least two of the following three conditions must apply:

(i) the population or habitat is observed or inferred* to be **severely fragmented**,

(ii) all or nearly all mature individuals are observed or inferred* to occur within a small number of **populations** or **locations**,

(iii) **extreme fluctuations** are observed or inferred* to occur in the key indicators defined in Clause 4:

(a) an index of abundance appropriate to the taxon, or

(b) geographic distribution, habitat quality or diversity, or genetic diversity.

The definitions of terms and concepts associated with these conditions are discussed in the following sections.

2.4.2.1 Severe fragmentation

Clause 24 defines severe fragmentation as follows: "The population or habitat of a species is severely fragmented if individuals of the species are distributed among subpopulations or patches of habitat that are small and isolated relative to the life cycle and habitat characteristics of the species." Species with severely fragmented populations or habitat are exposed to greater risks of extinction than other species because their small populations may go extinct, with a reduced probability of recolonisation (IUCN 2001). Furthermore, the reduced movement of individuals between populations or occupied habitat patches reduces the likelihood that declining populations will be rescued by migration from other patches (Levins 1969; Gonzalez *et al.* 1998).

Fragmentation must be assessed at a scale that is appropriate to biological isolation in the taxon under consideration. In general, taxa with highly mobile adult life stages or with a large production of small mobile diaspores are considered more widely dispersed, and hence not so vulnerable to isolation through fragmentation of their habitats. Taxa that produce only small numbers of diaspores (or none at all), or only large ones, are less efficient at long distance dispersal and therefore more easily isolated.

In certain circumstances severe fragmentation may be inferred from habitat information (IUCN 2001), for example, where there is evidence of trends indicating that habitat patches are becoming smaller, more numerous and more isolated from one another. If natural habitats have been fragmented (e.g. old growth stands within timber production forests, woodland remnants in agricultural landscapes, etc.), this can be used as direct evidence for fragmentation for taxa with poor dispersal ability.

* See Box 2 for definitions of 'observed', 'estimated' and 'inferred'
* See Box 2 for definitions of 'observed', 'estimated' and 'inferred'
* See Box 2 for definitions of 'observed', 'estimated' and 'inferred'

In cases where data are available on the size of individual populations, the corresponding listing criteria in IUCN (2001) provide indicative guidance for quantitative interpretation of ‘small populations’ when assessing whether a species population is severely fragmented. (Table 4). Even where these indicative thresholds are met, the assessor still needs to determine whether the populations are isolated.

Table 4. Corresponding thresholds for assessing severe fragmentation of species populations under the TSC Regulation 2010 and the IUCN (2001) Red List criteria.

Category of threat	Requirement under Clauses 7ei , 8ei & 24 of TSC Regulation 2010	Thresholds for Extent of Occurrence under criterion C2a(i) of IUCN (2001)
Critically Endangered	Populations small & isolated	No population estimated to contain more than 50 mature individuals
Endangered	Populations small & isolated	No population estimated to contain more than 250 mature individuals
Vulnerable	Populations small & isolated	No population estimated to contain more than 1000 mature individuals

In cases where data are available on (i) the spatial distribution of occupied habitat, (ii) some aspect of the dispersal ability of the taxon (e.g. average dispersal distance), and (iii) average population density in occupied habitat (e.g. information on territory size, home range size, etc.), IUCN (2014) proposes that species can be considered to be severely fragmented if most (>50%) of its total area of occupancy is in habitat patches that are (1) smaller than would be required to support a ‘viable’ population, and (2) separated from other habitat patches by a large distance relative to the dispersal capabilities of the species (IUCN 2014). **Note that the existence (or even a large number) of small habitat patches, of itself, is insufficient evidence of severe fragmentation.**

For (1), the area for a viable population should be based on rudimentary estimates of population density, and on the ecology of the taxon (IUCN 2014). For example, for many vertebrates, patches that can support fewer than a hundred individuals may be considered likely to be smaller than ‘viable’ size. For (2), the degree of isolation of patches should be based on dispersal distance of the taxon (IUCN 2014). For example, patches that are isolated by distances several times greater than the (long-term) average dispersal distance of the taxon may be considered isolated.

For many taxa, the information on population density and dispersal distance may be inferred from other similar taxa. Biologically informed values can be set by the assessors for large taxonomic groups (families or even orders), or for other groupings of taxa based on their biology. For example, in bryophytes information on the effects of isolation of subpopulations is often lacking. It is recommended that in most circumstances, a minimum distance greater than 50km between subpopulations of taxa without spore dispersal can indicate severe fragmentation, and a distance of between 100km and 1,000km for taxa with spores (Hallingbäck *et al.* 2000).

The definition of severe fragmentation is based on the distribution of populations. This is often confused with the concept of "location" (see section 2.4.2.3), but is independent of it. A taxon may be severely fragmented, yet all the isolated populations may be threatened by the same major factor (single location), or each population may be threatened by a different factor (many locations).

2.4.2.2 Number of populations

In the TSC Regulation 2010, usage of the term 'population' follows IUCN (2001) where a 'population' is defined as the total number of individuals of a taxon. This differs from common biological usage of the term (e.g. Begon *et al.* 2006). Instead, IUCN (2001) applies the term 'subpopulations' to the common biological meaning of 'population'. The TSC Regulation 2010 (listing criteria) follows the common biological usage of 'populations'. Based on IUCN's (2001) definition of subpopulations, populations are here defined as 'geographically or otherwise distinct groups of individuals within the same species, between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less).' Although populations typically have little demographic or genetic exchange, this may or may not amount to their complete isolation.

Operational methods for determining the number of populations may vary from species to species. In tree species, for example, a p opulation can be defined as a spatially distinct occurrence of the species that experiences insignificant seed or pollen migration from other populations within a generation.

Subclauses 7eii and 8eii are invoked for species in which all or nearly all mature individuals are observed or inferred to occur within a small number of **populations.** The corresponding listing criteria in IUCN (2001) provide indicative guidance for interpretation of cases where 'nearly all mature individuals' occur within a 'small number of populations' (Table 4).

2.4.2.3 Number of locations

Following IUCN (2001), "The term 'location' defines a geographically or ecologically distinct area in which a single threatening event can rapidly affect all individuals of the taxon present. The size of the location depends on the area covered by the threatening event and may include part of one or many populations. Where a taxon is affected by more than one threatening event, location should be defined by considering the most serious plausible threat."

Estimates of the number of locations of a species should include reference to the most serious plausible threat(s) (IUCN 2014). F or example, where the most serious plausible threat is habitat loss, a location is an area where a single development project can eliminate or severely reduce the population. W here the most serious plausible threat is volcanic eruption, hurricane, tsunami, frequent flood or fire, locations may be defined by the previous or predicted extent of lava flows, storm paths, inundation, fire paths, etc. Where the most serious plausible threat is collection or harvest, then locations may be defined based on t he size of jurisdictions (within which similar regulations apply) or on t he level of access (e.g. ease with which collectors may reach different areas), as well as on the factors that determine how the levels of exploitation change (e.g. if collection intensity in two separate areas changes in response to the same market trends in demand, these may be counted as a single location).

If two or more populations occur within an area that may be threatened by one such event, they must be counted as a single location. Conversely, if a single population covers an area larger than may be affected by any single event, it must be counted as more than one location (IUCN 2014).

Where the most serious plausible threat does not affect all of the species' distribution, other threats can be used to define and count locations in those areas not affected by the most serious plausible threat (IUCN 2014).

If there are two or more serious plausible threats, the number of locations should be based on the threat that results in the smallest number of locations (IUCN 2014).

When parts of the distribution are not affected by any threat, IUCN (2014) recommends the following options under different circumstances: (a) number of locations is not used in the assessment (i.e. the subcriteria that refer to the number of locations are consequently not met), especially if the unaffected area is more than half the species' range; (b) number of locations in the unaffected areas is set to the number of populations in those areas, especially if there are several populations; (c) the number of locations is based on the smallest size of locations in the currently affected areas; (d) the number of locations is based on the most likely threat that may affect the currently-unaffected areas in the future. In any case, the basis of the number of locations should be documented.

In the absence of any plausible threat for the taxon, the "location" part of Subclauses 7eii and 8eii cannot be invoked.

Subclauses 7eii and 8eii are invoked for species in which all or nearly all mature individuals are observed or inferred to occur within a small number of **locations.** The corresponding listing criteria in IUCN (2001) provide indicative guidance for interpretation of cases where 'nearly all mature individuals' occur within a 'small number of locations' (Table 5).

Table 5. Corresponding thresholds for assessing the number of [sub]populations and locations under the TSC Regulation 2010 and the IUCN (2001) Red List criteria.

Category of threat	Requirement under Subclauses 7eii & 8eii of TSC Regulation 2010	Thresholds for population structure under criterion C2a(ii) of IUCN (2001)	Thresholds for number of locations under criterion B1a of IUCN (2001)
Critically Endangered	All or nearly all mature individuals within a small number of populations or locations	At least 90% of mature individuals in one subpopulation	Known to exist at only a single location
Endangered	All or nearly all mature individuals within a small number of populations or locations	At least 95% of mature individuals in one subpopulation	Known to exist at no more than five locations
Vulnerable	All or nearly all mature individuals within a small number of	All mature individuals in one subpopulation	Known to exist at no more than ten locations

2.4.2.4 Extreme fluctuations

Clause 25 of the TSC Regulation 2010 states that "extreme fluctuations occur when the population or distribution of a species varies reversibly, widely and frequently, as:

(i) indicated by changes in either of the key indicators (Clause 4):
  (a) an index of abundance appropriate to the taxon, or
  (b) geographic distribution, habitat quality or diversity, or genetic diversity
(ii) inferred from the life history or habitat biology of the species.

The cause of fluctuations must be understood or inferred so that they may be distinguished from declines or reductions."

Extreme fluctuations are included in Subclauses 7eiii and 8eiii in recognition of the positive relationship between extinction risk and variance in the rate of population growth (Burgman *et al.* 1993). Populations that undergo extreme fluctuations are likely to have highly variable growth rates, and are therefore likely to be exposed to higher extinction risks than populations with lower levels of variability. The effect of extreme fluctuations on the extinction risk will depend on both the degree of isolation and the degree of synchrony of the fluctuations between populations (IUCN 2014).

Population fluctuations may vary in magnitude and frequency (IUCN 2014, Figure 4.1). **For the 'extreme fluctuations' subclauses to be invoked, populations would normally need to fluctuate by at least 10-fold** (i.e. an order of magnitude difference between population minima and maxima). Fluctuations may occur over any time span, depending on their underlying causes (IUCN 2014). Short-term fluctuations that occur over seasonal or annual cycles will generally be easier to detect than those that occur over longer time spans, such as those driven by rare events or climatic cycles such as El Niño. Fluctuations may occur regularly or sporadically (i.e. with variable intervals between successive population minima or successive population maxima).

If there is regular or occasional dispersal (of even a small number of individuals, seeds, spores, etc) between all (or nearly all) of the populations, then the degree of fluctuations should be measured over the entire population (IUCN 2014). In this case, Subclauses 7eiii and 8eiii would be met only when the overall degree of fluctuation (in the total population size) is larger than one order of magnitude. If the fluctuations of different populations are independent and asynchronous, they would cancel each other to some extent when fluctuations of the total population size are considered.

If, on the other hand, the populations are totally isolated, the degree of synchrony between the population is not as important (IUCN 2014) and it is sufficient that a majority of populations each show extreme fluctuation to meet Subclauses 7eiii and 8eiii. In this case, if most of the populations show fluctuations of an order of magnitude, then the subclauses would be met (regardless of the degree of the fluctuations in total population size).

Between these two extremes, if dispersal is only between some of the populations, then the total population size over these connected populations should be considered when assessing fluctuations; each set of connected populations should be considered separately (IUCN 2014).

Population fluctuations may be difficult to distinguish from directional population changes, such as continuing declines, reductions or increases. Figure 4.1 s hows examples where fluctuations occur independent of, and in combination with, directional changes. A reduction should not be interpreted as part of a fluctuation unless there is good evidence for this. Fluctuations must be inferred only where there is reasonable certainty that a population change will be followed by a change in the reverse direction within a generation or two. In contrast, directional changes will not necessarily be followed by a change in the reverse direction.

There are two main ways that extreme fluctuations may be diagnosed (IUCN 2014): (i) by interpreting population trajectories based on an index of abundance appropriate for the taxon; and (ii) by using life history characteristics or habitat biology of the taxon.

i) Population trajectories must show a r ecurring pattern of increases and decreases (Figure 4.1). Normally, several successive increases and decreases would need to be observed to demonstrate the reversible nature of population changes, unless an interpretation of the data was supported by an understanding of the underlying cause of the fluctuation (see ii). Successive maxima or minima may be separated by intervals of relatively stable population size.

ii) Some organisms have life histories prone to boom/bust dynamics. Examples include fish that live in intermittent streams, granivorous small mammals of arid climates, and plants that respond to stand-replacing disturbances. In these cases there is dependence on a p articular resource that fluctuates in availability, or a response to a disturbance regime that involves predictable episodes of mortality and recruitment. An understanding of such relationships for any given taxon may be gained from studies of functionally similar taxa, and inference of extreme fluctuations need not require direct observation of successive increases and decreases.

In all cases, assessors must be reasonably certain that fluctuations in the number of mature individuals represent changes in the total population, rather than simply a flux of individuals between different life stages. For example, in some freshwater invertebrates of intermittent water bodies, the number of mature individuals increases after inundation which stimulates emergence from larval stages. Mature individuals reproduce while conditions remain suitable, but die out as the water body dries, leaving behind immature life stages (e.g. eggs) until the next inundation occurs. Similarly, fires may stimulate mass recruitment from large persistent seed banks when there were few mature individuals before the event. As in the previous example, mature plants may die out during the interval between fires, leaving a store of immature individuals (seeds) until they are stimulated to germinate by the next fire. <u>Such cases do not fall within the definition of extreme fluctuations unless the dormant life stages are exhaustible by a single event or cannot persist without mature individuals</u>. Plant taxa that were killed by fire and had an exhaustible canopy-stored seed bank (serotinous obligate seeders), for example, would therefore be prone to extreme fluctuations because the decline in the number of mature individuals represents a decline in the total number.

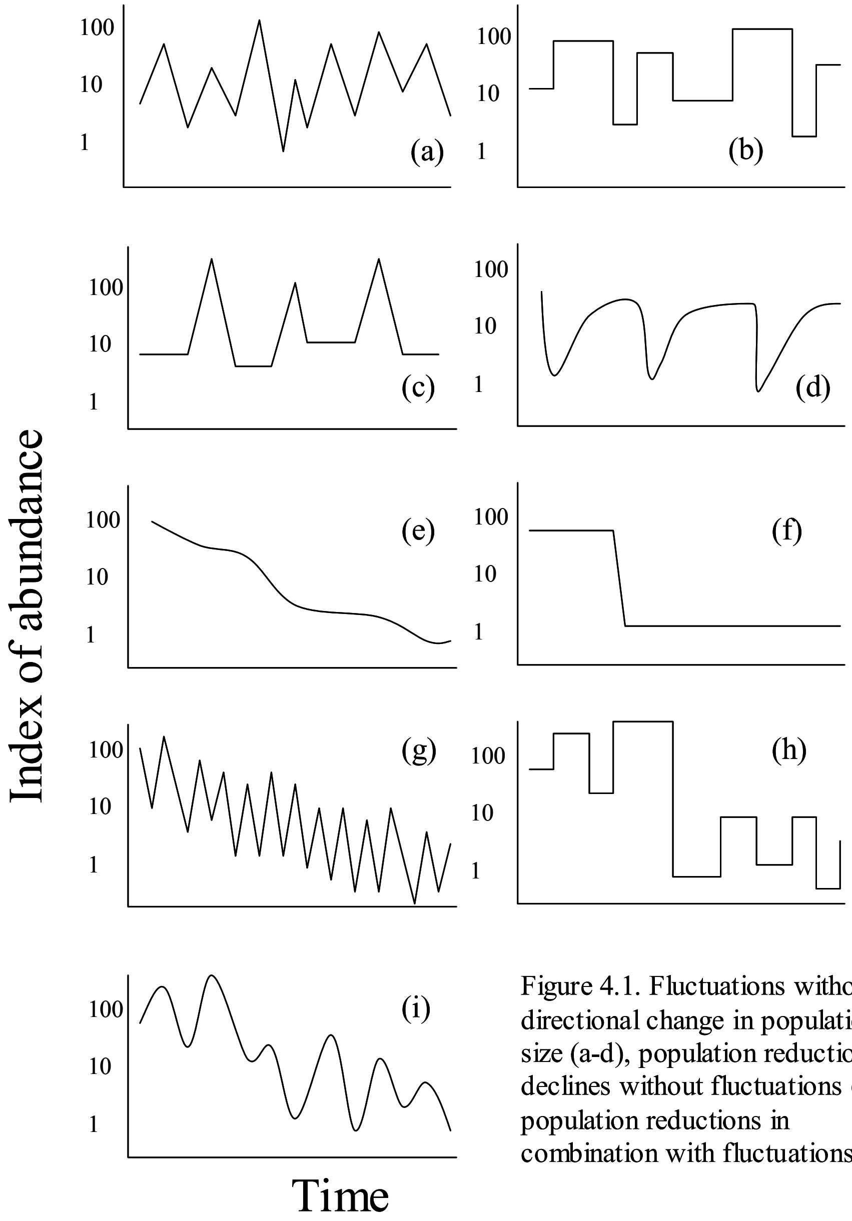


Figure 4.1. Fluctuations without directional change in population size (a-d), population reductions or declines without fluctuations (e-f), population reductions in combination with fluctuations (g-i).

2.5 *Clause 10 – very highly restricted geographic distribution*

Under Clause 10, a species may be eligible for listing as Vulnerable if "the geographic distribution of the species is observed, estimated or inferred to be very highly restricted such that it is prone to the effects of human activities or stochastic events within a very short time period."

Clause 10 is based on IUCN (2001) criterion D2 and the small population paradigm (Caughley 1994). The very highly restricted geographic distribution under Clause 10 is defined such that the population is prone to the effects of human activities or stochastic events within a very short time period in an uncertain future, and is thus capable of becoming Endangered, Critically Endangered or even Extinct in a very short time period (e.g., within one or two generations after the threatening event occurs, IUCN 2014). The numerical thresholds of area of occupancy (typically less than 20 km^2) and number of locations (typically less than 5) given by (IUCN 2001) are indicative examples and are not intended to be interpreted as strict thresholds.

Unlike Clause 7, Clause 10 has no a dditional requirements relating to continuing decline, fragmentation, population concentration or fluctuation. However, the focus of Clause 10 is not only on the area threshold or threshold number of locations (for which many taxa could qualify), but the risk that the taxon could suddenly become highly threatened or extinct. S o, <u>simply meeting the indicative (or any other) threshold for AOO or number of locations is not sufficient, of itself, for a taxon to be eligible for listing under Clause 10</u>. Unlikely events (e.g. eruption of an inactive volcano), non-specific events that were not observed in similar species (e.g. an unspecified disease epidemic), or events unlikely to cause extinction (e.g. because the species has survived many hurricanes, or is likely to adapt to global warming, etc.) would not qualify for listing under Clause 10. The threatening processes (stochastic events or human activities) that lead to this listing must be specified in the justification for listing.

3 Assessments of Populations

The TSC Act 1995 de fines 'population' as a group of organisms, all of the same species, occupying a particular area. Listings of Endangered Populations should clearly establish the 'particular area' of the population. This area may be large or small, but will usually be defined by some spatial discontinuity or other discriminating feature in the distribution of the species to help distinguish indivduals that belong to the listed population from those of other populations. Many currently listed populations are defined by the Local Government Area in which they occur. Others are geographic areas, water catchments or suburbs (e.g. Woronora Plateau, Hunter valley, Bateau Bay) defined by a cited source.

The IUCN (2001) Red List criteria assign a different meaning to the term, 'population' which, in that system, refers to the total number of individuals of a taxon. IUCN (2001) defines 'subpopulation' as a geographically or otherwise distinct in the population between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less). 'Subpopulation' as defined by IUCN (2001), is therefore is the analogous term to a 'population', as defined under the TSC Act 1995

Nominations of populations for listing as Endangered under the TSC Act 1995 must be assessed under Clauses 11-15 of the TSC Regulation 2010. A population is eligible for listing if it meets any of the subclauses of Clause 11 AND any one of Clauses 12-15. Guidance for interpreting Clauses 12-15 may be obtained by references to the relevant sections on Clauses 6-10 for assessing species, as these are essentially identical to Clauses 12-15, respectively. Clause 11 includes three alternative eligibility requirements for listing of an Endangered Population:

(a) it is disjunct or near the limit of its geographic range,
(b) it is or is likely to be genetically, morphologically or ecologically distinct,
(c) it is otherwise of significant conservation value.

These are discussed below.

3.1 Disjunct populations or those near the limit of the species' geographic range

'Disjunct' is undefined in the TSC Act 1995. Dictionary definitions refer to a state of disconnectness or disjointedness. Disjunct populations therefore imply a substantial level of demographic and genetic isolation from other individuals of the species (i.e. greater isolation than is required for a group of conspecific organisms to be recognised as a population). With reference to IUCN's (2001) definition of a subpopulation, the level of isolation required for a disjunct population would need to entail less than a 'little demographic or genetic exchange with other populations of the species (typically one successful migrant individual or gamete per year or less)'. The geographic distances associated with this isolation will vary between species, depending on life history, dispersal and breeding behaviour. For listed plant and animal populations, identified disjunctions vary from the order of tens to hundreds of kilometres. In some cases, dispersal barriers such as topographic features, water bodies, roads or other stretches of unsuitable habitat have been identified as factors contributing to the disjunct status of listed populations. Examples include the Greater Glider population in the Bingi-Congo area of the Eurobodalla LGA, and the Koala population in the Pittwater LGA.

Populations near the limit of the species' geographic range are those near the edge of the species distribution. This may refer to any edge (e.g. northern, western, south-eastern, coastal, high-altitude, etc.), but requires consideration of the distribution in other states.

3.2 Genetic, morphological or ecological distinctness

Populations may be genetically distinct if they include unique loci or alleles or unique distinctive combinations of alleles. Identification of such populations will usually require molecular analyses with sufficiently stratified sampling to draw comparative inferences about these properties.

Morphologically distinct populations comprise individuals that share morphological features that set them apart from individuals that belong to other populations of the species. Identification of such populations may emerge from comparative morphometric studies, but may also be based on possession of unique character traits. *Zieria smithii* Jackson at Diggers Head and Riverina population of the Glossy Black-Cockatoo *Calyptorhynchus lathami* (Temminck 1807) are examples of Endangered Populations currently listed because of their morphological distinct characteristics.

Ecologically distinct populations display life history or behavioural traits or habitat relationships that are distinctive or unique. For example, the Black Cypress Pine population on the Woronora Plateau population occurs at a site that receives more than double the mean annual rainfall than other extant parts of the species distribution (Mackenzie & Keith 2009).

3.3 Populations otherwise of significant conservation value

Populations may otherwise be of conservation value for a variety of reasons. Examples (see Table 6) include the largest remaining population of the species (e.g. *Rhizanthella slateri* population in the Great Lakes LGA), the last remaining example of the species within a region (e.g. *Chorizema parviflorum* Benth. population in the Wollongong and Shellharbour local government areas), a relic of antecedent environmental conditions (*Callitris endlicheri* population on the Woronora Plateau), an occupant of a refugial habitat, a population of historical or cultural significance, etc.

Table 6. Corresponding thresholds for assessing the number of [sub]populations and locations under the TSC Regulation 2010 and the IUCN (2001) Red List criteria.

Significant Conservation Value	Example listings
Large population size or high population density relative to other occurrences of the species	Broad-toothed Rat, *Mastacomys fuscus* Thomas, population at Barrington Tops in the Local Government Areas of Gloucester, Scone and Dungog
Remnant population representing flora or fauna of a region largely transformed by human activities	Gosford Wattle *Acacia prominens* Cunn. ex Don in the Hurstville and Kogarah Local Government Areas
	Chorizema parviflorum Benth. in the Wollongong and Shellharbour
	Lespedeza juncea subsp. *sericea* (Thunb.) Steenis in the Wollongong LGA
	Pomaderris prunifolia in the Parramatta, Auburn, Strathfield and Bankstown Local Government Areas
	Tadgell's Bluebell *Wahlenbergia multicaulis* Benth., in the local government areas of Auburn, Bankstown, Strathfield and Canterbury
	Koala, *Phascolarctos cinereus* (Goldfuss 1817) in the Pittwater Local Government Area (LGA)
	Squirrel Glider, *Petaurus norfolcensis* on the Barrenjoey Peninsula, north of Bushrangers Hill
Unique or unusual habitat relative to other occurrences of the species	*Eucalyptus oblonga* at Bateau Bay, Forresters Beach and Tumbi Umbi in the Wyong LGA
Populations at the climatic extremes of their species' range	Black cypress pine *Callitris endlicheri* on the Woronora plateau
Relictual or refugial habitat	Black cypress pine *Callitris endlicheri* on the Woronora plateau
Population of a species that has an important role in ecosystem function	Emu population in the NSW North Coast
Outstanding example of a species with biological features of scientific significance	*Rhizanthella slateri* in the Bulahdelah area
Populations of value for scientific reference and research	North Head population of the Long-nosed Bandicoot, *Perameles nasuta*

Cultural and educational values	Manly Point population of the Little Penguin *Eudyptula minor* North Head population of the Long-nosed Bandicoot, *Perameles nasuta*

4 Assessments of Ecological Communities

Nominations of ecological communities for listing as threatened under the TSC Act 1995 must be assessed under Clauses 17-19 of the TSC Regulation 2010. An ecological community is eligible for listing if it meets any one of these clauses.

4.1 Definition of an ecological community

The TSC Act (section 4(1)) defines and ecological community as 'an assemblage of species occupying a particular area'. This definition closely follows modern scientific texts (e.g. Begon *et al.* 2006) and embodies three requirements (Preston & Adam 2004a):

i) the constituents of a community must be species;
ii) the species need to be brought together into an assemblage; and
iii) the assemblage of species must occupy a particular area.

4.1.1 <u>Constituent species</u>

Section 4(1) of the Act adopts a pragmatic and inclusive definition of a species as including 'any defined sub-species and taxon below subspecies and any recognisable variant of a sub-species or taxon.' The Act does not require such variants to have been formally described, only that they be recognisable. Appropriate taxonomic specialists and institutions should be consulted to determine whether any particular variant of a species or subspecies is recognisable.

Terrestrial ecological communities are often conveniently described by nominating characteristic vascular plant species, as plants are typically the most detectable species of an assemblage. However, this does not preclude descriptions of ecological communities based on other taxonomic groups. For example, ecological communities have been listed under the TSC Act based primarily on descriptions that include birds, lichens, fungi and snails/slugs. Many currently listed ecological communities described primarily by reference to their plant species composition, also mention vertebrate and invertebrate species, but note that these components are usually poorly documented.

4.1.2 <u>Assemblage of species</u>

An assemblage of species involves the bringing together or gathering into a location or locations the identified species (Preston & Adam 2004a). The ecological concept of a community also involves interactions between the constituent species (Whittaker 1975), although this is not explicitly mentioned in the definition in the TSC Act. Interactions between at least some of the species within a community are implicit in their co-occurrence (Keith 2009).

The TSC Act (Section 24(4)) provides for challenges to listings for up to six months after publication of Final Determinations. One potential avenue of challenge concerns whether a particular listing meets the definition of an Ecological Community under the Act. Legal action challenging the efficacy of one listing, heard by the Land and Environment Court with subsequent appeal heard by the Supreme Court of NSW (VAW Kurri Kurri vs NSW Scientific Committee), established important precedents in the interpretation of ecological communities (Preston & Adam 2004a,b). Review of the efficacy of the 'assemblage of species' is therefore an essential requirement for evaluation of all nominations of ecological communities.

The notion of species co-occurrence is central to the existence of an assemblage of species. An important aspect of co-occurrence is the notion that a common, albeit variable, group of species occur within the distribution of a community. Structurally dominant species, those most abundant or with greatest height or biomass, are sometimes used as abbreviated descriptions of assemblages. However, the occurrence of one or two dominant species, of itself, is not evidence of the existence of an ecological community. Moreover, such an approach assumes a discrete model of ecological communities (see Box 6 for graphical illustration) in which all or most species show highly correlated co-occurrence with the dominants. Such models are unlikely to hold true (Box 6, Begon *et al.* 2006, Keith 2009). Hence, 'communities' defined solely on the basis of dominant species may be poor representations of the broader assemblage of species in which those species are 'dominant'. This is because the same assemblage may sometimes be dominated by other species and because the nominated dominants may sometimes be part of other assemblages. For these reasons, the emphasis of description and diagnosis of ecological communities should address <u>overall species composition</u> of the assemblage, rather than occurrence of selected species (dominant or otherwise). Preston & Adam (2004a) further reinforce the importance of overall species composition in the classification of communities,

> *'Given the primary stress in the definition of ecological community in the TSC Act on an assemblage of species, approaches to vegetation description and classification that involve assessment of total floristics are clearly desireable.'*

Box 6. The nature of ecological communities (from Keith 2009)

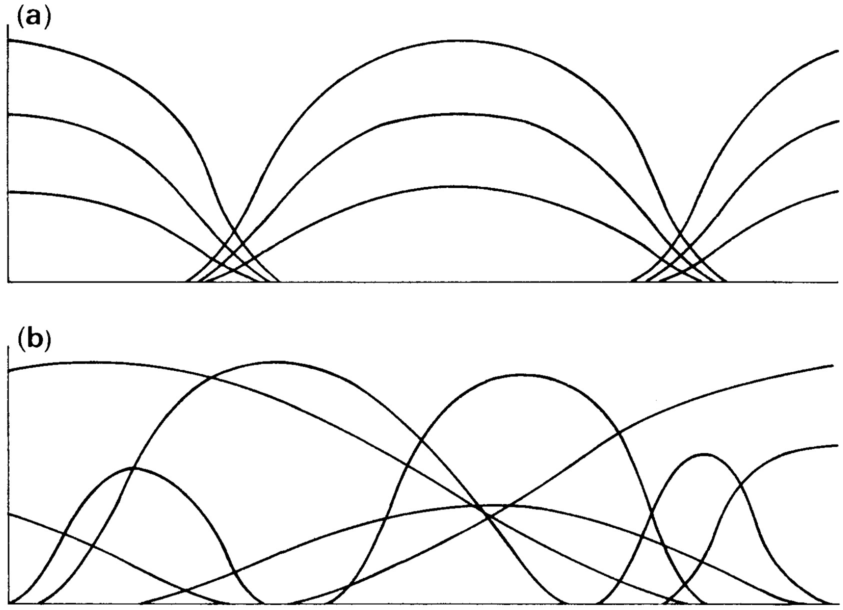


Figure 1 from Keith (2009): Alternative models of ecological communities showing variation in abundance of component species along a simple environmental gradient (after Austin 1985). (a) Organismal discrete model (Clements 1916). (b) Individualistic continuum model (Gleason 1926). The alternative models 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. While spatial scale may influence how overlapping species distributions are perceived, communities are not necessarily any more discrete at larger or smaller scales. See Keith (2009) for further discussion of the issue.

Unlike species, there is no currently accepted typology of ecological communities (Preston & Adam 2004a). Even where such typologies exist or are under construction (e.g. Rodwell *et al.* 2002; Benson 2006), they will inevitably be subject to constraints and limitations associated with available data, methods, biases and knowledge gaps, and therefore may not represent the existence of particular ecological communities. This necessitates the use of a variety of approaches to examine the efficacy of nominated communities. If a given community can be verified by independent methods or several independent means, this provides reasonable evidence of its efficacy. This may also require amalgamation of different types of data in order to develop a workable description of an ecological community (Preston & Adam 2004a).

While not an essential basis for listing ecological communities, numerical analyses of compositional data can often provide stronger evidence for the efficacy of a community than other approaches. This is because the information contributing to a community description is explicit in the input data set, the assumptions and logic of the methods are transparent, and inferences drawn from the output can be justified. Methods that may provide relevant insights include: cluster analysis, ordination, analysis of similarity, fidelity analysis, similarlity components analysis, etc. (Belbin 1993, Clarke & Warwick 1994). The Committee has reviewed a number of such analyses (published and unpublished) to assess the efficacy of nominated communities and has also undertaken meta-analyses to test the efficacy of a number of nominated communities (Table 7).

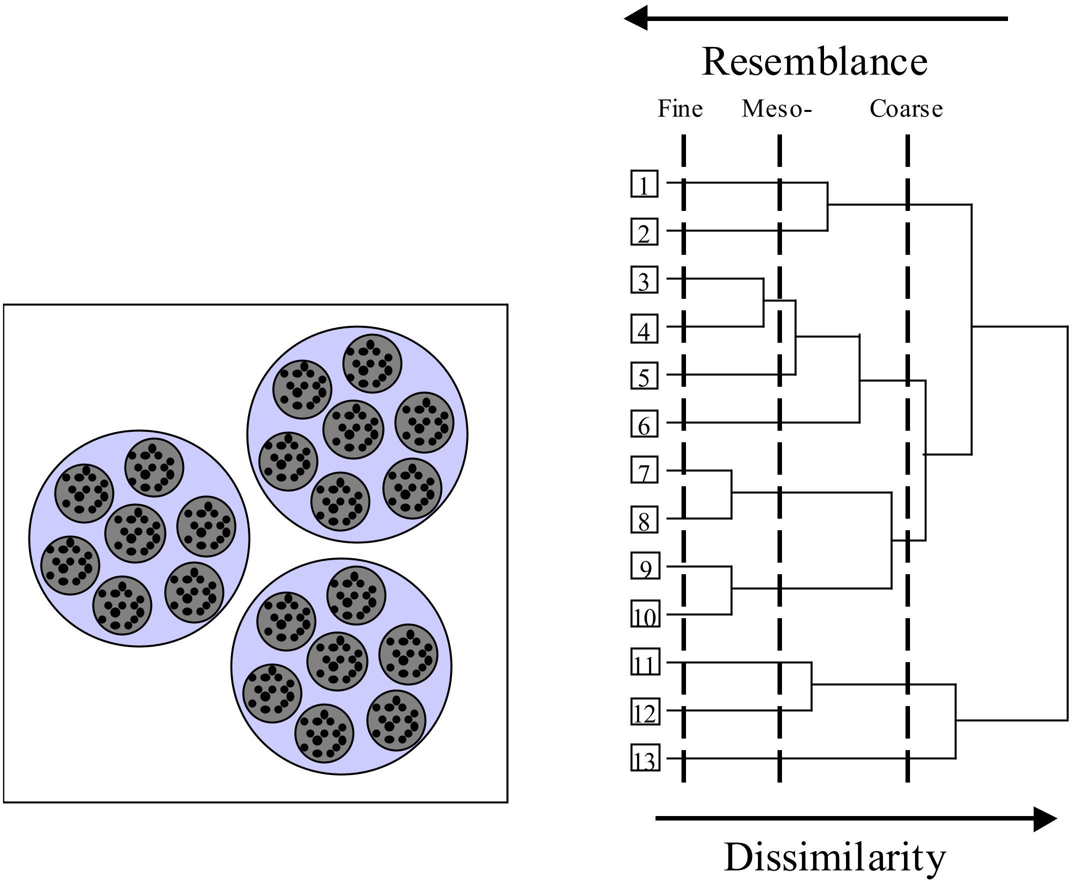
Table 7. Examples of data analyses that have been used to test and support or reject listed communities

Reference	Ecological Community
Keith & Scott (2005)	Coastal floodplain communities
Mackenzie & Keith (2007)	Sandhill Pine Woodland
Tozer (2003)	Cumberland Plain Woodland, Blue Gum High Forest
Kendall & Snelson (2009)	Brown Barrel
Soderquist & Irvin (2008)	Old Man Saltbush
Tindall *et al.* (2004)	Highland Basalt
Keith (1994)	O'Hares Creek Shale Forest
Orscheg et al. (2006)	Southern Sydney Sheltered Forest
Sivertsen & Metcalfe (1995)	Myall Woodland
Adam *et al.* (1988)	Coastal saltmarsh

The review process needs to examine whether the methods have been validly applied and the outputs validly interpreted. It also needs to examine whether the input data have been adequately proofed and edited (where necessary) to ensure consistency of species nomenclature and abundance estimates. These issues are discussed further in Section 4.2.1.

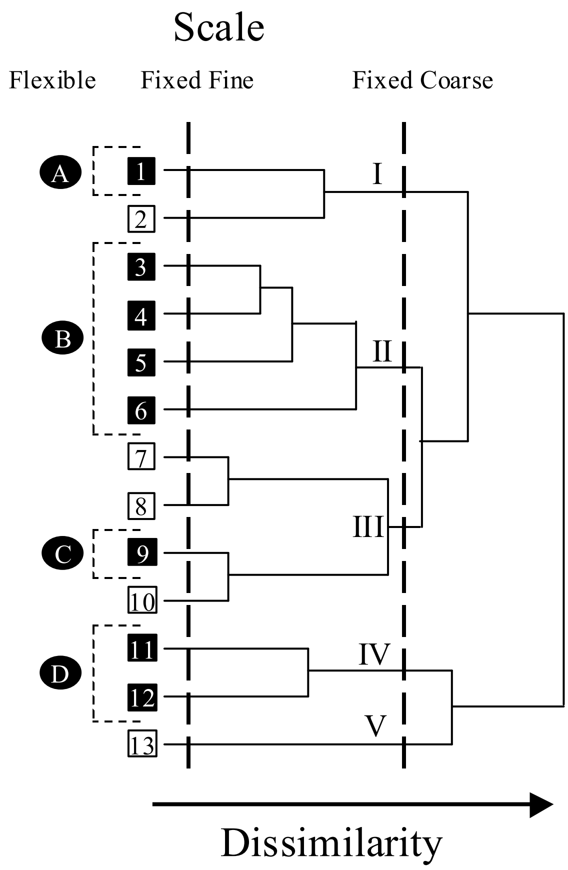
Ecological communities exist at a range of thematic, spatial and temporal scales. It is axiomatic that any community comprises two or more other communities, which are defined at finer levels of organisation (Keith 2009, Box 7). The TSC Act does not specify any requirement for ecological communities to be defined at any particular scale in order to be eligible for listing. Preston & Adam (2004a) note that 'it is possible to satisfy the three requirements [in the definition of an ecological community] at various levels of specificity [thematic scales] and spatial scales.' This provides important flexibility in the listing of communities at a range of scales to cover the diversity of assemblages in nature and deal with the processes that threaten them, also at a range of scales. Nicholson *et al.* (2009) suggest that there are economies in listing communities at the broadest scale at which they meet criteria for listing (see Box 7). Keith (2009) includes further discussion of the concepts and community theory relevant to definition and description of ecological communities.

Box 7. Ecological communities at different scales.



(a) (b)

From Keith (2009). 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 13 c ommunities 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).



From Nicholson *et al.* (2009). 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). F or 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.

4.1.3 <u>Particular area</u>

The particular area defines the location(s) at which species of the assemblage co-occur. According to Preston & Adam (2004a), this represents the ‘natural habitat in which the assemblage of species occurs or has historically occurred and is capable of recurring if measures are taken to restore or allow the habitat to recover.' It therefore excludes captive or cultivated occurrences because they are not within the 'natural habitat' of the community. In Determinations, the particular area of an ecological community is often described by identifying the bioregions in which it occurs and the Local Government Areas in which it has been recorded (see Section 4.2.2).Description of an ecological community

4.2 Description of an ecological community

Descriptions of ecological communities have multiple roles in listing statements or determinations, where they support regulatory, conservation and recovery operations, as well as legal actions (Keith 2009). 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.

Preston & Adam (2004a) outline the legal and practical requirements for a description of an ecological community under the TSC Act. There are three legal requirements of a description, which are implicit in the statutory definition of an ecological community (see section 4.1). The principal practical requirement of a description is that it be 'interpretable by a reasonably well-informed lay person, at least to the extent of knowing they need to seek professional advice' (Preston & Adam 2004a).

The multiple roles of determinations sometimes necessitate trade-offs in the content and form of community descriptions (Keith 2009). For example, a requirement for simplified expression of the key features of a community to inform 'lay persons' might not include sufficient detail or caveats to support a rigorous scientific diagnosis of a community. Similar trade-offs apply to the flexibility of community descriptions. Flexibility that acknowledges some degree of uncertainty is essential in the description of a community to accommodate the natural variability in its properties (Preston & Adam 2004a). 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). The NSW Courts have taken a pragmatic approach to uncertainty. Chief Justice Spigelman[1] 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."*

[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

The Scientific Committee has developed a format for listing of threatened ecological communities that contains the following elements:

Parts 1 & 2: Section 4 of the Act defines an ecological community as "an assemblage of species occupying a particular area". These features of an ecological community are described in Parts 1 and 2 of this Determination, respectively.

Part 3: Part 3 of this Determination describes the eligibility for listing of this ecological community in Part 2 of Schedule 1A of the Act according to criteria as prescribed by the Threatened Species Conservation Regulation 2010.

Part 4: Part 4 of this Determination provides additional information intended to aid recognition of this community in the field.

4.2.1 Describing the assemblage of species

Part 1 of the current format of Scientific Committee determinations provides details on the assemblage of species in an ecological community. The description of an ecological community should include a list of characteristic constituent species. The construction of the list of characteristic species will depend on available information (e.g. qualitative descriptions *cf.* quantitative floristic data), and will vary from case to case (see Preston & Adam 2004a). However, descriptions would usually aim to include frequently occurring species, those that may be locally abundant, though not necessarily present throughout the distribution of the community, and species whose occurrence may help to distinguish the community from other similar communities. It may also be appropriate to list rare or threatened species to draw attention to their occurrence within the community. In some cases, it may be appropriate to subdivide the list of characteristic species to make it clear which species are likely to occur frequently throughout the community distribution (e.g. see Final Determination of Sandhill Pine Woodland in the Riverina, Murray-Darling Depression and NSW South Western Slopes bioregions).

Ideally, there should be a clear rationale for including any given species in the list of characteristic species that describes a listed ecological community. This rationale will depend on the nature of available data on constituent species, which varies from case to case. For example, in some cases, the entire distribution of a nominated community is covered by a systematic ecological survey (e.g. Cumberland Plain Woodland, Tozer 2003) or it may be possible to carry such an analysis as part of the assessment of a nominated community (see section 4.1.2). Where this includes systematic sampling of species composition, it may be appropriate to select species for inclusion in the description if their frequency of occurrence within plots assigned to the community exceeds a given threshold. Inclusion of less frequent species may also be warranted if they exhibit high fidelity with the nominated community (*cf.* other communities). Suitable thresholds for frequency and fidelity will vary depending on the level of sampling and species richness of the community. Where quantitative compositional data are unavailable or provide only partial coverage, an alternative rationale is needed. For example, several independent qualitative descriptions or species lists may be available for different or overlapping parts of the community distribution. It may be appropriate to compile a list of characteristic species from those that occur most frequently across those multiple sources.

Given the inherent spatial and temporal variability in ecological communities and limited detectability of some species, the list of characteristic species will be a sample of the total number that comprise the community. The NSW courts have recognised the impracticality of providing a complete enumeration of all constituent species in a community (Preston & Adam 2004a). Nonetheless, appropriate caveats on the interpretation of such a list should be included in Determinations of ecological communities (Box 8).

Box 8. Example text explaining caveats on the list of characteristic species describing ecological communities listed under the TSC Act.
The total species list of the community across all occurrences is likely to be considerably larger than that given above. Due to variation across the range of the community, not all of the above species are present at every site and many sites may also contain species not listed above.

Characteristic species may be abundant or rare and comprise only a subset of the complete list of species recorded in known examples of the community. Some characteristic species show a high fidelity (are relatively restricted) to the community, but may also occur in other communities, while others are more typically found in a range of communities.

The number and identity of species recorded at a site is a function of sampling scale and effort. In general, the number of species recorded is likely to increase with the size of the site and there is a greater possibility of recording species that are rare in the landscape.

Species presence and relative abundance (dominance) will vary from site to site as a function of environmental factors such as soil properties (chemical composition, texture, depth, drainage), topography, climate, and through time as a function of disturbance (eg fire, logging, grazing) and weather (eg flooding, drought, extreme heat or cold).

At any one time, above ground individuals of some species may be absent, but the species may be represented below ground in the soil seed bank or as dormant structures such as bulbs, corms, rhizomes, rootstocks or lignotubers.

The species listed above are vascular plants, however the community also includes micro-organisms, fungi and cryptogamic plants as well as vertebrate and invertebrate fauna. These components of the community are less well documented.

Patterns of species co-occurrence may sometimes be complex and recognisable at a range of spatial, thematic and temporal scales (Keith 2009). As mentioned above, multi-variate analyses of species composition, where adequate data are available, may assist in resolving descriptions of ecological communities (Kendall & Snelson 2009). Given the emphasis on assemblages of species in the TSC Act, quantitative approaches that address overall species composition are clearly desirable (Preston & Adam 2004a). In vegetation science, for example, the Zurich-Montpellier system is an internationally established approach to the classification of species assemblages, the principles of which are applicable to any biological system (Muller-Dombois & Elenberg 1974, Bridgewater 1981). A rigorous analysis of this sort will usually require evaluation and 'quality control' of the data (e.g. Keith & Bedward 1999) and supplementary analyses to assess the likelihood that the results are influenced by artefacts in the data or coverage of samples. The standardisation of taxonomic nomenclature, removal of indeterminate taxa and standardisation of abundance measures are three common pre-requisites, particularly for meta-analyses that draw data from multiple sources. Keith & Bedward (1999), Tozer *et al.* (2010) and recent examples of such applications include Keith & Scott (2006), Mackenzie & Keith (2007), NSW Scientific Committee & Mackenzie (2008), Soderquist & Irvin (2008) and others listed in Table 7.

In addition to listing the characteristic species of the assemblage, a Determination of an ecological community may include a text description that incorporates features such as the relative abundance, dominance, growth forms, vertical stratum and geographic occurrence of component species. Box 9 provides an example description incorporating these features.

Box 9. Example of a text description of an ecological community (from the Final Determination of Sandhill Pine Woodland as an Endangered Ecological Community).

Sandhill Pine Woodland is characterised by an open tree stratum, which may be reduced to isolated individuals or may be absent as a result of past clearing. The tree layer is dominated by *Callitris glaucophylla* (White Cypress Pine), either in pure stands or with a range of other less abundant trees or tall shrubs. These may include *Acacia melvillei, A. oswaldii, Allocasuarina luehmannii* (Buloke), *Callitris gracilis* subsp. *murrayensis* (Slender Cypress Pine), *Hakea leucoptera* (Needlewood), *H. tephrosperma* (Hooked Needlewood), *Myoporum platycarpum* (Sugarwood) and *Pittosporum angustifolium* (Berrigan). A scattered shrub layer is sometimes present and may include *Dodonaea viscosa* subsp. *angustifolia, Enchylaena tomentosa* (Ruby Saltbush), *Sclerolaena muricata* (Black Rolypoly) and/or *Maireana enchylaenoides* (bluebush). The groundcover is highly variable in structure and composition. It may be sparse or more continuous, depending on the history of disturbance, grazing and rainfall events. It comprises grasses, such as *Austrodanthonia caespitosa* (Ringed Wallaby Grass), *A.setacea* (Small-flowered Wallaby Grass), *Austrostipa nodosa* (a speargrass), *A.scabra* (Rough Speargrass), *Enteropogon acicularis* (Curly Windmill Grass), *Panicum effusum* and *Paspalidium constrictum*; and forbs including *Atriplex semibaccata* (Creeping Saltbush), *Einadia nutans* (Climbing Saltbush), *Erodium crinatum* (Blue Storksbill), *Oxalis perennans, Sida corrugata* (Corrugated Sida) and *Wahlenbergia* spp. (bluebells). The structure of the community varies depending on past and current disturbances, particularly clearing, logging, grazing and soil erosion.

4.2.2 Describing the particular area

Part 2 of the current format of Scientific Committee determinations provides details on the particular area occupied by an ecological community. The 'particular area' occupied by an ecological community needs to be described with reasonable specificity, but need not be highly prescriptive (Preston & Adam 2004a). In NSW, the Land and Environment Court and the Court of Appeal have held that it is sufficient to specify the bioregions in which a community occurs and the local government areas in which it has been recorded (Preston & Adam 2004a). Consequently, for most ecological communities listed in NSW, the 'particular area' is defined by one or more bioregions (as described by SEWPaC (2012) or in older determinations, Thackway & Creswell 1995), which are usually incorporated into the name of the community. The bioregions in which a community occurs are currently detailed in Part 2 (particular area occupied by the ecological community) of current format determinations, while also information relating to Local Government Areas in which the community occurs is provided in Part 4 (additional information). This is to assist regulatory applications under the TSC Act but it is not intended to be exhaustive and that the community may be recorded in other LGAs as knowledge of its distribution develops.

Although not legally required, information about 'supplementary descriptors' (section 4.2.3, Part 4 of the current format of Scientific Committee determinations deals with additional information about the ecological community) may also assist in the interpretation of the particular area occupied by an ecological community. For example, environmental conditions such as typical climate, terrain, substrates and other abiotic, biotic or ecological factors that influence the community can assist in drawing inferences about its likely occurrence at particular locations (Preston & Adam 2004b). However, supplementary descriptors must be regarded as a useful adjunct, rather than a substitute for a description of the particular area occupied by a community (Preston & Adam 2004b).

The NSW Land and Environment Court and the Court of Appeal have held that maps showing the distribution of an ecological community are not an essential legal requirement for describing the particular area in which they are found (Preston & Adam 2004a). Mapped boundaries of communities are subject to a range of uncertainties (Keith 2009), commonly mis-interpreted, and generally unsuitable for a definitive delineation of areas subject to listings under the TSC Act. Consequently, with the exception of highly localised assemblages, maps are rarely included as part of Determinations of ecological communities to avoid overly prescriptive mis-interpretations of the 'particular area' of occurrence.

Despite their limitations, maps can provide indicative guidance on distribution of ecological communities and useful data for assessing their status (see section 4.3.1). To provide non-prescriptive guidance, many Determinations therefore make reference to studies (if available) that map the distributions of units that are related to the listed ecological community. The nature of the relationship between the listed community and the mapped unit will determine how the map can inform about the particular area of the listed community and this varies from case to case. Sometimes, the listed community corresponds directly with the mapped unit. In other cases, the mapped unit is part of the listed community, or the listed community may be included within a broader mapped unit. The reference to relevant mapping studies, rather than direct incorporation of maps as part of Determinations, 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.

4.2.3 Supplementary descriptors

Part 4 of the current format of Scientific Committee determinations deals with additional information about the ecological community. Preston & Adam (2004b) identify 'supplementary descriptors' that may assist interpretation, providing greater certainty to the description of a community and permitting easier recognition in the field. Structural features are examples of supplementary descriptors of the community mentioned in Box 8. Structural features include the vertical and horizontal spatial arrangement of individual organisms within the community (Keith 2009). Other supplementary descriptors include the following:

- physiognomic features of a community (e.g. the texture, size and orientation of leaves, and the range of life forms in a plant community)
- relationships of the community to abiotic factors (e.g. the climate, landforms, hydrological features and substrates that define the environmental conditions in which the community occurs);
- other biotic features (e.g. interactions between component species, responses to processes such as herbivory, etc.); and
- dynamic features (e.g. relationships to disturbance regimes, successional properties, etc.).

As with other features of a community, its structure, physiognomy, relationships with abiotic factors, biotic and dynamic features may be variable and uncertain. Consequently, appropriate qualifiers should be used to describe these features. In some cases, it may be possible to describe the nature of variation. Box 10 includes an example of reference to the variation in a number of supplementary descriptors of an ecological community. The phrases underlined illustrate the indicative, rather than prescriptive intent of the information provided on supplementary descriptors.

Box 10. Extract from the Final Determination of Lowland grassy woodland of the South East Corner bioregion, including reference to supplementary descriptors.

Lowland Grassy Woodland in the South East Corner bioregion is the name given to the ecological community associated with rainshadow areas of the south coast and hinterland of New South Wales. These rainshadow areas receive less rainfall than more elevated terrain that partially surrounds them, with mean annual rainfall typically in the range of 700-1100 mm. The community typically occurs in undulating terrain up to 500 m elevation on granitic substrates (e.g. adamellites, granites, granadiorites, gabbros, etc.) but may also occur on locally steep sites and on acid volcanic, alluvial and fine-grained sedimentary substrates. Lowland Grassy Woodland in the South East Corner bioregion is characterised by the assemblage of species listed in paragraph 2 and typically comprises an open tree canopy, a near-continuous groundcover dominated by grasses and herbs, sometimes with layers of shrubs and/or small trees. Undisturbed stands of the community may have a woodland or forest structure. Small trees or saplings may dominate the community in relatively high densities after partial or total clearing. The community also includes 'derived' native grasslands which result from removal of the woody strata from the woodlands and forests.

The question of whether supplementary descriptors can be determinative regarding the occurrence of a listed community at a given location has been controversial. Some environmental consultants have argued that a listed community cannot be present at a site if the features of the site do not match the supplementary descriptors in the Final Determination, irrespective of whether the assemblage of species and particular area match those described in the Final Determination (e.g. NSWLEC 102[2]). This determinative interpretation is rarely consistent with the Committee's intent in providing information about supplementary descriptors to assist identification of a community. Courts have generally taken a broad (non-determinative) interpretation of supplementary descriptors (e.g. NSWLEC 297[1] - VAW Kurri Kurri vs Scientific Committee 2003, NSWLEC 770[3]). Preston & Adam (2004b) stress that supplementary descriptors…

> *"cannot be used as a substitute for a description of the assemblage of species and the particular area in which the community is located. Rather they should be seen as a valuable adjunct."*

This reasoning stems from the statutory definition of an ecological community. Nonetheless determinative interpretations of supplementary descriptors continue to be presented (e.g. NSWLEC 102[2]), and it is important that wording of Determinations gives guidance as to whether a broad interpretation is intended.

4.3 Clause 17 – reduction in geographic distribution

Reductions in geographic distribution are one of the key symptoms of extinction risk for ecological communities (Rodriguez *et al.* 2007; Nicholson *et al.* 2009, Keith *et al.* 2013). Ecological communities that have undergone large reductions, or are likely to undergo large reductions in the future, are generally exposed to greater risks of extinction than those that have undergone or are likely to undergo smaller reductions, or unlikely to undergo any reduction. Furthermore, a significant reduction in geographic distribution almost certainly entails a significant loss of diversity in the community, particularly in communities with strong spatial structure and component species with limited dispersal abilities. Clause 17 specifies varying levels of reduction as eligibility criteria for listing in respective categories of threat, and is therefore analogous to Clause 6 for the assessment of species. Interpretation of the two clauses must therefore be logically consistent, and much of the interpretive guidance for Clause 6 is relevant to the interpretation of Clause 17. To be eligible for listing under Clause 17, communities that have undergone a sufficiently large reduction within the relevant past time frame need not exhibit evidence of a continuing decline.

4.3.1 <u>Estimating reduction in distribution</u>

Estimating the reduction in the geographic distribution of an ecological community for assessment under Clauses 17 will usually require spatial data from which the mapped extent of the community can be determined at two or more points in time.

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[2] Gordon Plath of the Department of Environment and Climate Change v Vurlow; Gordon Plath of the Department of Environment and Climate Change v H ockey; Gordon Plath of the Department of Environment and Climate Change v Southton [2009] NSWLEC 102, Pain J. http://www.lawlink.nsw.gov.au/lecjudgments/2009nswlec.nsf/c45212a2bef99be4ca256736001f37bd/6dd06e8b6ddcb133ca2575de002b80c1?OpenDocument

[3] Motorplex (Australia) Pty Ltd v Port Stephens Council [No 2] [2007] NSWLEC 770, Preston CJ. http://www.lawlink.nsw.gov.au/lecjudgments/2007nswlec.nsf/00000000000000000000000000000000/10994c3cfc78186bca25739c0000631e?opendocument

Most commonly, temporal reference point for mapping will include a contemporary distribution and a historic distribution that has been projected (by various means) to a point just prior to European settlement. Assumptions and mathematical interpolations with then need to be made to calculate reductions over the time period relevant to assessment of Clause 17 (see Section 4.3.3). The map data can be used directly to estimate a reduction where one or more map unit(s) correspond(s) directly with the ecological community (section 4.2.2). Where there is an indirect relationship (e.g. the distribution of the ecological community is incompletely represented by the map unit(s)) further spatial analysis may be required, given plausible assumptions about the maximum and minimum likely extent of the community. In some cases, it may be prudent to combine spatial data from two or more different mapping studies that cover different parts of the community's distribution. Keith *et al.* (2009) and Keith *et al.* (2013) outline a detailed protocol with worked examples for estimating changes in distributions of ecological communities. An important aspect of the protocol is to calculate reductions for a set of scenarios that cover a plausible range of assumptions about the relationships between the map data and the distribution of the community. Different cases are likely to require different sets of scenarios and different assumptions which require evaluation and justification on a case by case basis.

Where spatial data on the distribution of an ecological community are unavailable or incomplete, it may be possible to infer reductions from proxy spatial data. For example, maps of land use may allow inferences to be drawn about the area of land within the distribution of a community that has been converted to urban or agricultural uses.

4.3.2 Magnitude of reductions in distribution

To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, an ecological community must have undergone or be projected to undergo **very large** (Subclause 17a)**, large** (Subclause 17b) **or moderately large** (Subclause 17c) **reductions** in population size. Keith *et al.* (2013) have developed the IUCN red list criteria for ecosystems and these represent the internationally accepted criteria for assessing the status of ecological communities. Nicholson *et al.* (2009), reviewing twelve assessment protocols developed in different countries, found considerable variation in quantitative thresholds of reduction and little or no evidence of a clear rationale for the thresholds applied in each case. Keith *et al.* (2013) have proposed thresholds for decline and these largely reflect the IUCN (2001) species thresholds.

For the purpose of interpreting Clause 17, rather than setting arbitrary thresholds, it is recommended that indicative guidance be sought by comparison with corresponding thresholds for reduction in species populations adopted by IUCN (2001) (Table 8). This reasoning is justified as follows:

- The descriptors for reductions in the geographical distribution of ecological communities in Clause 17 are identical to those in Clause 6 for species populations (CR- very large, EN- large, VU- moderate), which suggests that similar thresholds of reduction are intended to apply to species and communities.
- Thresholds of reduction for communities should be no more stringent than equivalent thresholds for reduction of species populations to ensure logically consistent outcomes between assessments of species and communities and that ecological communities fulfil their roles in conservation of poorly known or rare

species, genetic variability and processes that operate above the species level (Nicholson *et al.* 2009, Keith 2009). If thresholds for communities were more stringent than those for species, a community would need to decline further than an individual species to be at similar risk of extinction.

- Communities commonly have strongly spatially structured patterns of species diversity and many of their constituent species have poor dispersal ability, and these features predispose communities to significant losses of diversity when their distributions are depleted. This spatial structure also violates a key assumption of species-area curves, that variation in species composition is randomly distributed across space.
- Loss of area of suitable habitat may result in disproportionate declines in species populations (Rodriguez 2002) and may therefore result in disproportionate loss of community diversity.
- Alignment of thresholds for reductions in species and communities follows leading international protocols including the NatureServe protocol (Faber-Langendorn *et al.* 2007) and the Finnish National Assessment of threatened habitat types (Kontula & Raunio in press).

Table 8. Corresponding thresholds for reductions in population size for the TSC Regulation 2010 and the IUCN (2001) Red List criteria. Based on IUCN (2001), past and projected reductions in species populations may be interpreted under Subclauses 17a, 17b and 17c using more stringent numerical thresholds of criterion A1 if the causes of reduction are clearly reversible AND understood AND ceased. If any of these conditions do not apply, the standard thresholds in criteria A2, A3 and A4 are appropriate (IUCN 2001).

Category of threat	Requirement under Clause 17 of TSC Regulation 2010	Thresholds for reduction under criteria A2, A3 and A4 of IUCN (2001)	Thresholds for reduction under criterion A1 of IUCN (2001)
Critically Endangered	very large	≥80%	≥90%
Endangered	large	≥50%	≥70%
Vulnerable	moderately large	≥30%	≥50%

4.3.3 <u>Time frames for assessing reductions in distribution</u>

Reductions in the distribution of an ecological community must be assessed over a **time span appropriate to the life cycle and habitat characteristics of its component species**. Based on IUCN (2001), a time frame appropriate to the life cycle is three generation lengths or 10 years, whichever is the longer. Generation length is defined by IUCN (2001, 2014) (see Box 1). In most cases, habitat characteristics will not alter the appropriate time frame determined from generation length. In exceptional circumstances, where an appropriate time frame for assessing reductions cannot be inferred from generation length, turnover in habitat may be used as a proxy for generation length. Usually, this will be appropriate for communities in which dynamics is linked to environmental processes, such as recurring climatic cycles (e.g. El Niño Southern Oscillation), flood regimes, extreme droughts, stand-replacing storm or fire events, fire regimes, etc. In most forests, woodlands and many shrub-dominated communities, three generation lengths will extend beyond 200 years into the past and the appropriate time span for assessing reductions will be between European settlement and the present day. For many terrestrial plant communities, historical distributions have been reconstructed using quantitatively or intuitively derived relationships between the occurrence of a community and environmental variables for which spatial data are available (e.g. Keith & Bedward 1999).

4.3.4 Influence of spatial scale

The spatial scale of map data will influence estimates of reduction in distribution. Finer-scale maps will be more proficient than coarse-scale maps at detecting small patches from which the community has disappeared and other small patches in which the community remains extant. For this reason, it is **important that chronosequential maps used in temporal analysis to estimate reductions are at similar spatial scales and resolution**. If this is not the case, then it is advisable to convert the finer-scale map(s) to the scale of the coarser map(s), even though this may involve loss of information from the finer-scale map(s). Scale standardisation will reduce any scale-related artefacts in estimates of change based on the map data. In contrast to estimates of area (section 4.4.2, Box 5), estimates of change are unlikely to be very sensitive to the actual scale of maps used in assessment, so long as the chronosequential maps are of a scale consistent with one another. However, where relevant spatial data sets are available at a variety scales, it is advisable to use the finest available spatial data that is common to both chronological reference points to obtain estimates with the highest level of precision.

4.3.5 Calculating reductions in distribution

IUCN (2014) sections 4.5 and 5 provide extensive advice on the interpretation and calculation of reductions in species' populations, much of which is relevant to calculating reductions in the geographic distributions of ecological communities.

Reductions should be averaged across the entire distribution of a community. The extent of a community may be changing at different rates in different parts of its distribution. Where changes in distribution have been sampled in different parts of a community's distribution, the changes should be weighted by the extent of the community at the beginning of the assessment period in each sampled area. For example, declines in areas with extensive occurrences of the community will outweigh increases or stability in areas with small occurrences. Box 3 and IUCN (2014) give example calculations for species, while Keith *et al.* (2009) demonstrate a similar application for an ecological community. Where spatial data on reduction are only available for part of the distribution of the community, an inference about the level of reduction across the entire distribution will depend on plausible assumptions about whether the reduction observed in the sampled area is likely to be representative of more general trends. This will usually involve reference to the causes and mechanisms that are driving the reduction.

Available data on reductions in the distribution of a community may not correspond to the "time frame appropriate to the life cycle and habitat characteristics of its constituent species" over which reductions must be assessed against the listing criteria. For example, estimates may be available for reductions that occurred over the last 50 years (where remote imagery is available), but reductions may need to be assessed over a shorter time frame. Analogous situations may arise in species (see section 2.1.7 & Box 3). In such cases, interpolation or extrapolation may be required where the data are available for a longer or shorter period than the required time frame for assessing reductions. In both cases, the best approach is to fit a regression model to the available data and use the appropriate time interval (e.g. between the present year and three generations lengths prior) on the fitted line to calculate the reduction. Fitting a model in this way helps to eliminate some of the variability that may be attributable to short-term causes. Interpolation or extrapolation will require assumptions about the data and the trend that should be justified with reference to the processes driving the decline (e.g. pattern of land use change). For example, depending on the shape of the data, a linear or exponential regression model may be fitted. Keith *et al.* (2009) provide an example in which future reductions are projected assuming alternative linear and exponential patterns of future decline. Assumptions about the rate of decline remaining constant, increasing or decreasing, relative to the observed interval must be justified, especially where the reduction in distribution is estimated over long generation times from data over shorter time frames.

4.4 Clause 18 – size of geographic distribution

An ecological community will be eligible for listing as threatened under Clause 18 if it meets two conditions: its geographic distribution is more restricted than a threshold extent; and. a threatening process could cause it to decline in distribution or ecological function. Clause 18 is therefore analogous to Clause 8 for assessing the status of species and comparable with Keith et al. (2013). Communities with restricted geographic distributions will have fewer opportunities for persistence because it is more likely that a single or small number of threatening processes or events will adversely affect the entire distribution. Conversely, the larger the distribution of a community, the more these risks of exposure to threats will be spread across different locations.

4.4.1 Geographic distribution

Geographic distribution is defined in Clause 23(1) of the TSC Regulation 2010 as 'the area or areas in which a species [or ecological community] occurs, excluding cases of vagrancy.' Geographic distribution may be assessed in a number of different ways, including the extent of occurrence, area of occupancy and area of suitable habitat.

4.4.2 Measures of geographic distribution

Under Clause 18, the geographic distribution of an ecological community may be assessed by estimating the extent of occurrence, the area of occupancy or the area of suitable habitat. Each of these terms is defined in Clause 23(2).

(a) **Extent of occurrence** (EOO) is defined in Clause 23(2a) as the area of the total geographic range that includes all extant occurrences of the community. Its application in Clause 18 follows criterion B1 in IUCN (2001). Extent of occurrence can often be measured by a minimum convex polygon or convex hull (the smallest polygon in which no internal angle exceeds 180 degrees and which contains all the sites of occurrence). IUCN (2001) states that EOO may exclude discontinuities or disjunctions within the overall distributions of taxa (e.g. large areas of obviously unsuitable habitat). However, the consequences of excluding discontinuities vary, depending on whether the estimate of EOO is to be used for assessing the total distribution in Clause 18 or whether it is to be used for estimating or inferring reductions (Clause 17). Box 4 summarises guidance from IUCN (2014) on how to estimate EOO under these different criteria.To ensure consistency with the definition of Area of Occupancy (AOO), if EOO is less than AOO, EOO should be changed to make it equal to AOO.

(b) **Area of occupancy** (AOO) is defined in Clause 23(2b) as the area within the total range (and hence within EOO) that is currently occupied by the community. It. excludes unsuitable and unoccupied habitat. Its application in Clause 18 follows criterion B2 in IUCN (2001). IUCN (2014) explains the rationale underpinning AOO as follows: "Suppose two species [or communities] have the same EOO, but different values for AOO, perhaps because one has more specialised habitat requirements. For example, two species [or communities] may be distributed across the same desert (hence EOO is the same), but one is wide ranging throughout (large AOO) while the other is restricted to oases (small AOO). The species [or community] with the smaller AOO may have a higher risk of extinction because threats to its restricted habitat (e.g. degradation of oases) are likely to reduce its habitat more rapidly to an area that cannot support viable populations of constituent species. The species [or community] with the smaller AOO is also likely to have smaller population sizes than the one with a larger AOO, and hence is likely to have higher extinction risks for that reason" (IUCN 2014). Estimates of AOO are highly sensitive to scale of measurement (Keith 2009, Nicholson *et al.* 2009). The scale-dependence of AOO is similar, irrespective of whether species or communities are being assessed, and recommendations for a method of estimation are discussed in Box 5. In summary, the area of occupancy of the community should be assessed at a standard assessment scale, for which the community' distribution can be represented by occurrence uniform grid cells of appropriate size, irrespective of the format of the raw spatial data as polygons, grids, lines or points (see Box 5). The standard assessment scale is determined essentially by the value of the thresholds that discriminate different categories of threat (Table 9), not be characteristics of the available data or the community. For this reason, it is often necessary to convert fine-scale mapping to a coarser scale for the purpose of assessing estimates of AOO against the thresholds. The recommended scale for assessment of thresholds adopted by IUCN (2001) is 2 x 2 km grid cells (IUCN 2014, Box 5). Electronic supplementary material available in Nicholson *et al.* (2009) gives examples of estimated areas of occupancy of communities at different spatial scales.

(c) Area of suitable habitat is defined in Clause 23(2c) as the area within the total range that includes occupied and unoccupied suitable habitat, but excludes unsuitable habitat. Maps of suitable habitat for communities may be derived (as those for species) from interpretation of remote imagery and/or analyses of spatial environmental data using simple combinations of GIS data layers, or by more formal statistical habitat models (e.g. generalised linear and additive models, decision trees, Bayesian models, regression trees, etc.). Habitat maps can provide a basis for estimating AOO and EOO and, if maps are available for different points in time, rates of change can be estimated (IUCN 2014). They cannot be used directly to estimate a community's AOO because they often map an area that is larger than the occupied habitat (i.e. they also map areas of suitable habitat that may presently be unoccupied). However, they may be a useful means of estimating AOO indirectly, for which IUCN stipulates three conditions that must be met.

i) Maps must be justified as accurate representations of the habitat requirements of the community and validated by a means that is independent of the data used to construct them.
ii) The mapped area of *suitable* habitat must be interpreted to produce an estimate of the area of *occupied* habitat.
iii) The estimated area of occupied habitat derived from the map must be scaled to the grid size that is appropriate for AOO of the species.

Habitat maps can vary widely in quality and accuracy (condition i). A map may not be an accurate representation of habitat if key variables are omitted from the underlying model. For example, a map would over-estimate the habitat of a montane forest-community if it id entified all forest areas as suitable habitat, irrespective of altitude. The spatial resolution of habitat resources also affects how well maps can represent suitable habitat. For example, specialised edaphic, hydrological or physical habitat features, such as a flood regime or cliffs with particular moisture seepage regimes, may not lend themselves to mapping at coarse scales. Application of habitat maps to the assessment of communities for listing under the TSC Act, should therefore be subject to an appraisal of mapping limitations, which should lead to an understanding of whether the maps over-estimate or under-estimate the area of suitable habitat.

Habitat maps may accurately reflect the distribution of suitable habitat, but only a fraction of suitable habitat may be occupied (condition ii). Therefore the area of suitable habitat may be an upper bound of the possible AOO although, depending on the proportion of suitable habitat actually occupied, it could be substantially larger than any plausible upper bound of AOO. Low habitat occupancy may result because other factors are limiting – such as past disturbance, dispersal limitations, etc. In such cases, the area of mapped habitat could be substantially larger than AOO and will therefore need to be adjusted (using an estimate of the proportion of habitat occupied) to produce a valid estimate of AOO. This may be done by random sampling of suitable habitat grid cells, which would require multiple iterations to obtain a stable mean value of AOO (IUCN 2014).

Habitat maps are produced at a resolution determined by the input data layers (satellite images, digital elevation models, climate surfaces, etc.). Often these will be at finer scales than those required to estimate AOO (condition iii), and consequently scaling up will be required (see Box 5).

4.4.3 Size of geographic distribution

To be eligible for listing as Critically Endangered, Endangered or Vulnerable, respectively, an ecological community must have a **geographic distribution** that is estimated or inferred[*] to be **very highly restricted** (Subclause 18a)**, highly restricted** (Subclause 18b) **or moderately restricted** (Subclause 18c) in addition to meeting the other condition relating to the action of a threatening process (section 4.4.4). Nicholson *et al.* (2009), reviewing twelve assessment protocols developed in different countries, found considerable variation in quantitative thresholds of geographic

[*] See Box 2 for definitions of 'estimated' and 'inferred'

distribution and little or no evidence of a clear rationale for the thresholds applied in each case.

- For the purpose of interpreting Clause 18, rather than setting arbitrary thresholds, it is recommended that indicative guidance be sought by comparison with corresponding thresholds for the geographic distributions of species adopted by IUCN (2001) (Table 9). The justification for this approach follows the same reasoning outlined in Section 4.3.2 f or threshold reductions in geographic distribution. Further testing of the thresholds and scale of assessment developed by Keith *et al*. (2013) may warrant the future adoption of those measures.

Table 9. Corresponding thresholds of geographic distribution size for the TSC Regulation 2010 and the IUCN (2001) Red List criteria. The recommended spatial scale for assessment is 2 km grid cells.

Category of threat	Requirement under Clause 18 of TSC Regulation 2010	Thresholds for Extent of Occurrence under criterion B1 of IUCN (2001)	Thresholds for Area of Occupancy under criterion B2 of IUCN (2001)
Critically Endangered	very highly restricted	≤100 km^2	≤10 km^2
Endangered	highly restricted	≤5000 km^2	≤500 km^2
Vulnerable	moderately restricted	≤20000 km^2	≤2000 km^2

4.4.4 Action of a threatening process

An ecological community that meets the requirements for very highly restricted, highly restricted or moderately restricted geographic distribution will only be eligible for listing under Clause 18 if "the nature of its distribution makes it likely that the action of a t hreatening process could cause it to decline or degrade in extent or ecological function over a time span appropriate to the life cycle and habitat characteristics of the ecological community's component species." To invoke this clause, first requires identification and description of relevant threatening processes. One or more of these may be currently listed as Key Threatening Processes, and it will usually be appropriate to address the evidence for these in one or more dedicated paragraphs of a Determination. A second requirement is that the decline in extent or degradation in ecological function must be likely to continue into the future and be non-trivial relative to the total distribution of the community in NSW. For example, a decline in the area of one small patch of the community, when there are many other larger patches of the community, would not be evidence of a 'decline' in the sense implied by Clause 18. Similarly, degradation of community structure that affected less than (say) 1% of the community distribution would not be evidence of a 'decline' in the sense implied by Clause 18. Interpreting whether declines and degradation are non-trivial requires an overall assessment of the total distribution and inference about whether threats observed at particular locations, in summation, are likely to operate across a substantial proportion of the distribution. This evaluation of threats and associated declines and degradation may be qualitative and need not be quantitative, as required for reductions (Box 3).

4.5 Clause 19 – reduction in ecological function

4.5.1 Definition of ecological function

Ecological function is defined in Clause 21 of the TSC Regulation 2010 as follows:

> (1) Ecological function encompasses the ecological processes and interactions that occur within an ecological community.
>
> (2) Ecological function includes the following:
>
> > (a) provision of habitat for native biota,
> > (b) provision of food and other resources for native biota,
> > (c) maintenance of interactions between species (for example, pollination, dispersal, mutualism, competition, predation),
> > (d) cycling, filtering and retention of nutrients,
> > (e) carbon storage or sequestration,
> > (f) maintenance of soil processes,
> > (g) maintenance of catchment scale hydrological and geochemical processes,
> > (h) maintenance of landscape scale ecological processes.
>
> (3) Some of the processes and interactions within ecological communities may depend upon the presence of non-living components such as leaf litter and fallen or standing dead trees.

Ecological function therefore refers to the ability of communities to support their full diversity of species and to sustain their functional roles (e.g. nutrient and water cycling, carbon storage, provision of food, shelter and breeding sites, etc.) in landscapes (Nicholson *et al.* 2009). It also includes functions of component species as predators, decomposers, pollinators, etc. The functions of ecological communities may decline, irrespective of whether their distribution is declining or restricted.

Reductions in ecological function are key symptoms of extinction risk for ecological communities (Rodriguez *et al.* 2007; Nicholson *et al.* 2009, Keith *et al.* 2013). Ecological communities that have undergone large reductions in function, or are likely to undergo large reductions in the future, are generally exposed to greater risks of extinction than those that have undergone or are likely to undergo smaller reductions, or unlikely to undergo any reduction. Furthermore, a significant reduction in ecological function almost certainly entails a significant loss of diversity in the community, as any assemblage embodies interactions and dependencies between its component species. Clause 19 specifies varying levels of reduction as eligibility criteria for listing in respective categories of threat. To be eligible for listing under Clause 19, communities that have undergone a sufficiently large reduction in ecological function within the relevant past time frame need not exhibit evidence of a continuing decline in function.

4.5.2 Estimating reduction in ecological function

Estimating the reduction in the ecological function of an ecological community for assessment under Clause 21 will require evidence as indicated by any of the following Subclauses:

> (d) change in community structure,
> (e) change in species composition,
> (f) disruption of ecological processes,

(g) invasion and establishment of exotic species,
(h) degradation of habitat,
(i) fragmentation of habitat.

To invoke Clause 19, the magnitude of these changes must be non-trivial, as outlined for declines in geographic distribution resulting from the action of threatening processes in Clause 18 (see section 4.4.4).

4.5.3 Magnitude of reductions in ecological function

Reduction in the ecological functions (as opposed to reduction in distribution) of ecological communities is a very real threat to biodiversity, although it remains difficult to incorporate quantitatively in assessment protocols (Nicholson *et al.* 2009), but see Keith *et al.* (2013) for recent attempts to do this. Reductions in function should be interpreted by considering specific types and symptoms of degradation (see section 4.5.1) that are based in ecological theory. 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. They therefore remain to some extent reliant on expert judgment (see Table 10 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 (Faber-Langendoen *et al.* 2007; Master *et al.* 2007) for characterising direct threats to species and communities, based on severity (degree of degradation), immediacy (time frame), 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. Nicholson *et al.* (2009), for example, suggest that communities could be classified 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. Similarly Keith *et al.* 2013 provide examples of how reductions in ecological function may be quantified both in terms of severity and spatial extent of impact.

Setting robust quantitative thresholds for scope, severity and immediacy will require a substantial research effort, and in the interim, assessments will rely upon relative evaluations against examples. While scope and immediacy imply readily quantifiable parameters (e.g. proportion of the total distribution, time span over which reduction takes place, etc.), severity is more complex, although quantification may be feasible in many cases. For example, several quantitative parameters already exist for fragmentation (e.g. Rodriguez *et al.* 2007; McGarigal 2002). Table 10 gives examples of parameters representing ecological functions from current Determinations under the TSC Act (Nicholson *et al.* 2009).

Table 10. 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 (from Nicholson *et al.* 2009).

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 a pprox. 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	-	-
Invasion and establishment of exotic species			
	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	-

4.5.4 Time frames for assessing reductions in ecological function

Reductions in the ecological of an ecological community must be assessed over a **time span appropriate to the life cycle and habitat characteristics of its component species**. This time frame is identical to that required to assess reductions in geographic distribution of ecological communities, and therefore involves the same interpretation. Refer to section 4.3.3.

5 Dealing with uncertainty

The listing criteria should be applied on the basis of available evidence. (IUCN 2001). Inevitably, some aspects of this evidence will be uncertain to varying degrees, but this will not necessarily preclude species, populations or ecological communities from being assessed against the listing criteria. Absence of high-quality data should not deter attempts to apply the criteria (IUCN 2001). Given scarcity of data in many cases, it is appropriate to use the information that is available to make intelligent inferences about the assessment criteria, and hence the overall status of a species, population or ecological community. Inherent uncertainties have been recognised by Courts dealing with TSC matters and are taken into account in Court decisions (Section 4.2, Keith 2009). The following sections provide guidance to identify sources of uncertainty, reduce it where possible, and deal with it explicitly in the listing process.

5.1.1 Sources of uncertainty

Uncertainty is an inherent and pervasive characteristic of all knowledge (Keith 2009). 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. Epistemic uncertainty is generally reduceable by improving knowledge of a system. 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. Estimates of population size are usually subject to these sources of uncertainty. Another example is the species composition of an assemblage, which can be based only on a sample of all occurrences at all times and places, and is 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. Such data are rarely available (see Keith 2009 for an example). Consequently, the limitations concerning community description, and potential for undocumented variations need to be reported as transparently as possible.

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. Other examples include inferences drawn about the overall rate of decline in a species, when only a few populations have been sampled. Uncertainty stems from the fact that trends over the observed populations and time frames may not represent the combined trend across all populations and the full time frame required for assessment. This sort of uncertainty can be reduced with improved size and selection (stratification) of samples. Even where inference and estimation is made less subjective by statistical methods, expert judgements of some sort are sometimes essential to the application of listing criteria. Where this is the case, different experts may make different judgements and many exhibit serial over-confidence about their areas of expertise (Burgman 2005). In such cases a range of methods exist for eliciting and synthesising information from experts (e.g. the Delphi process - see MacMillan & Marshall 2006 and references therein). Key elements of sound elicitation include seeking multiple independent opinions (cf consensus methods), appropriate weighting of different experts, corroboration with independent evidence, and examining the sensitivity of listing decisions to alternative advice.

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 species, population or community, its underlying theory and how these concepts apply to particular cases (Keith 2009). 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 (see Section 4.1.2 and Box 6). 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 (Regan *et al.* 2002). 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). Unlike, epistemic uncertainty, many forms of linguistic uncertainty cannot be reduced by collecting more information. 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 (see Keith 2009).

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 species or community, as well as wide application of biological concepts and ideas (Regan *et al.* 2002). For example, when a species or 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 species or 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 (Keith 2009). Note that there are also issues of vagueness and ambiguity surrounding the interpretation of 'coastal' in this example. Similar issues arise when terms or concepts are subject to context dependence - what appears to be a reduction in one population may be viewed as a fluctuation when more populations and longer time scales are considered.

5.1.2 Dealing with uncertainty in decision-making

All decisions and their outcomes pertaining to the listing of species and communities 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 obtaining more information. This may involve consulting a wider or more balanced set of experts, undertaking more stringent evaluation of their opinions, and/or measuring characteristics such as composition, population trends 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 assessment 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) listing decisions aim to be robust by reducing the risk of 'bad' outcomes as defined by the listing objectives.

Rather than ask deterministically whether a nominated species or community has been reduced below a threshold fraction of its prior distributiuon, in probabilistic language the initial question becomes, 'how likely is it that the nominated community has undergone a particular level of reduction?' The estimation of likelihood then requires relevant sources of uncertainty to be evaluated and any estimates of reduction to be quantified with both a best estimate (most likely value) and an estimate of uncertainty about the best estimate. .

Elith *et al.* (2003) suggest a variety of methods for reducing and quantifying each type of uncertainty. IUCN (2014, section 5.8.2) gives extensive guidance for the treatment of uncertainty for estimating population reduction. 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). The simplest method for characterising uncertainty with subjective probabilities is to estimate upper and lower bounds, which represent the plausible range around the best estimate. Examples for a variety of parameters are given in Box 11.

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). This second piece of information is often overlooked, but must be elicited from relevant sources (data sets, published reports, experts, etc.) to ensure that listing decisions adequately incorporate uncertainty in the best available knowledge.

To that end, the range of values (best estimate, upper and lower bounds) can each be propagated through the assessment to evaluate the sensitivity of the assessment outcome to uncertainty in parameter estimates. The simplest method is by interval arithmetic, though other more sophisticated methods are available (e.g. fuzzy arithmetic; Akcakaya *et al.* 2000). Where the assessment outcome is relatively insensitive to uncertainty, it will produce a single category of threat (e.g. EN). In other cases, the bounded estimates may result in a range of plausible threat categories (e.g. EN-VU), and the Committee will need to decide on a category for listing, given an appropriate attitude to the uncertainty. These attitudes may vary from precautionary (assigning a higher category of threat within the plausible range) to evidenciary (assigning a lower category of threat from within the plausible range), depending on the circumstances and consequences. IUCN (2001, 2014) recommend a 'precautionary but realistic' attitude to uncertainty in liusting decisions.

Box 11. Examples of deriving bounded estimates from uncertain information on listing parameters

<u>Estimating the number of mature individuals</u>
Callitris endlicheri – Many plants are either too small or too numerous to count reliably and/or in a timely manner over large areas. Mackenzie & Keith (2009) estimated the number of seedlings in a population from samples of seedling density. The best estimate was obtained by multiplying the mean density by the area occupied by the population. Confidence intervals representing upper and lower bounds were calculated from the variance in density scaled by the proportion of the total population that was sampled (see Keith 2000 and Mackenzie & Keith 2009 for relevant equations).
Cynachum elegans – The available information on population status often varies within a species. This species comprises a large number of populations (c. 90), of which 40% were surveyed in detail, producing high-precision population estimates, while others were recorded only with cursory observations and no estimate of population size. Scott (2009) calculated the mean and standard deviation of population sizes (13±5) from the surveyed subset of populations to draw inferences about the likely sizes of unsurveyed populations. A best estimate of the total population size (1170 plants) was calculated by multiplying the mean population size by the total number of populations. Upper and lower bounds (720-1620) were calculated from the 90% confidence intervals. The bounded estimate was well within the Endangered threshold (2500 mature individuals) under Clause 8 of the listing criteria, suggesting that the status is unlikely to change, even if a number of previously unrecorded populations are discovered.

Estimating the area of occupancy

Genoplesium baueri – Species locality records commonly extend back to the nineteenth century. In some cases, a species may not have been recently recorded from locations where historical records exist. This may be due to extirpation of the population or because the species has eluded detection. Consequently, the records represent some populations that are certainly extant, some that are almost certainly extinct (e.g. long undetected and habitat destroyed) and some whose status is uncertain (e.g. no recent records, but habitat remains in tact). Detectability may be low for inconspicuous plants such as orchids. Copeland (2008) compiled 25 records on *G. baueri*, of which 17-19 were considered likely by experts to represent extant populations. He estimated that the area of occupancy was 64-72 km^2 , depending on whether the uncertain populations were considered extant.

Sandhill Pine Woodland – Where vegetation mapping studies exist, the assignment of map units to a nominated ecological community can be uncertain for a variety of reasons. Mackenzie and Keith (2007) identified candidate map units from several different mapping studies that may be part of Sandhill Pine Woodland. Some mapping studies suggested that the distribution of the community is more restricted than others. Calculations based on different combinations of map units indicate that the mapped extent of the community is likely to be between 50 000 ha and 120 000 ha.

Estimating the proportional decline in population size or geographic distribution

Pseudophryne coroboree (Southern Corroboree Frog) – Sources of uncertainty in estimation of animal populations include imperfect detectability and variability of numbers in space and time. Bray (2008) compiled data from Hunter *et al.* (2007) and Osborne *et al.* (1999) to estimate population trends from two different kinds of surveys: i) an extensive survey of 40 sites to record whether male frogs were calling during the breeding season; and ii) an intensive census of a large population to record the number of individual males calling during the breeding season. The number of sites with calling males declined by 89% and the number of calling males at the census site declined by 97% over a ten year period. Assuming that the population was either stable or declining (or even underwent a modest increase) prior to the 10-year survey period, the species is highly likely to have undergone a reduction of more than 80% over three generations (21 years) and hence qualifies for listing under Clause 6 of the listing criteria.

Coolibah – Black Box Woodland – Maps of ecological communities are inherently uncertain. Maps produced by different observers will differ to varying degrees for various reasons. Temporal comparisons of maps by different observers can compound these uncertainties. Keith *et al.* (2009) estimated the proportional decline in distribution of this community (since European settlement) using a set of vegetation maps that covered various portions of the community distribution. They calculated a best estimate of decline from maps identified as the most reliable and consistent throughout the distribution. They estimated a lower bound of decline by substituting maps that had the smallest historic distribution and largest contemporary distribution into the calculation. The upper bound was similarly calculated from the maps with the largest historic distribution and smallest contemporary distribution.

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7 References

Adam P, Wilson NC, Huntley B (1988) The phytosociology of coastal saltmarsh vegetation in New South Wales. *Wetlands (Australia)* **7**, 35-45.

Akçakaya HR, Ferson S, Burgman MA, Keith DA, Mace GM, Todd CR (2000) Making consistent IUCN classifications under uncertainty. *Conservation Biology* **14**, 1001-1013.

Austin MP (1985) Continuum concepts, ordination methods, and niche theory. *Annual Review of Ecology and Systematics* **16**, 39-61.

Austin MP, Smith TM (1989). A new model for the continuum concept. *Vegetatio* **83**, 35-47.

Begon M, Harper JL, Townsend CR (2006) 'Ecology: from individuals to ecosystems.' (Blackwell Publishing: Boston, MA)

Beissinger SR, McCullough DR (2002) 'Population Viability Analysis.' (University of Chicago Press: Chicago)

Belbin L (1993) 'PATN Pattern Analysis Package.' (CSIRO Division of Wildlife Ecology: Canberra)

Ben-Haim Y (2001) 'Information-Gap Decision Theory: Decisions Under Severe Uncertainty.' (Academic Press: London)

Benson JS (2006) New South Wales vegetation classification and assessment: introduction - the classification, database, assessment of protected areas and threat status of plant communities. *Cunninghamia* **9**, 331-382.

Bray C (2008) 'Southern Corroboree Frog (*Pseudophryne corroboree*): Review of Current Information in NSW.' Report to the NSW Scientific Committee.

Bridgewater PB (1981) Potential application of the Zurich-Montpellier system of vegetation description and classification in Australia. In 'Vegetation Classification in Australia.' (Eds. AN Gillison and DJ Anderson) pp. 1-9. (CSIRO and ANU Press: Canberra)

Bruelheide H (2000) A new measure of fidelity and its application to defining species groups. *Journal of Vegetation Science* **11**, 167-178.

Burgman MA (2005) 'Risks and decisions for conservation and environmental management.' (Cambridge University Press: Cambridge)

Burgman MA, Ferson S, Akçakaya HR (1993) 'Risk Assessment in Conservation Biology.' (Chapman and Hall: London)

Burgman MA, Fox JC (2003) Bias in species range estimates from minimum convex polygons: implications for conservation and options for improved planning. *Animal Conservation* **6**, 19-28.

Burrows CJ (1990) 'Processes of Vegetation Change.' (Unwin Hyman: London)
Caughley G (1994) Directions in conservation biology. *Journal of Animal Ecology* **63**, 215-244.
Clarke KR, Warwick RM (1994) 'Change in marine communities: An approach to statistical analysis and interpretation.' (Plymouth Marine Lab: Plymouth)
Clements FE (1916) 'Plant succession: an analysis of the development of vegetation.' (Carnegie Institution of Washington: Washington DC)
Copeland L (2008) Conservation status of *Genoplesium baueri* R. Br. (Orchidaceae) in NSW. Report to the NSW Scientific Committee.
Elith J, Burgman MA, Regan HM (2003) Mapping epistemic uncertainties and vague concepts in predictions of species distribution. *Ecological Modelling* **157**, 313-329.
Faber-Langendoen D, Master LL, Tomaino A, Snow K, Bittman R, Hammerson GA, Heidel B, Nichols J, Ramsay L, Rust S (2007) 'NatureServe Conservation Status Ranking System: Procedures for Automated Rank Assignment.' NatureServe, Arlington, Virginia, U.S.A. [https://transfer.natureserve.org/download/longterm/ERWG/Network_Review/NatureServe%20Conservation%20Status%20Rank%20Procedures&Calculator_14December2007.doc]
Frankham R, Ballou JD, Briscoe DA (2002) 'Introduction to conservation genetics.' (Cambridge University Press: Cambridge)
Gaston K (1992) 'Rarity.' (Chapman and Hall: London).
Gleason HA (1926) The individualistic concept of the plant association. *Bulletin of the Torrey Botanical Club* **53**, 7-26.
Gonzalez A, Lawton JH, Gilbert FS, Blackburn TM, Evans-Freke I (1998) Metapopulation dynamics, abundance, and distribution in a microecosystem. *Science* **281**, 2045-2047.
Hallingbäck T, Hodgetts N, Raeymaekers G, Schumacker R, Sérgio C, Söderström L, Stewart N, Váòa J (2000) Guidelines for application of the 1994 IUCN Red List Categories of threats to bryophytes. Appendix 1 In: T. Hallingbäck and N. Hodgetts (compilers) *Mosses, Liverworts, and Hornworts. Status survey and Conservation Action Plan for Bryophytes.* IUCN/SSC Bryophyte Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.
Hanski, IA, Gilpin ME (1997) 'Metapopulation Biology.' (Academic Press: San Diego, California)
Hunter D, Pietsch R, Marentelli G (2007) 'Recovery actions for the southern and northern corroboree frogs (*Pseudophryne corroboree* and *Pseudophryne pengilleyi*): Annual report and recommendations.' NSW Department of Environment and Climate Change.
IUCN (2001) 'IUCN Red List Categories and Criteria: Version 3.1.' IUCN Species Survival Commission, Gland, Switzerland and Cambridge, U.K. [http://www.iucnredlist.org/info/categories_criteria]
IUCN (2014) 'Guidelines for Using the IUCN Red List Categories and Criteria: Version 11.0.' Standards and Petitions Subcommittee of the IUCN Species Survival Commission. [http://www.iucnredlist.org/documents/RedListGuidelines.pdf]
Kontula T, Raunio A (2009) New method and criteria for national assessments of threatened habitat types. *Biodiversity and Conservation* **18**, 3861-3876.
Keith D (1994) Floristics, structure and diversity of natural vegetation in the O'Hares Creek catchment south of Sydney. *Cunninghamia* **3**, 543-594.

Keith DA (1998) An evaluation and modification of World Conservation Union Red List criteria for classification of extinction risk in vascular plants. *Conservation Biology* **12,** 1076-1090.

Keith DA (2009) The interpretation, assessment and conservation of ecological communities and ecosystems. *Ecological Management and Restoration* **10**, S3-S15.

Keith DA, Bedward M (1999) Vegetation of the South East Forests region, Eden, New South Wales. *Cunninghamia* **6**, 1-218.

Keith DA, Scott J (2005) Native vegetation of coastal floodplains - a diagnosis of the major plant communities in New South Wales. *Pacific Conservation Biology* **11**, 81-104.

Keith DA, Orscheg C, Simpson CC, Clarke PJ, Hughes L, Kennelly SJ, Major RE, Soderquist TR, Wilson AL, Bedward M (2009) A new approach and case study for estimating extent and rates of habitat loss for ecological communities. *Biological Conservation* **142**, 1469-1479.

Keith D. A., Rodríguez J. P., Rodríguez-Clark K. M. *et al*. (2013) Scientific Foundations for an IUCN Red List of Ecosystems. *PLoS ONE* **8,** e62111.

Kendall P, Snelson B (2009) The role of floristic survey data and quantitative analysis in identification and description of ecological communities under threatened species legislation: a case study from north-eastern New South Wales. *Ecological Management and Restoration* **10**, S16-S26.

Kyburg HE, Smokler H (1964) 'Studies in subjective probability.' (Wiley: New York)

Levins R (1970) Extinctions. In 'Some Mathematical Problems in Biology.' (Ed M Gesternhaber) pp. 77-107. (American Mathematical Society: Providence, Rhode Island)

Levins R (1969) Some demographic and genetic consequences of environmental heterogeneity for biological control. *Bulletin of the Entomological Society of America* **15**, 237-240.

Mace GM, Lande R (1991) Assessing extinction threats: towards a re-evaluation of IUCN threatened species categories. *Conservation Biology* **5**, 148-157.

Mace GM, Collar N, Gaston KJ, Hilton-Taylor C, Akçakaya HR, Leader-Williams N, Milner-Gulland EJ, Stuart SN (2008) Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology* **22**, 1424-1442.

Mackenzie BDE, Keith DA (2007) 'Assessment of Woodland Assemblages dominated by *Callitris* for listing as a threatened ecological community under the Threatened Species Conservation Act.' Report to the NSW Scientific Committee.

Mackenzie BDE, Keith DA (2009) Adaptive management for conservation of an endangered population of Black Cypress Pine, *Callitris endlicheri. Ecological Management and Restoration* **10**, S129-S135.

MacMillan, D.C., Marshall, K. (2006). The Delphi process--an expert-based approach to ecological modelling in data-poor environments. *Animal Conservation* **9**, 11-19.

Master LL, Faber-Langendoen D, Bittman R, Hammerson GA, Heidel B, Ramsay L, Tomaino A (2007) 'NatureServe Conservation Status Factors.' NatureServe, Arlington, Virginia, U.S.A.

McGarigal K (2002) 'Fragstats. Spatial Pattern Analysis Program for Categorical Maps.' University of Massachusetts, Amherst. http://www.umass.edu/landeco/research/fragstats/fragstats.html

Moilanen A, Franco AMA, Early R, Fox R, Wintle B, Thomas CD (2005) Prioritising multiple use landscapes for conservation: methods for large multi species planning problems. *Proceedings of the Royal Society B: Biological Sciences* **272**, 1885-1891.

Mueller-Dombois D, Ellenberg H (1974) 'Aims and Methods of Vegetation Ecology.' (John Wiley and Sons: New York)

Nicholson E, Keith DA, Wilcove DS (2009) Assessing the conservation status of ecological communities. *Conservation Biology* **23**, 259-274.

NSW Scientific Committee, Mackenzie BDE (2008) 'Floristic analysis of Coolibah – Black Box Woodland.' NSW Scientific Committee, Sydney.

Orscheg C, Ooi M, Keith D (2006) 'The ecological relationships and conservation status of Tall Apple-Blackbutt Shale Forest.' Report to the NSW Scientific Committee, Sydney.

Osborne WS, Hunter DA, Hollis GJ (1999) Population declines and range contraction in Australian alpine frogs. In 'Declines and Disappearances of Australian Frogs'. (Ed A. Campbell) pp 145-157. (Environment Australia: Canberra)

Possingham H, Ball I, Andelman S (2000) Mathematical methods for identifying representative reserve networks. In: 'Quantitative methods for conservation biology' (Eds S Ferson and MA Burgman) pp291-305. (Springer-Verlag: New York)

Preston BJ, Adam P (2004a) Describing and listing threatened ecological communities under the Threatened Species Conservation Act 1995 (NSW). Part 1. *Environmental Planning and Law Journal* **21**, 250-263.

Preston BJ, Adam P (2004b) Describing and listing threatened ecological communities under the Threatened Species Conservation Act 1995 (NSW). Part 2. *Environmental Planning and Law Journal* **21**, 372-390.

Regan HM, Colvan M, Burgman MA (2002) A taxonomy of and treatment of uncertainty for ecology and conservation biology. *Ecological Applications* **12**, 618-628.

Rodriguez JP (2002) Range contraction in declining North American bird populations. *Ecological Applications* **12**, 238-248.

Rodriguez JP, Balch JK, Rodriguez-Clark KM (2007) Assessing extinction risk in the absence of species-level data: quantitative criteria for terrestrial ecosystems. *Biodiversity and Conservation* **16**, 183-209.

Rodwell JS, Schamineé JHJ, Mucina J, Pignatti S, Dring J, Moss D (2002) 'The diversity of European vegetation: an overview of phytosociological alliances and their relationships to EUNIS habitats, Report LC-EMV nr. 2002/054.' Wageningen, Netherlands.

Scott J. (2009) Additional information for *Cynanchum elegans* (Benth.) Domin. Report to the Scientific Committee.

SEWPaC (2012) Interim Biogeographic Regionalisation for Australia, Version 7. Department of Sustainability, Environment, Water, Population and Communities. http://www.environment.gov.au/parks/nrs/science/bioregion-framework/ibra/maps.html

Sivertsen DP, Metcalfe L (1995) Natural vegetation of the southern wheatbelt. *Cunninghamia* **4**, 103-128.

Soderquist T, Irvin M (2008) 'Pattern analysis of species composition and site associations of Old Man Saltbush vegetation communities.' Report to the NSW Scientific Committee, Hurstville.

Thackway R, Creswell ID (1995) (Eds) 'An interim biogeographic regionalisation of Australia: a framework for establishing the national system of reserves.' (Australian Nature Conservation Agency: Canberra).

Tindall D, Pennay C, Tozer MG, Turner K, Keith DA (2004) 'Native Vegetation Map Report Series. No. 4. Araluen, Batemans Bay, Braidwood, Burragorang, Goulburn, Jervis Bay, Katoomba, Kiama, Moss Vale, Penrith, Port Hacking, Sydney, Taralga, Ulladulla,Wollongong.' NSW Department of Environment and Conservation and NSW Department of Infrastructure, Planning and Natural Resources, Sydney.

Toro MA, Caballero A (2005) Characterization and conservation of genetic diversity in subdivided populations. *Philosophical Transactions of the Royal Society London Biological Sciences* **360**, 1367–1378.

Tozer M (2003) The native vegetation of the Cumberland Plain, western Sydney: systematic classification and field identification of communities. *Cunninghamia* **8**, 1-75.

Tozer MG, Turner K, Keith DA, Tindall D, Pennay C, Simpson C, Mackenzie B (in press) Native vegetation of southeast NSW: a revised classification and map for the coast and eastern tablelands. *Cunninghamia* **11,** 359-406**.**

Whittaker RH (1975) 'Communities and ecosystems.' (Macmillan: New York)