Guidelines for Using the IUCN Red List Categories and Criteria

Version 16

(March 2024)

Prepared by the Standards and Petitions Committee of the IUCN Species Survival Commission.

Citation: IUCN Standards and Petitions Committee. 2024. Guidelines for Using the IUCN Red List Categories and Criteria. Version 16. Prepared by the Standards and Petitions Committee. Downloadable from https://www.iucnredlist.org/documents/RedListGuidelines.pdf.

Contents

1. Introduction

The IUCN Red List Categories and Criteria were first published in 1994 following six years of research and broad consultation (IUCN 1994). The 1994 IUCN Categories and Criteria were developed to improve objectivity and transparency in assessing the conservation status of species, and therefore to improve consistency and understanding among users. The 1994 categories and criteria were applied to a large number of species in compiling the 1996 Red List of Threatened Animals. The assessment of many species for the 1996 Red List drew attention to certain areas of difficulty, which led IUCN to initiate a review of the 1994 categories and criteria, which was undertaken during 1998 to 1999. This review was completed and the IUCN Red List Categories and Criteria (version 3.1) are now published (IUCN 2001, 2012b).

This document provides guidelines to the application of version 3.1 of the categories and criteria, and in so doing addresses many of the issues raised in the process of reviewing the 1994 categories and criteria. This document explains how the criteria should be applied to determine whether a taxon belongs in a category of threat, and gives examples from different taxonomic groups to illustrate the application of the criteria. These guidelines also provide detailed explanations of the definitions of the many terms used in the criteria. The guidelines should be used in conjunction with the official IUCN Red List Categories and Criteria booklet (IUCN 2001, 2012b).

We expect to review and update these guidelines periodically, and input from all users of the IUCN Red List Categories and Criteria are welcome. We especially welcome IUCN Specialist Groups and Red List Authorities to submit examples that are illustrative of these guidelines. We expect that the changes to these guidelines will be mostly additions of detail and not changes in substance. In addition, we do not expect the IUCN Red List Criteria to be revised in the near future, because a stable system is necessary to allow comparisons over time.

2. An Outline of the Red List Categories and Criteria

2.1 Taxonomic level and scope of the categorization process

2.1.1 Taxonomic scale of categorization

The criteria may be applied to any taxonomic unit at or below the species level. In these guidelines, the terms '*taxon*' and 'taxa' are used to represent species or lower taxonomic levels, including forms that are not yet fully described, and excluding humans. There is sufficient range among the different criteria to enable appropriate listing of taxa from the complete taxonomic spectrum, with the exception of micro-organisms. In presenting the results of applying criteria, the taxonomic unit used (species, subspecies, etc.) should be specified. It should be noted that taxa below the rank of variety (e.g., forma, morph, cultivar), are NOT included on the IUCN Red List, with the exception of assessments of subpopulations. An assessment of the full species is required before assessments of taxa below the species level (subspecies, variety or subpopulation) can be included on the IUCN Red List.

Subpopulations: If a subpopulation assessed under the criteria is not isolated (i.e., if it may be exchanging individuals with other subpopulations), its assessments must follow the regional guidelines (IUCN 2003, 2012a). In addition, it must be a biological subpopulation (i.e., not defined by political or national boundaries). Although the regional guidelines can in principle be applied at any geographical scale, application within very small geographical areas is strongly discouraged. The smaller the subpopulation as a proportion of the global population of the species, the more often the subpopulation will exchange individuals with other subpopulations. Therefore, the assessment of extinction risk based on the criteria would become more unreliable (IUCN 2003, 2012a). See also *Geographical scale of categorization* below.

Newly described species: The inclusion of newly described species on the IUCN Red List is addressed on a case-by-case basis. The designated IUCN Red List Authority and/or IUCN Global Species Programme staff (including staff from partner institutions working on Global Species Assessment projects) will consult with relevant experts to ascertain how widely accepted these are.

Undescribed species: The listing of undescribed species on the IUCN Red List is discouraged, but in exceptional circumstances these may be included. There must be a clear conservation benefit to justify the inclusion of such listings, or in the case of projects to completely assess a taxonomic group, undescribed species that are listed as Least Concern (LC) or Data Deficient (DD) may only be included if there is clear evidence that work is underway to describe the species concerned and that the new species will be widely accepted. The new species description should be published within four years of an undescribed species being included on the IUCN Red List; if it is not published or is not in press after that time, the assessment will be removed. For an undescribed species to be included on the IUCN Red List the following conditions must be met:

- There must be general agreement that the undescribed form is a clearly circumscribed species.
- There must be a clear indication that work is underway to describe the species (e.g., a draft manuscript in preparation or a paper with the new description already submitted for publication).
- Clear distribution information must be provided.
- Listing the undescribed species will potentially aid in its conservation.
- Specimen reference numbers (voucher collection details) must be provided to enable the species to be traced without confusion.
- The museum, herbarium or other institution holding the collection/s and the individual/s responsible for the proposal must be identified.
- Undescribed species sometimes have a local common name, if so, this should be provided, but if not, a recognizable common name should be coined, so that it can be used to clearly indicate the identity of this taxon without any implication about scientific validity.

Undescribed species are presented on the IUCN Red List by using the generic name and the abbreviation sp. or sp. nov., sometimes followed by a provisional name in inverted commas (e.g., *Philautus* sp. nov. 'Kalpatta'). Voucher collection details (collector's name, specimen number and institution where housed) must be provided so that they can be cited under the Taxonomic Notes section of the species' account on the Red List web site. Instances may arise where there are valid reasons for voucher collection details to be withheld. If this is clearly indicated and justified by the assessor(s) concerned, the voucher information will be suppressed from the public version of the species account. However, in such cases the voucher information must still be supplied for the assessment to be accepted, and this information will be held in confidence.

Undescribed species assessed as Least Concern (LC) or Data Deficient (DD) are not included on the IUCN Red List as there is little conservation benefit to such listings.

Undescribed infraspecific taxa are not included on the IUCN Red List.

In summary, assessments of the following taxa may be included on the IUCN Red List
- Species
- Subspecies
- Varieties (only for plants)
- Subpopulations (provided certain conditions as described above are met)
- Undescribed species (provided certain conditions as described above are met, and they are not listed as LC or DD)

Assessments of the following taxa may NOT be included on the IUCN Red List
- Taxa assessed locally, nationally or regionally unless they are global or subpopulation assessments (see "*Subpopulation*" above, and section 2.1.2)
- Hybrids (except for apomictic plant hybrids, which are treated as 'species')
- Infraspecific ranks such as formas, morphs, subvarieties, varieties of subspecies, cultivars, etc.
- Domesticated taxa (in the case where a taxon comprises both domesticated and wild individuals, only the wild population may be assessed and included; feral animals derived from a domesticated source should not be included)
- Taxa known to have gone Extinct before 1500 CE
- Undescribed species assessed as Data Deficient or Least Concern (except in the case of complete global assessments for a taxonomic group, see above)
- Undescribed infraspecific taxa
- Assessments of higher taxa (i.e., above the species level).

2.1.2 Geographical scale of categorization

The IUCN criteria are designed for global taxon assessments. However, many people are interested in applying them to subsets of global data, especially at regional, national or local levels. To do this it is important to refer to guidelines prepared by the IUCN SSC Regional Applications Working Group (e.g., Gärdenfors *et al.* 2001; IUCN 2003, 2012a; Miller *et al.* 2007). When applied at national or regional levels it must be recognized that a global category may not be the same as a national or regional category for a particular taxon. For example, taxa classified as Least Concern globally might be Critically Endangered within a particular region where numbers are very small or declining, perhaps only because they are at the margins of their global range. Conversely, taxa classified as Vulnerable on the basis of their global declines in numbers or range might, within a particular region where their populations are stable, not even nearly meet the criteria for Vulnerable, i.e. be Least Concern. Although this appears illogical, it is a result of the structure of the criteria. When such a situation occurs, interactions among sub-units should be carefully considered when planning conservation actions.

Although the criteria (along with regional guidelines; IUCN 2012a) may be applied at any geographical scale, application within very restricted geographical areas is strongly discouraged (IUCN 2012a). In a small region, a wide-ranging taxon will frequently exchange individuals with neighbouring regions, leading to unreliable assessments (IUCN 2012a).

It is also important to note that in any regional or national applications of the criteria, an assessment of taxa endemic to that region or nation will be a global assessment; in these cases great care must be taken to check that a global assessment has not already been undertaken by an IUCN SSC Red List Authority (RLA), and that the final categorization is agreed with the relevant RLA; see the regional guidelines for more details (IUCN 2003, 2012a).

2.1.3 Introduced taxa and subpopulations

In addition to wild subpopulations (see section 2.1.4) inside the natural range of a taxon, the categorization process should also be applied to wild subpopulations resulting from introductions outside the natural range, if all of the following conditions are met:

(a) The known or likely intent of the introduction was to reduce the extinction risk of the taxon being introduced. In cases where the intent is unclear, the assessors should weigh the available evidence to determine the most likely intent.

(b) The introduced subpopulation is geographically close to the natural range of the taxon. What is considered to be geographically close enough should be determined by the assessor, considering factors such as the area of the natural range, the nature of the landscape separating the natural and the introduced range, and whether the taxon could have dispersed to the introduced range without the effects of human impacts such as habitat loss and fragmentation. For example, an introduced subpopulation in a continent distant from the natural range would not qualify. On the other hand, most introduced subpopulations within the same ecoregion as the natural range would qualify.

(c) The introduced subpopulation has produced viable offspring (i.e., offspring that have reached maturity or are likely to do so).

(d) At least five years have passed since the introduction.

In cases where such introduced subpopulations are included in the assessment, assessors must state and justify their inclusion in the assessment supporting documentation.

In some cases, taxa have successfully expanded their natural ranges into urban or semi-urban areas, e.g., primates, foxes and some birds. In these instances urban areas should be considered as part of the natural range, as the taxa have not been introduced.

In addition to taxa within their natural range and subpopulations resulting from introductions outside the taxon’s natural range that conform to the conditions above (also referred to as "benign introductions"), the criteria should also be applied to self-sustaining translocated or re-introduced subpopulations (within the taxon’s natural range), regardless of the original goal of such translocations or re-introductions. In such cases, the listing should indicate whether all or part of the assessed population has been introduced. Also, in such cases, conditions (a) and (b) above are not relevant, but conditions (c) and (d) must be met.

2.1.4 Managed subpopulations

The IUCN Red List assessment should only be applied to wild populations. There is a continuum of management intensities, from captive populations in zoos, aquaria and botanical gardens to populations not benefiting from any conservation measure. Clearly, captive populations are not considered "wild" and would be excluded from a Red List assessment (i.e., data from such populations are not considered in determining the species' status, except for EW). On the other hand, subpopulations of many species are dependent on conservation measures (such as protected areas) that are largely directed at mitigating human impacts. Such subpopulations are generally considered "wild", and the data from such subpopulations are used in Red List assessments. In between these are subpopulations that are managed at moderate levels of intensity (Redford *et al.* 2011). For these subpopulations, the definition of "wild" may be based on the intensity of management, and the expected viability of the subpopulation without the management.

Subpopulations dependent on direct intervention are not considered wild, if they would go extinct within 10 years without “intensive” management such as:
- providing most of the food needs of most individuals in the subpopulation;
- regularly supplementing the population from captive stock to prevent imminent extinction;
- breeding manipulations, such as cross-fostering and down-brooding (i.e., removing extra chicks from large broods and giving to foster parents); or
- providing ongoing intensive veterinary care to most individuals.

Managed subpopulations are considered wild if the management is for counter-acting the effects of human threats, such as:
- protected areas;
- anti-poaching patrols;
- providing artificial shelters (e.g., nest boxes for birds, roosting sites for bats);
- providing preventative treatments against disease outbreaks;
- preventing natural vegetation succession to maintain the species' habitat;
- translocating individuals between existing subpopulations (also see section 2.1.3);
- control measures against non-native competitors or predators, including the establishment of exclusion fences, such as those used to keep out invasive predators;
- control measures against native competitors or predators if such species have increased because of human activities (e.g., removing cowbird which have increased because of habitat fragmentation); or
- occasionally supplementing the population from captive stock to increase genetic variability.

This delineation of "wild" from "not wild" roughly corresponds to the difference between "lightly managed species" and "intensively managed species" as defined by Redford *et al.* (2011).

2.2 Nature of the categories

There are nine clearly defined categories into which every taxon in the world (excluding micro-organisms) can be classified (Figure 2.1). Complete definitions of the categories are given in Box 2.1. The first two categories in Figure 2.1 are relatively self-explanatory. **Extinct** means that there is no reasonable doubt that the last individual has died. **Extinct in the Wild** means that the taxon is extinct in its natural habitat (see Introduced taxa above). The following three categories, **Critically Endangered, Endangered** and **Vulnerable**, are assigned to taxa on the basis of quantitative criteria that are designed to reflect varying degrees of threat of extinction; taxa in any of these three categories are collectively referred to as ‘threatened’. These criteria will be discussed further in the next section. The category **Near Threatened** is applied to taxa that do not qualify as threatened now, but may be close to qualifying as threatened, and to taxa that do not currently meet the criteria for a threatened category, but are likely to do so if ongoing conservation actions abate or cease.

The category **Least Concern** is applied to taxa that do not qualify (and are not close to qualifying) as threatened or Near Threatened. It is important to emphasize that "least concern" simply means that, in terms of extinction risk, these species are of lesser concern than species in other threat categories. It does not imply that these species are of no conservation concern.

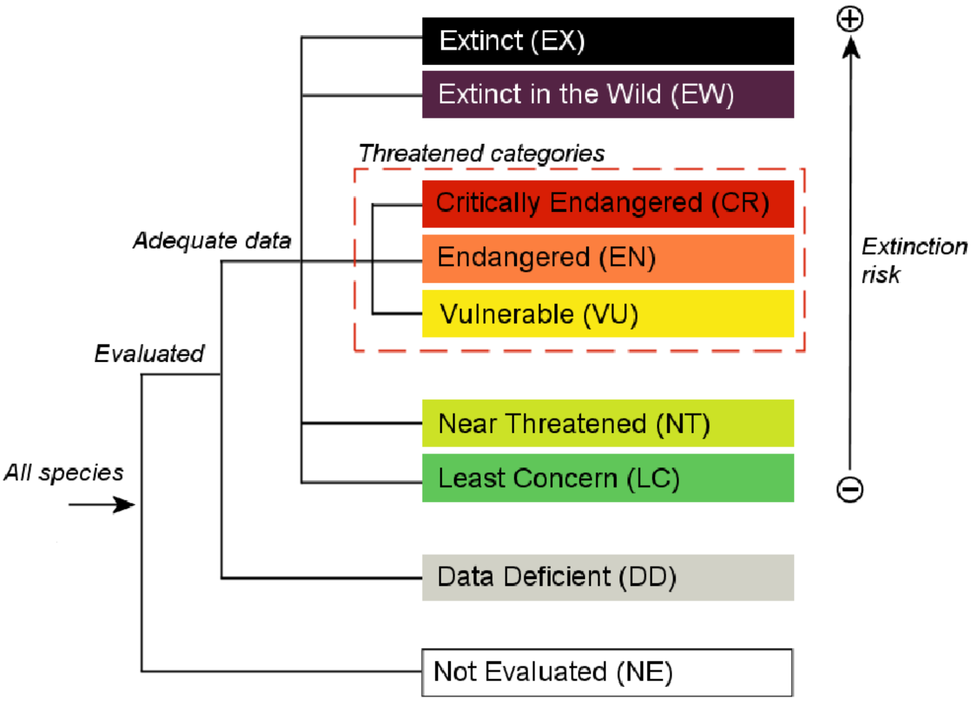


Figure 2.1. Structure of the IUCN Red List Categories

Box 2.1. **The IUCN Red List Categories**

EXTINCT (EX)
A taxon is Extinct when there is no reasonable doubt that the last individual has died. A taxon is presumed Extinct when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon’s life cycles and life form.

EXTINCT IN THE WILD (EW)
A taxon is Extinct in the Wild when it is known only to survive in cultivation, in captivity or as a naturalized population (or populations) well outside the past range. A taxon is presumed Extinct in the Wild when exhaustive surveys in known and/or expected habitat, at appropriate times (diurnal, seasonal, annual), throughout its historic range have failed to record an individual. Surveys should be over a time frame appropriate to the taxon's life cycle and life form.

CRITICALLY ENDANGERED (CR)
A taxon is Critically Endangered when the best available evidence indicates that it meets any of the criteria A to E for Critically Endangered, and it is therefore considered to be facing an extremely high risk of extinction in the wild.

ENDANGERED (EN)
A taxon is Endangered when the best available evidence indicates that it meets any of the criteria A to E for Endangered, and it is therefore considered to be facing a very high risk of extinction in the wild.

VULNERABLE (VU)
A taxon is Vulnerable when the best available evidence indicates that it meets any of the criteria A to E for Vulnerable, and it is therefore considered to be facing a high risk of extinction in the wild.

NEAR THREATENED (NT)
A taxon is Near Threatened when it has been evaluated against the criteria but does not qualify for Critically Endangered, Endangered or Vulnerable now, but is close to qualifying for or is likely to qualify for a threatened category in the near future.

LEAST CONCERN (LC)
A taxon is Least Concern when it has been evaluated against the criteria and does not qualify for Critically Endangered, Endangered, Vulnerable or Near Threatened. Widespread and abundant taxa are often included in this category.

DATA DEFICIENT (DD)
A taxon is Data Deficient when there is inadequate information to make a direct, or indirect, assessment of its risk of extinction based on its distribution and/or population status. A taxon in this category may be well studied, and its biology well known, but appropriate data on abundance and/or distribution are lacking. Data Deficient is therefore not a category of threat. Listing of taxa in this category indicates that more information is required and acknowledges the possibility that future research will show that threatened classification is appropriate. It is important to make positive use of whatever data are available. In many cases great care should be exercised in choosing between DD and a threatened status. If the range of a taxon is suspected to be relatively circumscribed, or a considerable period of time has elapsed since the last record of the taxon, threatened status may well be justified.

NOT EVALUATED (NE)
A taxon is Not Evaluated when it is has not yet been evaluated against the criteria.

The remaining two categories do not reflect the threat status of taxa. The category **Data Deficient** highlights taxa for which sufficient information is lacking to make a sound status assessment. The inclination to assess taxa as Data Deficient may be very strong; it should be emphasized that assessors must use all data available in full when making a Red List assessment. Precise information on scarce taxa is usually lacking, and although the criteria are highly quantitative and defined, one can use projections, assumptions and inferences (as long as they are explicitly stated and clearly justified) to place a taxon in the appropriate category. Since Data Deficient is not a category of threat, taxa placed in this category may not be so obviously targets for conservation action, although their needs might be very great. Assessors should use whatever information is available and relevant to make assessments and place taxa into the Data Deficient category only when there is really no alternative. Guidance on handling uncertainty is especially relevant in the case of poorly known taxa (see section 3.2). The category **Not Evaluated** applies to taxa that have not yet been evaluated against the Red List Criteria.

The term “red-listed” is not defined in IUCN (2001) and is not used in this document owing to ambiguity as to whether this includes Least Concern species or not, given that species assessed as Least Concern are included on the IUCN Red List. To refer to species that have assessments on the IUCN Red List, the phrase “assessed for the IUCN Red List” can be used. To refer to species that are Extinct in the Wild, threatened and Near Threatened (i.e., EW, CR, EN, VU, NT), the phrase “species of elevated conservation concern” may be used.

2.2.1 Transfer between categories

The following rules govern the movement of taxa between categories:

A. A taxon may be moved from a category of higher threat to a category of lower threat if and when none of the criteria of the higher category has been met for five years or more (i.e., if the taxon has qualified for a lower threat category for at least five years, regardless of when the previous assessment was published). Thus, the 5-year period commences when the data show that the taxon no longer meets the criteria for the category in which it is currently listed; this is not necessarily the date of the previous assessment. If it is not possible to identify the year in which the taxon qualified for the lower threat category, then the current assessment year is used as the start of the 5-year period. However, if the taxon is being moved from EW as a result of the establishment of a re-introduced population, this period must be five years or until viable offspring are produced, whichever is the longer.

B. If the original classification is found to have been erroneous, the taxon may be transferred to the appropriate category or removed from the threatened categories altogether, without delay. However, in this case, the taxon should be re-evaluated against all the criteria to clarify its status.

C. Transfer from categories of lower to higher risk should be made without delay.

D. The reason for a transfer between categories must be documented as one of the following:

Genuine change:

Genuine (recent). The change in category is the result of a genuine status change that has taken place since the previous assessment. For example, the change is due to an increase in the rate of decline, a decrease in population or range size or habitat, or declines in these for the first time (owing to increasing/new threats) and therefore new thresholds are met relating to the IUCN Red List Criteria.

Genuine (since first assessment). This applies to taxa assessed at least three times and is used to assign genuine category changes to the appropriate time period to calculate the Red List Index. The change in category is the result of a genuine status change that took place prior to the last assessment, but since the first assessment and that has only just been detected owing to new information or new documentation. If this new information had been available earlier, the new category would have been assigned during the previous assessment(s). When this code is used, the appropriate time period (between previous assessments) in which the status change occurred needs to be indicated. [See example below]

Nongenuine change:

Criteria revision. The change in category is the result of the revision of the IUCN Red List Criteria (e.g., 1994 v. 2001 versions). These largely relate to criteria A2, A3, A4, D2 and the removal of the 'Conservation Dependent' category.

New information. The change in category is the result of better knowledge about the taxon, e.g. owing to new or newly synthesized information about the status of the taxon (e.g., better estimates for population size, range size or rate of decline).

Taxonomy. The new category is different from the previous owing to a taxonomic change adopted during the period since the previous assessment. Such changes include: *newly split* (e.g., the taxon is newly elevated to species level), *newly lumped* (the taxon is recognized following lumping of two previously recognized taxa), and *no longer valid/recognized* (either the taxon is no longer valid e.g. because it is now considered to be a hybrid or variant, form or subspecies of another species, or the previously recognized taxon differs from a currently recognized one as a result of a split or lump).

Misinterpretation of the criteria ('Knowledge of criteria' in SIS). The previous category was applied in error because the assessor(s) misunderstood the IUCN Red List Criteria.

Incorrect data. The previous category was applied in error because incorrect data were used (e.g., the data referred to a different taxon).

Other. The change in category is the result of other reasons not easily covered by the above, and/or requires further explanation. Examples include change in assessor's attitude to risk and uncertainty (as defined in section 3.2.3) and changes in this guidelines document.

No change: When there is no change in category, one of the following must be specified.

Same category and criteria. In other words, no change in the listing.

Same category but change in criteria. For example, a change from EN A2 to EN D.

Determining the appropriate reason for change often requires careful consideration. Many category changes result from a combination of improved knowledge and some element of genuine deterioration or improvement in status. In such cases, "genuine" should only be assigned if the amount of genuine change (e.g., population size change, decline rate change, range size change, etc.) is sufficient on its own to cross the relevant Red List Category threshold. Genuine and non-genuine reasons for change should never be coded at the same time.

> e.g. Species A previously qualified as Endangered (D) with a population estimated to be 150 individuals; it is reassessed as Vulnerable (D1) because its population is now estimated to number 400 individuals; the new estimate is partly a result of the discovery of a new stable subpopulation numbering 50 individuals, and partly because the previously known subpopulation increased from 150 to 350 individuals. The genuine increase is sufficient to have taken the total population over the threshold for Vulnerable, so the category change is coded as Genuine (recent), and Nongenuine (New information) should not be coded as the reason for change in this case.

> e.g. Species B previously qualified as Endangered (D) with a population estimated to be 150 individuals; it is reassessed as Vulnerable (D1) because its population is now estimated to number 400 individuals; the new estimate is partly a result of the discovery of a new stable subpopulation numbering 200 individuals, and partly because the previously known subpopulation increased from 150 to 200 individuals. The genuine increase in this case is insufficient to have taken the total population over the threshold for Vulnerable, (it should have qualified as Vulnerable in the previous assessment also) so the reason for the category change is coded as Nongenuine (New information), and not as Genuine (recent) in this case.

In cases where a category change results from a combination of taxonomic splitting and genuine change, the change should be coded as Taxonomy if the currently recognised taxon would have qualified for the higher or lower Red List category previously; otherwise, it should be coded Genuine (recent).

> e.g. Species C previously qualified as VU D1 with a total population estimated to number 600 individuals. It is then split into species D (540 individuals and stable) and species E (now only 40 individuals, having declined from 60 individuals in the previous assessment). This category change for species E (previously 'Not Recognized' but now CR C1) should be coded as Genuine (recent) because it would have qualified as Endangered D in the previous assessment. (Species D would be classified as VU D1)

All Genuine (recent) or Genuine (since first assessment) category changes should be supported with appropriate notes to justify why the change is coded as genuine.

> e.g. Mauritius Kestrel *Falco punctatus* was downlisted from CR in 1988 to EN in 1994; this was coded as Genuine (recent) with the note: "Population increased from eight pairs in 1987–1988 to 56–68 pairs in 1994 as a result of a ban on hunting".

> e.g. Montserrat Oriole *Icterus oberi* was uplisted from NT in 1994 to CR in 2000; this was coded as Genuine (recent) with the note: "In the early 1990s, this species occurred throughout the three main forested hill ranges on the island, but volcanic eruptions in 1995–1997 destroyed two-thirds of remaining habitat. Recent evidence suggests that the decline may now have halted, and the population is estimated at c.100–400 pairs".

> e.g. Ethiopian Bush-crow *Zavattariornis stresemanni* was uplisted from Vulnerable to Endangered in 2005. This category change was coded as Genuine (since first assessment), with the genuine change assigned to the 1994–2000 period, and the note "Encounter rates declined 80% between 1989 and 2003. Assuming declines began in 1989, the cumulative decline would have exceeded 50% over 10 years for the first time during the period 1994–2000".

2.3 Nature of the criteria

There are five quantitative criteria that are used to determine whether a taxon is threatened or not, and if threatened, which category of threat it belongs in (Critically Endangered, Endangered or Vulnerable) (Table 2.1). These criteria are based around the biological indicators of populations that are threatened with extinction, such as rapid population decline or very small population size. Most of the criteria also include subcriteria that must be used to justify more specifically the listing of a taxon under a particular category. For example, a taxon listed as "Vulnerable C2a(ii)" has been placed in the Vulnerable category because its population is fewer than 10,000 mature individuals (criterion C) and the population is undergoing a continuing decline and all its mature individuals are in one subpopulation (subcriterion a(ii) of criterion C2).

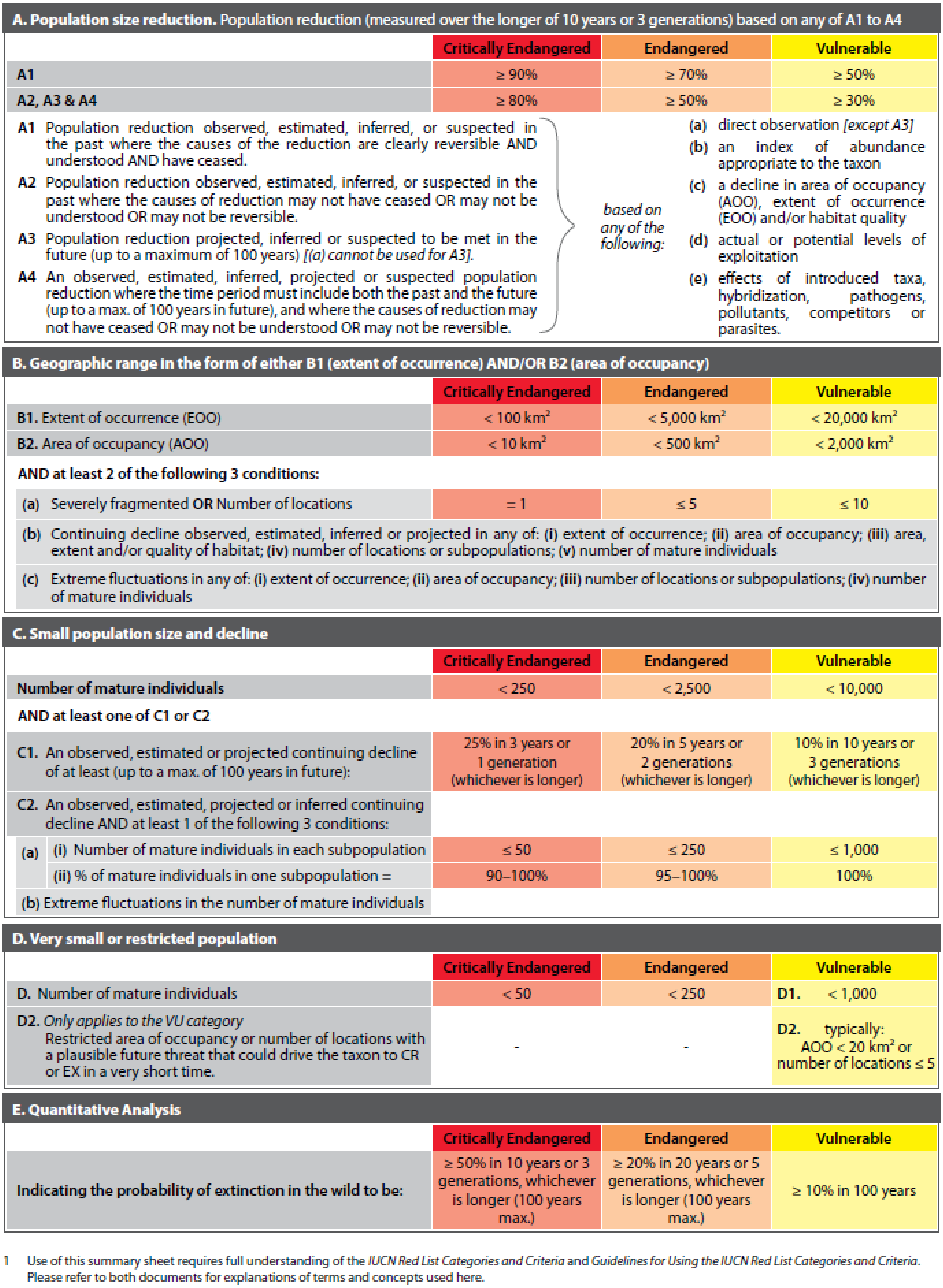
The five criteria are:

A. Population size reduction (past, present and/or projected)
B. Geographic range size, and fragmentation, few locations, decline or fluctuations
C. Small and declining population size and fragmentation, fluctuations, or few subpopulations
D. Very small population or very restricted distribution
E. Quantitative analysis of extinction risk (e.g., Population Viability Analysis)

To list a particular taxon in any of the categories of threat, only one of the criteria, A, B, C, D, or E needs to be met. However, a taxon should be assessed against as many criteria as available data permit, and the listing should be annotated by as many criteria as are applicable for a specific category of threat. For example, Critically Endangered: A2cd; B1ab(iv,v); C2a(i). Only the criteria for the highest category of threat that the taxon qualifies for should be listed. For example, if a taxon qualifies for criteria A, B, and C in the Vulnerable and Endangered category and only criterion A in the Critically Endangered category, then only the criterion A met in the Critically Endangered category should be listed (the highest category of threat). Assessors are encouraged to document criteria under which a species meets lower threat categories, because such information is critical to recovery planning.

Although the criteria for each of the categories of threat are based on quantitative thresholds, the system remains relatively flexible to ensure that taxa for which there is very little information can also be assessed. This has been achieved by incorporating inference, suspicion and projection into the assessment process. Therefore, the person conducting an assessment is expected to use the best available information in combination with inference, suspicion and projection to test a taxon against the criteria. However, if inference, suspicion and projection are used, the assumptions made must be documented. If there is any reasonable concern that a taxon is threatened with extinction in the near future, it should qualify for the criteria of one of the categories of threat.

Table 2.1. Summary of the five criteria (A-E) used to evaluate if a taxon belongs in a threatened category (Critically Endangered, Endangered or Vulnerable).



A. Population size reduction. Population reduction (measured over the longer of 10 years or 3 generations) based on any of A1 to A4

	Critically Endangered	Endangered	Vulnerable
A1	≥ 90%	≥ 70%	≥ 50%
A2, A3 & A4	≥ 80%	≥ 50%	≥ 30%

A1 Population reduction observed, estimated, inferred, or suspected in the past where the causes of the reduction are clearly reversible AND understood AND have ceased.

A2 Population reduction observed, estimated, inferred, or suspected in the past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

A3 Population reduction projected, inferred or suspected to be met in the future (up to a maximum of 100 years) *[(a) cannot be used for A3]*.

A4 An observed, estimated, inferred, projected or suspected population reduction where the time period must include both the past and the future (up to a max. of 100 years in future), and where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

based on any of the following:

(a) direct observation *[except A3]*

(b) an index of abundance appropriate to the taxon

(c) a decline in area of occupancy (AOO), extent of occurrence (EOO) and/or habitat quality

(d) actual or potential levels of exploitation

(e) effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites.

B. Geographic range in the form of either B1 (extent of occurrence) AND/OR B2 (area of occupancy)

	Critically Endangered	Endangered	Vulnerable
B1. Extent of occurrence (EOO)	< 100 km^2	< 5,000 km^2	< 20,000 km^2
B2. Area of occupancy (AOO)	< 10 km^2	< 500 km^2	< 2,000 km^2
AND at least 2 of the following 3 conditions:			
(a) Severely fragmented **OR** Number of locations	= 1	≤ 5	≤ 10

(b) Continuing decline observed, estimated, inferred or projected in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** area, extent and/or quality of habitat; **(iv)** number of locations or subpopulations; **(v)** number of mature individuals

(c) Extreme fluctuations in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** number of locations or subpopulations; **(iv)** number of mature individuals

C. Small population size and decline

	Critically Endangered	Endangered	Vulnerable
Number of mature individuals	< 250	< 2,500	< 10,000
AND at least one of C1 or C2			
C1. An observed, estimated or projected continuing decline of at least (up to a max. of 100 years in future):	25% in 3 years or 1 generation (whichever is longer)	20% in 5 years or 2 generations (whichever is longer)	10% in 10 years or 3 generations (whichever is longer)
C2. An observed, estimated, projected or inferred continuing decline AND at least 1 of the following 3 conditions:			
(a) **(i)** Number of mature individuals in each subpopulation	≤ 50	≤ 250	≤ 1,000
(ii) % of mature individuals in one subpopulation =	90–100%	95–100%	100%
(b) Extreme fluctuations in the number of mature individuals			

D. Very small or restricted population

	Critically Endangered	Endangered	Vulnerable
D. Number of mature individuals	< 50	< 250	**D1.** < 1,000
D2. *Only applies to the VU category* Restricted area of occupancy or number of locations with a plausible future threat that could drive the taxon to CR or EX in a very short time.	-	-	**D2.** typically: AOO < 20 km^2 or number of locations ≤ 5

E. Quantitative Analysis

	Critically Endangered	Endangered	Vulnerable
Indicating the probability of extinction in the wild to be:	≥ 50% in 10 years or 3 generations, whichever is longer (100 years max.)	≥ 20% in 20 years or 5 generations, whichever is longer (100 years max.)	≥ 10% in 100 years

1 Use of this summary sheet requires full understanding of the *IUCN Red List Categories and Criteria* and *Guidelines for Using the IUCN Red List Categories and Criteria*. Please refer to both documents for explanations of terms and concepts used here.

Listing under the highest category of threat (instead of, for instance, averaging extinction risk across the five criteria) ensures a more precautionary approach to making urgent decisions based on limited information. It also bases the overall risk assessment on the factors that are most critical to species persistence. This approach is akin to emergency room doctors focusing their assessment of patients on the most severe symptoms, instead of an average, where the positive symptoms cancel out the negative ones (Collen *et al.* 2016). The five criteria (A-E) are derived from a wide review aimed at detecting risk factors across the broad range of organisms and the diverse life histories they exhibit. The criteria are aimed at detecting symptoms of endangerment rather than causes. Consequently, the criteria are applicable to any threatening process that results in symptoms such as past and future population decline, small population sizes, and small geographic distributions. A taxon may be classified as threatened even if a threatening process cannot be identified. Regardless of the nature of threats, assessments must follow IUCN (2001, 2012b) and these guidelines to ensure valid application of the criteria. However, different threats, especially new threats or poorly understood processes such as global climate change may require further guidance in the application of definitions and criteria. Section 12 provides guidance specific to different threats.

Some studies suggest that when taxa are evaluated under all five criteria, there is a tendency for them to be listed under criteria A to D rather than under E. There are several possible reasons for this. First, a reliable assessment under criterion E generally requires more data and analysis, and in practice the process may often be incomplete. Second, even if each criterion on average corresponds to an identical risk of extinction, the probability that a specific species meets at least one of four criteria will be higher than the probability that it meets one criterion. Third, the thresholds in criteria A to D may be more precautionary. This would be justified because they are based on partial information and are often used in data-poor situations, whereas criterion E can (and should) incorporate all factors that influence population dynamics. In data-poor situations, where data permit only one or two of criteria A-D to be assessed, it would be very easy to ‘miss’ taxa that should be listed (Keith *et al*. 2000); in other words, the listing errors will be wider under A-D, so their thresholds should be more precautionary. Even so, it should be noted that while some studies suggest that criteria A-D are more precautionary than criterion E (e.g., Gärdenfors 2000), other studies indicate that criteria A-D may not be very precautionary, particularly when data are limited (e.g., Keith *et al*. 2004).

2.3.1 The quantitative thresholds

The quantitative values presented in the criteria associated with threatened categories were developed through wide consultation, and they are set at what are judged to be appropriate levels (i.e., levels that generate informative threat categories spanning the range of extinction probabilities; see below). Broad consistency between them was sought. The process and the technical background to the IUCN Red List system, and the fundamental biological processes underlying population decline and extinction that the criteria are based on, are described by Mace *et al*. (2008).

The quantitative values establish the thresholds between the Red List Categories CR and EN, EN and VU, and VU and NT. One misconception about the criteria has been that these thresholds are arbitrary. There is subjectivity in the establishment of boundaries among the categories of risk, and no theoretical reason why they should not be subjective (Collen *et al.* 2016). In fact, they have to be subjective, because they divide extinction risk, a continuous metric, into categorical blocks. Thus, their numerical values can only be based on practical, not theoretical, considerations.

Two types of practical considerations are relevant. The first is the usefulness or conservation-relevance of the resulting list of species in different threat categories. The continuum could have been divided differently, resulting in very few species, or a large majority of species, in one of the threat categories. In terms of informing conservation, this would not have been very useful. The current proportions of species in the three threatened categories show that the current boundaries are reasonable: both for fully assessed groups and for groups for which a random subset of species has been assessed, the proportion of taxa in each category is neither negligible nor overwhelming, meeting the Red List's goal to provide an informative index of extinction risk (Collen *et al.* 2016).

The second type of practical consideration involves limitations due to data availability and quality. For instance, the highest threshold for criterion A is set at 90% because if it were set any closer to 100% reduction, the taxon may go extinct before it can be classified as CR. The lowest threshold is set at 30%; it was increased from 20% in the previous version of the criteria (ver. 2.3; IUCN 1994) better to differentiate fluctuations from reductions. The remaining thresholds are then distributed more-or-less evenly between 30% and 90%. Similar considerations apply to the time horizon of criterion A, which needs to be long enough to allow actual declines to be detected and declines to be distinguished from fluctuations. The time horizon also needs to be short enough to allow reliable estimation, and to represent substantial extinction risk at a given overall decline. In terms of the spatial metrics, the thresholds balance the need for precision and the reality of the coarseness of spatial data for most taxa. For instance, the thresholds of area of occupancy (AOO) could have been set lower, but that would have required a more precise metric (a grid size smaller than the 2 × 2 km grid recommended; see section 4.10), which is impractical for many taxa.

An important attribute of the numerical thresholds in the criteria is that there is a single set of thresholds that applies to all taxa, allowing comparability across taxa. Of course, different taxa have different characteristics, and this variability is accounted for by using bespoke definitions, i.e., parameter definitions that incorporate life history characteristics of the taxon (such as the definition of the number of mature individuals). These definitions take into account the life history of the species; in addition, the criteria incorporate life history by scaling population responses to threatening processes with generation length to accommodate variation in population turnover (although, for practicality, the time horizon for future declines is limited to 100 years, regardless of generation length). Failure to consider correctly the definitions causes the majority of mistakes and misconceptions regarding the use of these standardized metrics. As a result, much of these guidelines (e.g., all of section 4) is devoted to definitions of terms and parameters used in the criteria.

Another important attribute of the numerical thresholds in the criteria is that they are exclusively tied to the definitions of the corresponding variables. In other words, comparing a measured value against a threshold requires that the value is measured as defined in IUCN (2001, 2012b) and in these guidelines. A common type of mistake is applying these thresholds to values of variables that are not calculated according to the definitions given here. For example, applying criterion A thresholds of reduction to declines over periods other than three generations/10 years (e.g., Shoo *et al.* 2005) would result in risk estimates that are not consistent with the Red List Categories (for other examples, see Akçakaya *et al.* 2006). Even when there are good reasons for measuring reduction over a different period (see section 4.5.1), the measured value must be scaled to the correct period before it can be compared to the criterion A thresholds. Similarly, applying the AOO thresholds to areas measured at high resolution (e.g., Cardoso *et al.* 2011), or applying the extent of occurrence (EOO) thresholds to areas calculated according to the definition of AOO (e.g., Ocampo-Peñuela *et al.* 2016) would result in threat categories that are not comparable to the

Red List Categories, and hence invalid application of the criteria. Consequently, the areas computed must be measured according to the definitions of AOO and EOO (or they must be scaled up or down as appropriate) before they can be compared to the thresholds of these variables. For further information, see sections 4.10.3 and 4.10.7.

2.4 Conservation priorities and actions

The category of threat is not necessarily sufficient to determine priorities for conservation action. The category of threat simply provides an assessment of the extinction risk under current circumstances, whereas a system for assessing priorities for action will include numerous other factors concerning conservation action such as costs, logistics, chances of success, and other biological characteristics (Mace and Lande 1991). The Red List should therefore not be interpreted as a means of priority setting (IUCN 2001, 2012b). The difference between measuring threats and assessing conservation priorities needs to be appreciated. However, assessment of taxa using Red List Criteria represents a critical first step in setting priorities for conservation action.

Many taxa assessed under the IUCN Red List Criteria will already be subject to some level of conservation action. The criteria for the threatened categories are to be applied to a taxon whatever the level of conservation action affecting it, and any conservation measures must be included with the assessment documentation. It is important to emphasize here that a taxon may require conservation action even if it is not listed as threatened, and that effectively conserved threatened taxa may, as their status improves over time, cease to qualify for listing.

2.5 Documentation

All assessments should be documented. Threatened classifications should state the criteria and subcriteria that are met. For example, for a taxon listed as Endangered A2cd, the criterion A2 indicates that the taxon has declined by 50% or more in the last 10 years or three generations (whichever is longer) and the subcriteria indicate that the decline in mature individuals has been caused by a decline in EOO, AOO, and/or the quality of habitat, as well as exploitation. Clearly listing the subcriteria provides the reasoning for placing a taxon in a specific category, and if necessary, the reasoning can be re-examined. No assessment at a threatened category or NT can be accepted for the IUCN Red List as valid unless at least one criterion and any qualifying subcriteria are given. If more than one criterion or subcriterion is met for the highest threat category, then each should be listed. Criteria meeting lower categories of threat should also be documented in the assessment Rationale. If a re-evaluation indicates that the documented criterion is no longer met, this should not result in automatic reassignment to a lower category of threat (downlisting). Instead, the taxon should be re-evaluated against all the criteria to clarify its status. The factors responsible for qualifying the taxon against the criteria, especially where inference, suspicion and projection are used, should be documented. All data used in a listing must be either referenced to a publication that is available in the public domain, or else be made available. Full documentation requirements are given in Annex 3 of the IUCN Red List Categories and Criteria (Version 3.1) (IUCN 2012b) and in *Documentation Standards and Consistency Checks for IUCN Red List Assessments and Species Accounts*, which is available for download at https://www.iucnredlist.org/resources/supporting-information-guidelines.

3. Data Quality

3.1 Data availability, inference, suspicion, and projection

The IUCN Red List Criteria are intended to be applied to taxa at a global scale. However, it is very rare for detailed and relevant data to be available across the entire range of a taxon. For this reason, the Red List Criteria are designed to incorporate the use of inference, suspicion, and projection, to allow taxa to be assessed in the absence of complete data. Although the criteria are quantitative in nature, the absence of high-quality data should not deter attempts at applying the criteria. In addition to the quality and completeness of the data (or lack of), there may be uncertainty in the dataset itself, which needs to be considered in a Red List assessment. Data uncertainty is discussed separately in section 3.2.

The IUCN criteria use the terms *Observed*, *Estimated*, *Projected*, *Inferred*, and *Suspected* to refer to the nature of the evidence (including aspects of data quality) for specific criteria. The relative order of data quality is Observed > Estimated (past) = Projected (future) > Inferred > Suspected. Some criteria have specific minimum data quality requirements (Table 3.1). For example, criterion A allows inferred or suspected reduction, whereas criterion C1 allows only estimated declines and criterion C2 specifies observed, projected, or inferred declines.

Table 3.1. Minimum data quality requirements for criteria A-E. If the data qualifier for the listed parameter is of lower quality than that specified ("Min. Quality") then the specified criterion is considered not to have been met, even if the numerical value meets the threshold for that criterion.

Criterion	**Parameter**	**Min. Quality**
A	Population reduction	suspected
B	Area of occupancy (AOO)	estimated
B	Extent of occurrence (EOO)	estimated
B1b, B2b	Continuing decline in EOO; AOO; area, extent and/or quality of habitat; number of locations or subpopulations; number of mature individuals	inferred
C, D	Number of mature individuals	estimated
C1	Estimated continuing decline	estimated
C2	Continuing decline in number of mature individuals	inferred
C2a(i)	Size of largest subpopulation	estimated
E	Extinction probability	estimated

Observed, Estimated, and Projected are similar in terms of their use in the criteria, and therefore in terms of their effects on a taxon's Red List category; the differences among them are important for documentation purposes only. The consequential differences are between this group (Observed/Estimated/Projected) and Inferred, and between Inferred and Suspected because these latter categories of evidence are not permissible for assessing some criteria (Table 3.1).

These terms are defined as follows:

Observed: information that is directly based on well-documented observations. Examples of observed information:

- Population size based on a census of all known mature individuals of the taxon.

- Population reduction derived from a census of all known mature individuals that took place three generations ago, and a current census of all known mature individuals.
- Continuing decline in the area, extent or quality of habitat derived from a survey of all known habitat, or from aerial photography of all known habitat.
- Continuing decline in the number of mature individuals derived from multiple censuses of all known mature individuals.

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) and the variable of interest (e.g., number of mature individuals). For example, transect counts of singing males of a bird species may be used with assumptions about the overall proportions of mature males these represent and about sex ratios to estimate the number of mature individuals. The assumptions used 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.

Additional examples of estimated information:

- Number of mature individuals calculated from a sample (cf. complete census) of (i) counts or density estimates carried out at sample areas across the taxon's range; (ii) an estimate of the proportion of mature individuals in the population derived from demographic information for the taxon; and (iii) the total area occupied by the taxon, derived from surveys sampling across its range.
- Population reduction or a continuing decline in the number of mature individuals derived from the estimated numbers of mature individuals at two or more time points, with or without extrapolation (see section 4.5.1).
- Population reduction or a continuing decline in the number of mature individuals derived from Catch Per Unit Effort data or trade estimates, with a known relationship to the species' population size (e.g., that CPUE and population size are linearly related).
- Continuing decline in the area or extent of habitat derived from remote-sensed land cover data or field surveys.
- EOO or AOO calculations that include 'inferred sites of occurrence', i.e., sites that are inferred from presence of known appropriate habitat, information about habitat requirements and dispersal capability of the taxon, rates and the effects of habitat destruction and other relevant factors (see section 4.10.7). Because the definition of EOO and AOO allow for 'inferred sites of occurrence', EOO and AOO based on such sites are considered to meet the criterion B data quality requirement for Estimated. However, inferred sites of occurrence should only be used to calculate the upper bound of the size of area, such that incorporating inferred sites results in a range of plausible values of EOO and AOO, which may lead to a range of plausible Red List Categories (see section 4.9).

Projected: same as "estimated", but the variable of interest is extrapolated in time towards the future, or in space. 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.

Examples of projected information:

- Population reduction derived from census data extrapolated into the future, either from the present (criterion A3), or from past and present (criterion A4), using statistical methods or population models (see sections 4.5.1 and 4.5.2).
- EOO or AOO calculations that are based on occurrences that include spatially projected sites of occurrence, i.e., sites that are based on a habitat model (see section 4.10.7).
- Continuing decline in the area or extent of habitat predicted by a statistical model of land-cover change that is based on an analysis of past land-cover changes derived from remote-sensed land cover data.

Inferred: information that is in the same general type of units but not a direct measure of the variable of interest (refer to definition of 'Suspected' below for examples that are not measured in the same general type of units). Examples include population reduction (A2d) 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. The main difference between Estimated (or Projected) and Inferred is that 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. When information replaces these additional assumptions, allowing calculation of the reduction based on data on trends in effort, efficiency, and distributions, the reduction can be considered Estimated. 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 4.5 for extrapolating population reduction for criterion A assessments).

Additional examples of inferred information:

- Number of mature individuals calculated by combining all of the following information: (i) density estimates based on counts of individuals carried out at sample areas (not in all subpopulations), or from a closely-related species with similar ecology and under similar threats, where they can be plausibly assumed to have similar densities; (ii) an estimate of the proportion of mature individuals in the population derived from demographic information for a closely-related taxon, which can be plausibly assumed to have similar population structure; and (iii) the total area occupied by the taxon, derived from remote-sensed data.
- Population reduction or a continuing decline in the number of mature individuals derived from numbers of mature individuals at multiple time points inferred from the types of information listed in the above point, with or without extrapolation (see section 4.5.1).
- Population reduction or a continuing decline in the number of mature individuals derived from trends in catch or trade statistics, with plausible assumptions about change (or lack of change) in effort and efficiency.
- Continuing decline in AOO or EOO based on land-cover changes derived from remote-sensed data, or based on evidence of decline in habitat quality.

- Continuing decline in area, extent and/or quality of habitat based on land-cover changes derived from remote-sensed data, or qualitative accounts of habitat loss or degradation.
- Continuing decline in the number of mature individuals (for criteria B and C2, but not C1) based on continuing decline in habitat of a species that is known to be a habitat specialist.

Suspected: information that is based on variables in different types of units, for example, % population reduction based on decline in habitat quality (A2c) or on incidence of a disease (A2e), or on circumstantial evidence. For example, qualitative information on habitat loss can be used to ***infer*** that there is a qualitative (continuing) decline in area of occupancy, 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.

Additional examples of information that could be used to ***suspect*** a reduction:

- Population reduction in the number of mature individuals based on information on trends in harvest, habitat quality, and sightings (e.g., from a structured elicitation of information from multiple experts familiar with the taxon).
- Population reduction in the number of mature individuals based on land-cover changes derived from remote-sensed data.
- Population reduction in the number of mature individuals based on a report of large numbers of individuals being hunted, poached or traded, where it is suspected that the level of exploitation will impact the population size, but there is no quantitative evidence to support this.

3.2 Uncertainty

The data used to evaluate taxa against the criteria are often obtained with considerable uncertainty. Uncertainty in the data should not be confused with a lack of data for certain parts of a species' range or a lack of data for certain parameters. This problem is dealt with in section 3.1 (Data availability, inference, suspicion and projection). Data uncertainty can arise from any one or all of the following three factors: natural variability, vagueness in the terms and definitions used in the criteria (semantic uncertainty), and measurement error (Akçakaya *et al.* 2000). The way in which uncertainty is handled can have a major influence on the results of an evaluation. Details of methods recommended for handling uncertainty are given below.

3.2.1 Types of uncertainty

Natural variability results from the fact that species' life histories and the environments in which they live change over time and space. The effect of this variation on the criteria is limited, because each parameter refers to a specific time or spatial scale. However, natural variability can be problematic, e.g. there is spatial variation in age-at-maturity for marine turtles, and a single estimate for these taxa needs to be calculated to best represent the naturally occurring range of values. Semantic uncertainty arises from vagueness in the definition of terms in the criteria or lack of consistency in different assessors' usage of them. Despite attempts to make the definitions of the terms used in the criteria exact, in some cases this is not possible without the loss of generality. These guidelines aim to reduce semantic uncertainty by explaining the terms in detail and for different contexts; and we encourage assessors to highlight remaining areas of semantic uncertainty. Measurement error is often the largest source of uncertainty; it arises from the lack of precise information about the quantities used in the criteria. This may be due to inaccuracies in estimating values or a lack of knowledge. Measurement error may be reduced or eliminated by acquiring additional data (Burgman *et al.* 1999, Akçakaya *et al*. 2000). Another source of measurement error is 'estimation error', i.e. sampling the wrong data or the consequences of estimating a quantity (e.g., natural mortality) based on a weak estimation method. This source of measurement error is not necessarily reduced by acquiring additional data.

3.2.2 Representing uncertainty

Uncertainty may be represented by specifying a best estimate and a range of plausible values for a particular quantity. The best estimate can itself be a range, but in any case the best estimate should always be included in the range of plausible values. The plausible range may be established using various methods, for example based on confidence or probability intervals, the opinion of a single expert, or the consensus view of a group of experts. The method used should be stated and justified in the assessment documentation.

3.2.3 Dispute tolerance and risk tolerance

When interpreting and using uncertain data, attitudes toward risk and uncertainty are important. First, assessors need to consider whether they will include the full range of plausible values in assessments, or whether they will exclude extreme values from consideration (known as dispute tolerance). Perceived uncertainty in the data is reduced when an assessor has a high dispute tolerance, and thus excludes extreme values from the assessment. On the one hand, it may sometimes be desirable to exclude the extreme values if these are unrealistic (e.g., the result of opinions that reflect biases rather than underlying data uncertainty). On the other hand, it is important that assessments accurately represent the range of uncertainty. We recommend that dispute tolerance (representing attitude towards uncertainty) is set to a low value, in most cases as low as 0.0 (including the whole range of possible outcomes).

Second, assessors need to consider whether they have a precautionary or evidentiary attitude to risk (known as risk tolerance). A precautionary attitude (i.e., low risk tolerance) will classify a taxon as threatened unless it is highly likely that it is not threatened, whereas an evidentiary attitude will classify a taxon as threatened only when there is strong evidence to support a threatened classification. A method developed for incorporating attitudes towards risk and uncertainty (Akçakaya *et al.* 2000) has been implemented in SIS as well as in RAMAS Red List (Akçakaya and Root 2007). Because these systems are used by a variety of institutions (e.g., for national assessments), it is not appropriate to set the values for attitude settings to specific constants. This is because the attitude settings are subjective, and reflect the assessors' values. However, for global listings in the IUCN Red List, it is appropriate to use a single risk tolerance value for all assessments, so that assessments are consistent across taxa. In particular, for the IUCN Red List, the risk tolerance value should not depend on factors such as ecological, evolutionary, economic, societal importance of the species; its chances of recovery; cost of measures to save it, etc. (such factors can be used in prioritization of conservation actions, but not for red-listing). This institutional setting for the IUCN Red List should reflect the reasons for this use (determination of global threat status), the overall objective of maintaining consistency of the IUCN Red List, and IUCN's values. IUCN (2001) specifies that "… when uncertainty leads to wide variation in the results of assessments, the range of possible outcomes should be specified. A single category must be chosen and the basis for the decision should be documented; it should be both precautionary and credible" and assessors "should resist an evidentiary attitude and adopt a precautionary but realistic attitude to uncertainty when applying the criteria". A precautionary but realistic attitude would require a slightly lower than mid-value for the risk tolerance parameter, perhaps a value in the range from 0.40 to 0.49.

3.2.4 Dealing with uncertainty

It is recommended that assessors should adopt a precautionary but realistic attitude, and to resist an evidentiary attitude to uncertainty when applying the criteria (i.e., have low risk tolerance). This may be achieved by using plausible lower and upper bounds, rather than only the best estimates, in determining the quantities used in the criteria. In cases where a statistical method is used to estimate a quantity, a 90% confidence interval or 90% credible interval may be used to set the plausible range of values. It is recommended that 'worst case scenario' reasoning be avoided because this may lead to unrealistically precautionary listings. All attitudes should be explicitly documented. In situations where the spread of plausible values (after excluding extreme or unlikely values) qualifies a taxon for two or more categories of threat, the precautionary approach would recommend that the taxon be listed under the higher (more threatened) category.

In some rare cases, uncertainties may result in two non-consecutive plausible threat categories. This may happen, for example, when extent of occurrence (EOO) or area of occupancy (AOO) is smaller than the EN threshold and one subcriterion is definitively met, but it is uncertain whether a second subcriterion is also met. Depending on this, the category can be either EN or NT. In such cases, the category could be specified as the range EN–NT in the documentation (giving the reasons why), and the assessors must choose the most plausible of the categories, of which VU could be one. This choice depends on the level of precaution (see section 3.2.3) and should be justified.

Specific guidelines for dealing with uncertainty in assessing taxa with widely distributed or multiple subpopulations against criterion A are given in section 4.5. This section offers clear guidance on using uncertain estimates, accounting for uncertainty about the pattern of population decline and using data with different abundance units.

3.2.5 Documenting uncertainty and interpreting listings

The level of uncertainty associated with a particular taxon's assessment is not apparent from the listing itself, potentially complicating and de-valuing interpretation of listings. When a plausible range for each quantity is used to evaluate the criteria, a range of categories may be obtained, reflecting the uncertainties in the data. However, only a single category, based on a specific attitude to uncertainty, will be listed along with the relevant criteria on the IUCN Red List. It is important to note that the range of possible categories should also be indicated, along with the assessors' attitudes to uncertainty, in the documentation accompanying the assessment. The inclusion of information on uncertainty in the documentation, allows users of the Red List access to important information that will assist in the interpretation of listings, and inform debates over particular issues or listings.

3.2.6 Uncertainty and the application of the categories Data Deficient and Near Threatened

The level of uncertainty in the data used for assessments may or may not affect the application of the categories Data Deficient and Near Threatened. Guidance on the application of these categories is given in section 10.

4. Definitions of Terms Used in the Criteria and their Calculation

The terms used in the IUCN Red List Categories and Criteria must be clearly understood to ensure that taxa are correctly assessed. The following terms are defined in the *IUCN Red List Categories* *and Criteria (version 3.1)* on pages 10–13 (IUCN 2001, 2012b). These definitions are reproduced here, with additional guidelines to assist in their interpretation and calculation.

4.1 Population and population size (criteria A, C and D)

“The term ‘population’ is used in a specific sense in the Red List Criteria that is different to its common biological usage. Population is here defined as the total number of individuals of the taxon. For functional reasons, primarily owing to differences between life forms, population size is measured as numbers of mature individuals only. In the case of taxa obligately dependent on other taxa for all or part of their life cycles, biologically appropriate values for the host taxon should be used.” (IUCN 2001, 2012b)

The definition above means that a "population" (*sensu* IUCN 2001, 2012b) includes all individuals (mature and other life stages) that are assigned to the taxon throughout its distribution. “Population” and “Population size” are, however, not synonymous. There are two important aspects of the definition of population size. First, population size is measured only in terms of mature individuals. Thus, the interpretation of this definition depends critically on an understanding of the definition of ‘mature individuals’, which is given and discussed below in section 4.3. Second, population size is defined as the total number of mature individuals in all areas. Even if some of the taxon exists in subpopulations that might be seen as distinct populations in a general biological sense, for the purposes of the criteria, the total number of mature individuals in all areas (or all subpopulations) is used to measure the "population size" of the taxon.

4.2 Subpopulations (criteria B and C)

“Subpopulations are defined as geographically or otherwise distinct groups in the population between which there is little demographic or genetic exchange (typically one successful migrant individual or gamete per year or less).” (IUCN 2001, 2012b)

The significance of subpopulations in the criteria relates to the additional risks faced by taxa where the population is either subdivided into many small spatial units or where most individuals are concentrated into one such unit. Operational methods for determining the number of subpopulations may vary according to the taxon; in the case of tree species, for example, a subpopulation can be defined as a spatially distinct segment of the population that experiences insignificant or reproductively unsuccessful migration (of seed or pollen) from other subpopulations.

Although subpopulations typically have little demographic or genetic exchange, this may or may not amount to their complete isolation in this regard. In other words, subpopulations need not be completely isolated. Even highly mobile species may have multiple subpopulations, as high mobility is not always a guarantee of genetic or demographic connectivity. For example, even if a species migrates thousands of kilometres annually, if it has very high fidelity to both natal and breeding sites, there could be few dispersers among subpopulations within the breeding range, making it necessary to recognize multiple subpopulations.

4.3 Mature individuals (criteria A, B, C and D)

“The number of mature individuals is the number of individuals known, estimated or inferred to be capable of reproduction. When estimating this quantity the following points should be borne in mind:

- Mature individuals that will never produce new recruits should not be counted (e.g., densities are too low for fertilization).
- In the case of populations with biased adult or breeding sex ratios, it is appropriate to use lower estimates for the number of mature individuals, which take this into account.
- Where the population size fluctuates, use a lower estimate. In most cases this will be much less than the mean.
- Reproducing units within a clone should be counted as individuals, except where such units are unable to survive alone (e.g., corals).
- In the case of taxa that naturally lose all or a subset of mature breeding individuals at some point in their life cycle, the estimate should be made at the appropriate time, when mature individuals are available for breeding.
- Re-introduced individuals must have produced viable offspring before they are counted as mature individuals." (IUCN 2001, 2012b)

4.3.1 Notes on defining mature individuals

This definition of mature individuals differs slightly from that given in version 2.3 of the Red List Categories and Criteria (IUCN 1994). Some groups have found the more recent definition of mature individuals to be less conservative and less precise, leading to a potential down-listing of some taxa (e.g., obligate co-operative breeders), even though their extinction risk has not changed. It must be stressed that the intention of the definition of mature individuals is to allow the estimate of the number of mature individuals to take account of all the factors that may make a taxon more vulnerable than might otherwise be expected. The list of points given with the definition is not exhaustive and should not restrict an assessor's interpretation of mature individuals, provided they are estimating the number of individuals known, estimated or inferred to be capable of reproduction. "Reproduction" means production of offspring (not just mating or displaying other reproductive behaviour). The ability of an assessor to estimate or infer which individuals are capable of reproduction is paramount and highly contingent on the particular features of the taxon or group. Juveniles, senescent individuals, suppressed individuals and individuals in subpopulations whose densities are too low for fertilization to occur will never produce new recruits, and therefore should not be counted as mature individuals. On the other hand, in many taxa there is a pool of non-reproductive (e.g., suppressed) individuals that will quickly become reproductive if a mature individual dies. These individuals can be considered to be capable of reproduction. For example, in social bees and ants there is often just one or a few actually reproducing females ("queens") at a time, but new such queens can be promoted from larvae under development or from reproductively suppressed workers, if a functional queen were to die. A possible template for the number of 'mature individuals' in such societies could be the number of queens * 10 (an expression for the number of potential queens that could realistically be produced) * 2 (the male counterpart). In general, the judgement will be best made by assessors with insight into the species' biology.

These considerations also apply to cases of populations with biased adult or breeding sex ratios. In such cases, it is appropriate to use lower estimates for the number of mature individuals, which take this into account. An appropriate lower estimate will depend on whether individuals of the limiting sex are biologically capable of reproduction. For example, if there are 100 males and 500 females capable of reproduction, then the number of mature individuals would be < 600, perhaps as small as 200 (=100*2, if mating is strictly monogamous). However, if these are the number of actual breeders in some years and there are 400 other males capable of reproduction (but that did not breed in the year the data were collected), then there would be 1,000 mature individuals.

Note that effective population size (N_e) cannot be used as an estimate of the number of mature individuals. One reason is that reproductively suppressed individuals do not contribute to the calculation of N_e, but, as explained above, they may be counted as mature individuals.

In the case of taxa obligately dependent on other taxa for all or part of their life cycles, biologically appropriate values of mature individuals for the host taxon might be used. This number may be much less than the total number of mature individuals of the host taxon, because generally other factors restrict the dependant taxon from utilizing all host individuals.

The number of mature individuals can be estimated using the equation $d * A * p$, where d is an estimate of population density, A is an estimate of area, and p is an estimate of the proportion of individuals that are mature. However, this approach often leads to gross overestimates of number of mature individuals if the parameters are not set appropriately. Therefore, great care should be taken when using this formula to ensure that: (a) the area is appropriately selected and d is an estimate of the average over the entire A (for example, the estimate will be positively biased if A is set to EOO and d is based on samples from areas of highest density), and (b) p should be selected based on knowledge of the taxon (or related taxa) rather than being set to a default value (such as 0.5) because the proportion of mature individuals in a population differs markedly among taxa. Bounds on the estimate of number of mature individuals can be obtained by placing bounds on each of d, A, and p. The value from this approach will be an *estimate* if the values for d, A and p are all estimates, but should be considered to be an *inference* if one or more of these values are based on inference (e.g., if the value for p is based on data for individuals from a subset of the area the taxon is found in rather than a random set of individuals across the area). Estimates of d (population density) should incorporate the imperfect detection of individuals, ideally using an estimate of the probability of detection.

4.3.2 Clonal colonial organisms

Clonal colonial organisms include most corals, algae, bryophytes, fungi and some vascular plants. As opposed to a unitary organism, such as a vertebrate, an insect and many vascular plants, the growth and development of a clonal (modular) colonial organism is an iterative process in which "modules" are added step by step to the existing structure. In principle, the growth of a modular organism never ends and it has no final shape, size or age. A modular organism (the genet) can sometimes exist in a form of many parts (ramets), which can become more or less isolated from each other. Consequently, what constitutes a 'mature individual' in a colonial or modular organism is not always clear. Still, it is important to define 'mature individual' for such organisms, since 'mature individual' is used under criteria C and D to capture the effects of threats and demographic stochasticity to a small population. In defining 'mature individual' for colonial organisms, it is important to identify entities that are comparable in demographic stochasticity and extinction proneness to a population of discrete individuals of animals. For some taxa (e.g., reef-forming corals), it may also help to consider what entity typically lives, is injured, and dies as a unit.

As a general rule, the ramet, i.e., the smallest entity capable of both independent survival and (sexual or asexual) reproduction should be considered a 'mature individual'. Reproducing units within a clone should be counted as individuals, except where such units are unable to survive alone (IUCN 2012b). For instance, in those cases where the organism appears in well-distinguishable units, each such unit would be counted as one mature individual. Examples may be a bryophyte tuft (e.g., of *Ulota*) or a discrete cushion (e.g., *Brachythecium*), a lichen thallus (e.g., *Pseudocyphellaria*) or foliose patch (e.g., *Parmelia*), or a coral discrete entity (e.g., a brain coral *Diploria* or sun coral *Tubastrea*).

If the delimitation of ramets is not obvious, but the species lives in or on a discrete and relatively small substrate unit limited by a certain resource, e.g. a piece of cow dung, a leaf or a dead tree branch, each unit colonized by the species should be counted as a single mature individual. In many other cases, like reef-forming corals, cliff-growing lichens and ground-growing fungi, the organism grows in large, more or less continuous entities that could be divided into smaller pieces without obviously harming the organism. In principle, the smallest such entity (ramet) that an organism could be divided into without causing its death or preventing reproduction, should be counted as one mature individual. Obviously, what such an entity would be is often not known. Therefore, in such cases, it may be necessary to adopt a pragmatic approach to defining ‘mature individuals’. Examples of possible interpretations of the definition of a mature individual are:

- For diffuse, wholly visible organisms in continuous habitats (e.g., reef-forming corals, algal mats) assessors may assume an average area occupied by a mature individual and estimate the number of mature individuals from the area covered by the taxon. The area covered by the taxon should be estimated at a scale (grid size; e.g. 1 m^2) that is as close as practicable to the area assumed to be occupied by a single mature individual. (However, note that AOO must still be estimated using the 2×2 km grids.)
- For diffuse organisms, not wholly visible, in continuous habitats (e.g., subterranean mycelial fungi) assessors may assume that each recorded presence separated by a minimum distance represents an assumed number of individuals. For example, each visible fungal fruiting body may be assumed to represent ten mature individuals, so long as they are separated by at least 10 metres. This kind of assumption is necessary because the size or area of a fungal mycelium is rarely known.
- For diffuse organisms that occur in discrete habitat patches (e.g., fungi living more or less concealed in dead wood), each patch (trunk or log colonized by the species) could – if no better information exists – be counted as 1–10 mature individuals, depending on the size of the tree.

In any case, it is recommended that authors of Red List assessments specify the way they have used ‘mature individual’.

4.3.3 Fishes

In many taxa of marine fish, reproductive potential is commonly closely related to body size. Since exploitation usually reduces the mean age and size of individuals, assessing declines in numbers of mature individuals may under-estimate the severity of the decline. When evaluating population decline, this factor should be kept in mind. One possible method is to estimate decline in the biomass of mature individuals rather than the number of such individuals when applying criterion A, where biomass is ‘an index of abundance appropriate to the taxon’.

4.3.4 Sex-changing organisms

Many marine taxa have the capacity to change sex as they grow. In such taxa, the sex ratio may be highly biased towards the smaller sex. The criteria acknowledge that the number of mature individuals can take biased sex ratios into account, by using a lower estimate for the number of mature individuals. For sex-changing organisms it is also appropriate to consider changes in sex ratio as an indicator of population perturbation, which may be of additional conservation concern because the larger sex (already less numerous) is often subject to higher harvest mortality. In these cases, the number of mature individuals may be estimated by doubling the average number of individuals of the larger (or less numerous) sex.

4.3.5 Trees

Individual trees that flower without producing viable seeds do not qualify as mature individuals. For example, *Baillonella toxisperma* first flowers at 50–70 years and does not fruit until roughly 20 years later. Conversely, *Sequoiadendron giganteum* may produce seed at less than 20 years of age and continue to do so for 3,000 years. However, not all trees between these ages may be mature individuals if the population includes some reproductively suppressed individuals. If little is known about age at fruiting, mature individuals should be counted as those of a typical reproductive size; e.g. estimates for canopy taxa should exclude sub-canopy individuals. Vegetative clones, apomictic taxa and self-fertilizing taxa may qualify as mature individuals, so long as they produce viable offspring and their survival is independent of other clones.

Where it is impossible to calculate the number of mature individuals, but information is available on the total population size, it may be possible to infer the number of mature individuals from the total population size.

4.4 Generation (criteria A, C1 and E)

"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." (IUCN 2001, 2012b)

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 (e.g., Bird *et al.* 2020).

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
$$G = \sum x l_x m_x \Big/ \sum l_x m_x$$
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 (see below). 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 reproducing), averaged over all reproducing individuals. If first reproduction (production of offspring) typically occurs by 12 months, use 0, not 1; if it occurs between 12 and 24 months, use 1, etc. (See below for further discussion on definition of "age").
3. Age of first reproduction + [z * (length of the reproductive period)], where z is a number between 0 and 1; z is usually <0.5, depending on survivorship and the relative fecundity of young vs. old individuals in the population. For example, for mammals, two studies estimated z= 0.29 and z=0.284 (Pacifici *et al.* 2013, Keith *et al.* 2015). 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. Generation length (as well as age of first reproduction for (2) and (3) above) should be calculated over all reproducing individuals. If the estimate of generation length differs between males and females it should be calculated as a weighted average, with the weighting equal to the relative frequency of reproducing individuals of the two sexes. However, if the two sexes are impacted differentially by some threat, this should be taken into account and pre-disturbance generation length should be used for both sexes before calculating the weighted average (see below for further discussion on pre-disturbance generation length).
5. For partially clonal taxa, generation length should be averaged over asexually and sexually reproducing individuals in the population, weighted according to their relative frequency.
6. 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.

The formula given in option 1 is implemented in the workbook (spreadsheet) file **Generation_Length_Workbook.xls**, which is available at https://www.iucnredlist.org/resources/generation-length-calculator. This workbook is also useful for exploring the effects of various assumptions in options 2 and 3 on the calculated generation length.

The correct use of the methods described above requires that "age" is defined in a specific way. The definition affects, for instance, the age of first reproduction for equations in (2) and (3) above, as well as fecundity (F) as a function of age for the equation in (1) and in the spreadsheet. For purposes of these methods, an individual is zero-years old until its first birthday. For species with a distinct reproductive season (e.g., many species in temperate regions), $F(0)$ is the number of offspring produced per individual in the reproductive season that is after the one in which the individual was born, regardless of how age is reckoned. In general (including other types of life histories, such as species with no specific, or a much longer, "reproductive season"), $F(0)$ is the number of offspring produced per individual in its first 12 months. If an alternate definition is used, the formulae need to be modified to reflect the definition. For example, if age is defined such that age of first reproduction is 1 (not zero) when the first reproduction occurs by 12 months, then the formula in (2) should be "1/adult mortality + age of first reproduction – 1" (see Bird *et al.* 2020 for an example of the application of this formulation).

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 reproducing 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 three generations ago (see detailed explanation and examples in section 4.5.3).

The reason IUCN (2001, 2012b) requires using "pre-disturbance" generation length for exploited populations is to avoid a shifting baseline effect. This would arise because using current, shorter generation length (under disturbance, such as harvest) may result in a lower threat category (because a shorter period is used to calculate the reduction), which may lead to further harvest. Thus, using generation length under harvest would represent a case of shifting baseline based on a change caused by human impacts. Harvest mortality shifts the age structure and the survival rates, and in some cases (e.g., some terrestrial mammals) harvest of older individuals allows younger individuals, whose reproduction had been suppressed by the older individuals, to reproduce. In addition, in many cases, the reduction in generation length is a demographic response (rather than a genetic response) resulting from overexploitation; this may result in reduced bet-hedging (risk-spreading) capacity and a lower, more variable population growth rate, which then increases the probability of extinction. Even in cases where the response has a genetic basis, it represents an artificial selection that would still lead to the shifting baseline described above.

4.5 Reduction (criterion A)

"A reduction is a decline in the number of mature individuals of at least the amount (%) stated under the criterion over the time period (years) specified, although the decline need not be continuing. A reduction should not be interpreted as part of a fluctuation unless there is good evidence for this. The downward phase of a fluctuation will not normally count as a reduction." (IUCN 2001, 2012b)

In the subsections below, various approaches to calculating population reduction are discussed, including statistical methods (4.5.1) and population models (4.5.2). Main issues involved in calculating population reduction using statistical methods include the patterns of decline, and the methods of extrapolation based on these patterns. Finally, methods for combining information from multiple regions or subpopulations to calculate the reduction for the taxon are discussed (4.5.3). The methods discussed in these sections also apply to calculating estimated continuing decline (4.6), except that the time period for calculating estimated continuing decline depends on the category (e.g., for CR, the longer of 1 generation or 3 years).

> Many of the calculations discussed in the sections below are implemented in the workbook (spreadsheet) file **CriterionA_Workbook.xls**, which is available at https://www.iucnredlist.org/resources/criterion-a . Make sure to check all the tabs in the file.

4.5.1 Calculating population reduction using statistical methods

Statistical models can be used to extrapolate population trends so that a reduction of three generations can be calculated. The model to be fitted should be based on the pattern of decline (which may be exponential, linear, accelerated, or a more complex pattern), which may be inferred from the type of threat. The assumed pattern of decline can make an important difference. Assessors should indicate the basis on which they have decided the form of the decline function. The best information about the processes that contribute to changes in population size should be used to decide what form of decline function to apply over the three-generation period. Specifically, if a model is fitted, the assumptions of the model must be justified by characteristics of life history, habitat biology, pattern of exploitation or other threatening processes, etc. For example:

(1) If a taxon is threatened by exploitation, and the hunting mortality (proportion of individuals taken) does not change as the population size declines, then the population is likely to be declining exponentially, and this model should be fitted.

(2) A linear model is appropriate when the number of individuals removed from the population on an annual basis (rather than their proportion to the total population) remains the same as the population changes. For example, if a taxon is threatened with habitat loss, and a similar sized area of habitat is lost every year, this could lead to a linear decline in the number of individuals.

(3) A model with an accelerating decline rate is appropriate if the threat processes have increased in severity over time and these are affecting the population in an increasingly severe manner.

(4) No model need be fitted in cases where there are only two estimates of population size (at the start and end of the time period specified in the criteria) – the reduction can be calculated from these two points.

The population data from which a reduction can be calculated are likely to be variable, and it may not be obvious how a reduction should best be calculated. Depending on the shape of the data, a linear or exponential model may be fitted (see section 4.5.2), and the start and end points of the fitted line used 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, which should not be included. Fitting a time series longer than three generations or 10 years (as applicable) may give a more representative estimate of the long-term population reduction, especially if populations fluctuate widely, or oscillate with periods longer than the generation time (Porszt *et al.* 2012). However, regardless of the length of the time series fitted, the reduction should be calculated for the most recent three generations or 10 years, as applicable (Akçakaya *et al.* 2021). Figure 4.1 shows an example where the three-generation period is from 1920 to 2000, but data are available from 1900. The relationship between the number of mature individuals and time is based on all the data (dashed line) but the reduction is calculated over years 1920 to 2000.

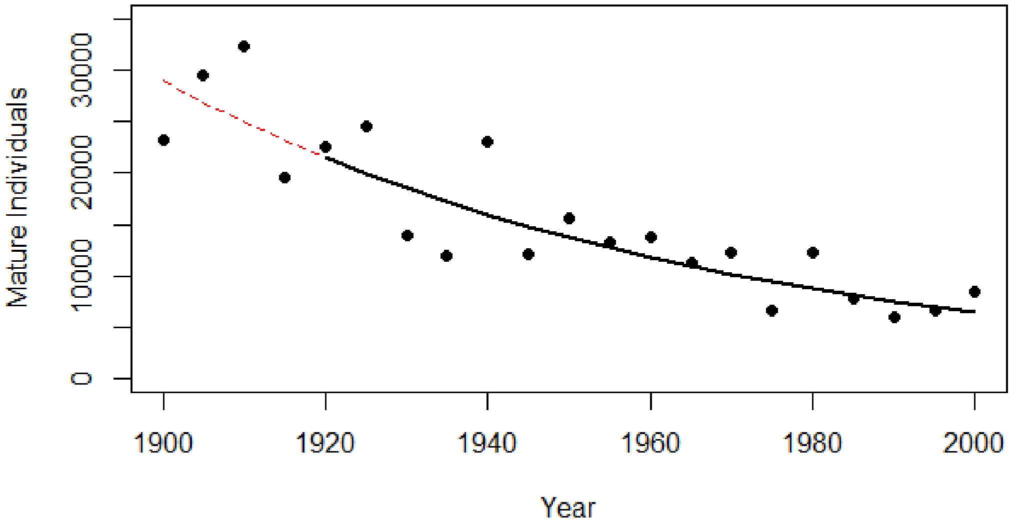


Figure 4.1. Example of using data for more than three generations (1900 to 2000) to estimate a reduction over the period 1920 to 2000.

Here, we briefly discuss various assumptions, and where they might be applicable. Consider a species with a 20-year generation time and suppose population size was estimated as 20,000 in 1961 and 14,000 in 1981 (these are shown as square markers in the graphs below). To calculate past reduction, we need to extrapolate back in time to 1941 and forward to 2001.

The simplest assumptions are those that involve no change in early or late years. For example, if it is assumed that decline did not start until the early 1960s, the reduction can be based on the initial population of 20,000. If it can be assumed that the decline stopped before 1981, then 14,000 can be used as the current population size (Figure 4.2a), resulting in a 30% reduction (1–(14,000/20,000)). However, it is necessary to make an assumption about the pattern of decline if some decline is suspected to have occurred outside this period. The documentation should include a rationale for the assumed pattern of decline.

Exponential decline

Exponential decline can be assumed in cases where the proportional rate of decline of the population is believed to be constant. For example, an exponential decline can be assumed if the taxon is threatened by exploitation, and the hunting mortality (proportion of individuals taken) does not change as the population size declines. For the case where there are estimates of population size, the reduction is calculated using the equations:

$$\text{Reduction} = 1 - (\text{Observed Change})^{(3\text{Generations}\ /\text{Observed Period})}$$

where "Observed Change" is the ratio of the second population size to the first population size (in this case $N(1981)/N(1961)$), and "Observed Period" is the number of years between the first and last observation years. For example, in Figure 4.2b, the Observed Change is 14,000/20,000 and the Observed Period is 20 years. Thus, the 60-year reduction is 65.7% [$=1-(14{,}000/20{,}000)^{(60/20)}$]. The annual rate of change is calculated as:

$$\text{Annual Change} = (\text{Observed Change})^{(1/\text{Observed Period})}$$

For this case, the annual rate of change is 0.9823, which suggests about 1.8% annual rate of decline. The population size three generations ago can be estimated as 28,571 [=20,000/0.9823^20], and the current population as 9,800 [=14,000*0.9823^20] (Figure 4.2b). The worksheet "Exponential decline" in the spreadsheet **CriterionA_Workbook.xls** mentioned above can be used to calculate reductions.

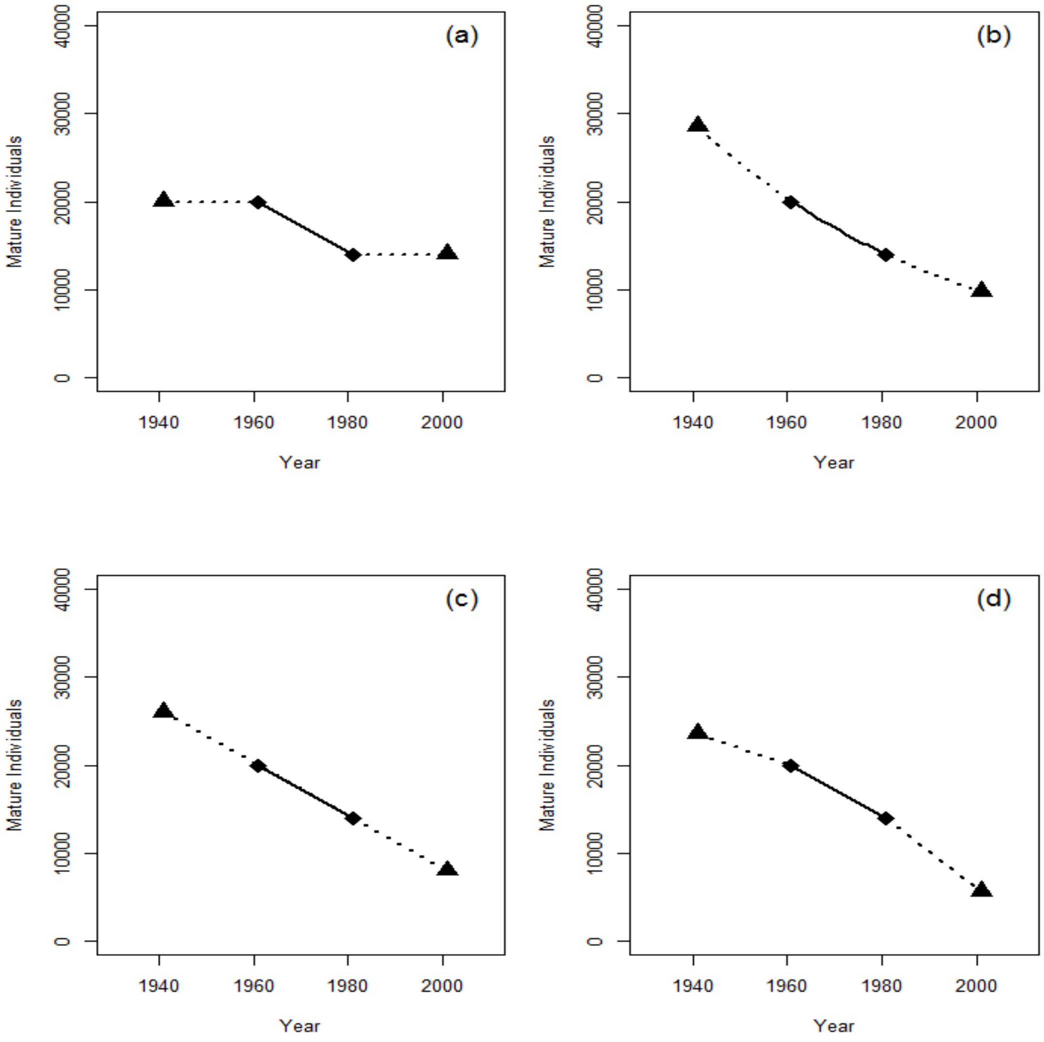


Figure 4.2. Examples of calculating past population reduction, for an assessment made in 2001 of a species with a generation length of 20 years. Population size was estimated as 20,000 in 1961 and 14,000 in 1981; extrapolations were made because past reduction is to be calculated over the last three

generations, from 1941 to 2001. Calculations assume: (a) no change from 1941 to 1961 and from 1981 to 2001, (b) exponential change between 1941 and 2001, (c) linear decline between 1941 and 2001, and (d) accelerated decline from 1941 to 2001.

Linear decline

In some cases, the number of individuals removed from the population (rather than their proportion to the total population) may remain constant. For example, if a species is threatened with habitat loss, and a similar sized area of habitat is lost every year, this could lead to a linear decline in the number of individuals. Note that this means that the rate of decline is increasing every year, because the same amount of habitat is lost out of a decreasing amount of remaining habitat. So, we cannot calculate a single rate of decline (as a percentage or proportion of population size), as we did in the exponential case. Instead, we can calculate annual reduction in units of the number of individuals:

Annual Reduction in N= (First N – Second N)/(Observed Period)

where "First N" is the population size observed at the start of the observed period, and "Second N" is the population size observed at the end. For the example, the annual reduction is 300 individuals ((20,000-14,000)/20). Now, we need to calculate the population sizes at the start and end of the 3-generation period. To do this, we first calculate:

Abundance1 = First N + (Annual Reduction * Period1)

Abundance2 = larger of zero or: Second N – (Annual Reduction * Period2)

where Abundance1 is the calculated population size at the start of the 3-generation period and Abundance2 is the calculated population size at the end of the 3-generation period. Abundance1 and Abundance2 are calculated from the calculated annual reduction in mature numbers, the two population sizes and the number of years between when the population sizes were obtained. Period1 is the difference in the number of years between the start of the 3-generation period and the year for which the first population size observation is available (1941 and 1961 for the example) and Period2 is the difference in the number of years between the end of the 3-generation period and the year for which the second population size observation is available (1981 and 2001 for the example). Finally, we calculate the 3-generation proportional (percentage) reduction as follows:

Reduction = (Abundance1 – Abundance2) / Abundance1

For the example, the annual reduction is 300 individuals per year so the number of individuals in 1941 and 2001 would be 26,000 [20,000+(300*20)] and 8,000 [14,000–(300*20)] respectively (triangle markers in Figure 4.2c), giving a 3-generation reduction of about 69.2%. In this case, the rate of decline is only 23% for the 1st generation, but increases to 43% for the 3rd generation. The worksheet “Linear decline” in the spreadsheet **CriterionA_Workbook.xls** mentioned above can be used to calculate reductions.

Accelerated decline

Although a linear decline in the number of individuals means that the rate of decline is increasing, this increase can be even faster, leading to an accelerated decline in the number of individuals. This may happen when the exploitation level increases, for example when the number of individuals killed is larger every year because of increasing human population, or improving harvest efficiency.

To extrapolate under an assumption of accelerated decline, it is necessary to know or guess how the rate of decline has changed. For instance, in the above example, the observed 1-generation decline (from 1961 to 1981) is 30%. One assumption might be that the rate of decline doubled in each generation, from 15% in the 1st generation to 30% in the 2nd and 60% in the 3rd. This assumption would lead to population size estimates of 23,529 for 1941 (20,000/(1–0.15)) and 5,600 for 2001 (14,000*(1–0.6)), giving a 3-generation past reduction of about 76% (Figure 4.2d). Of course, different assumptions about how the rates of decline may have changed in the past will give different results.

The same approach can be used to make the calculation based on an assumption of decelerating decline.

Complex patterns of decline

It is possible to assume different patterns of decline for different periods. For example, decline can be assumed to be zero until the first observation, and then exponential. This would give a population of 20,000 for 1941 and 9,800 for 2001, giving a three-generation past reduction of about 51%.

The examples in Figure 4.2 were based on two values for the number of individuals. When multiple estimates of population size are available the data need to be smoothed, using for example regression (Figure 4.1). When applying regression, it is important to check that the fitted line goes through the data well. For example, Figure 4.3 shows a case where a linear model is not an adequate fit to the data. In this case a past reduction could be calculated as the ratio of the average population size for the last 8 years (10,329) to that for the years before overexploitation occurred (19,885). The reduction would be 48% (1–(10,329/19,885)).

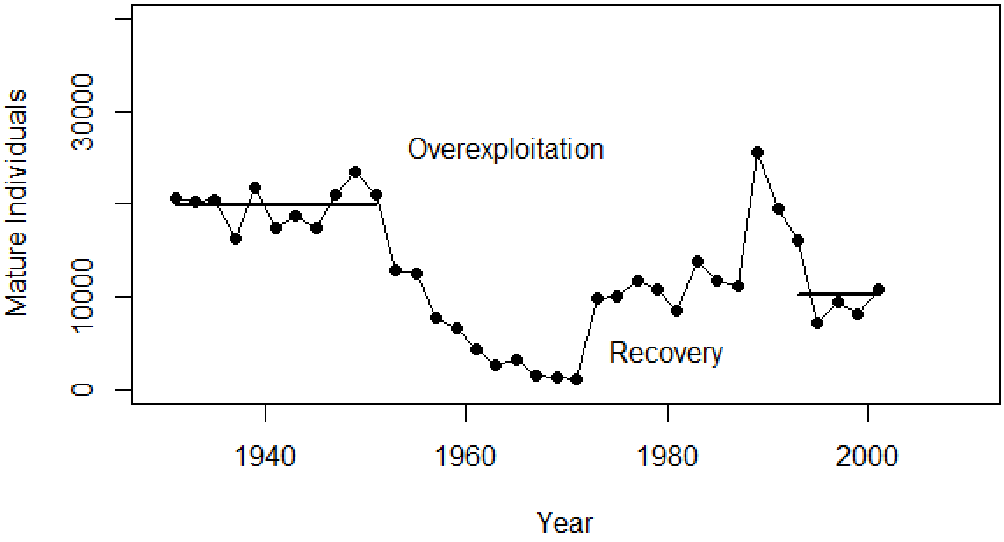


Figure 4.3. An example of calculating past reduction for a population that is initially stable but then subject to overexploitation followed by recovery. Reduction is based on the average population sizes of the last few years and the years before overexploitation occurred.

Calculating reductions by the ratio of the average population size at the start of the 3-generation period to the average population size at the end of the 3-generation period is appropriate when there is evidence for change in trend (e.g., due to changes in threatening processes). In contrast, regression (linear or exponential) should be used to calculate reductions if there is no such evidence or the population size estimates are very imprecise.

Finally, when there is no basis for deciding among various patterns of decline, the rate of decline can be specified as an uncertain number, based on the declines predicted by the different patterns. For example, in the set of four examples in Figure 4.2 above, the rate of decline can be expressed as the interval 66%–69%, if both exponential and linear patterns of decline are considered plausible, or as the interval 30%–76%, if all four possibilities discussed are considered plausible.

4.5.2 Calculating population reduction using population models

Past and future population reduction can be calculated using population models, provided that: (i) the model meets the requirements outlined in section 9 ("Guidelines for Applying Criterion E"), (ii) the effects of future levels of threat are included in the population model, represented as changes in model parameters, and (iii) the model outputs are not inconsistent with expected changes in any estimated or inferred current or recent rates of decline. When using a population model to project a reduction under criterion A3, the median or mean of the projections for a range of plausible scenarios should be used to calculate a best estimate of the magnitude of the projected reduction. Assessments may be based on the best estimate, lower or upper bound but, for reasons of transparency, assessors must justify the rationale for their choice if a value other than the best estimate is used. The projected variability may be used to quantify uncertainty. For example, upper and lower quartiles of the projected magnitude of the future reduction (i.e., reductions with 25% and 75% probability) may be considered to represent a plausible range of projected reduction, and used to incorporate uncertainty in the assessment, as described in sections 3.2 and 4.5.3. The bounds on the plausible range should incorporate uncertainty about the model used for projection, as well as measurement error; or a justification of the model structure, and why it is the most appropriate in the face of model uncertainty, should be provided.

4.5.3 Taxa with widely distributed or multiple subpopulations

This section addresses the issues related to the presentation and use of information from subpopulations (or from parts of the range) of a widely distributed taxon, in assessing the taxon against criterion A. For such taxa, it is recommended that the available data on past reduction be presented in a table that lists all known subpopulations (or parts of the range), and gives at least two of the following three values for each subpopulation:

1. the estimated population size at a point in time close to three generations ago[1], and the year of this estimate;
2. the most recent estimated population size and its year; and
3. estimated, suspected or inferred reduction (in %) over the last three 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 subpopulation should be summarized in a separate column, as well as quantities calculated based on the presented data (see examples below). Population sizes and reductions should be estimated separately for each subpopulation using the methods described above, taking into account that different subpopulations may exhibit different patterns of decline.

---

[1] The criteria are defined in terms of the maximum of 10 years or three generations. However, for clarity of presentation, reference is only made in this section to "three generations".

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 subpopulations should be non-overlapping. This does not mean that there is no or infrequent dispersal among subpopulations. The point of this requirement is to avoid double-counting as much as possible.

(c) Together, the subpopulations should include all of the taxon. If this is not possible, a "subpopulation" named *Remainder* should include an estimate of the total number of mature individuals not included in the listed subpopulations. This estimate, like others, can be uncertain (see below).

If these requirements cannot be met, the taxon cannot be assessed under criterion A.

In this section, we refer to subpopulations, but the discussion applies to any type of non-overlapping subunits of the taxon, such as parts of the taxon's range. In the next subsection on *Estimating overall reduction*, we discuss the basic methods of using the data table outlined above for assessing a taxon under criterion A. In many cases, there will be uncertainty, because the population sizes are not known precisely, are in different units for different subpopulations, or are available only from one or few subpopulations. These cases will be discussed later, in a subsection on *Dealing with uncertainty.*

4.5.4 Estimating overall reduction

To assess a taxon against criterion A, it is necessary to estimate the overall reduction over three generations or 10 years. All available data should be used to calculate a reduction as an average over all subpopulations, weighted by the estimated size of each subpopulation at the beginning of the period. Inferences regarding reductions should not be based on information for any single subpopulation (whether it is the fastest declining, most stable, largest or smallest)[2].

The recommended methods for estimating reduction are explained below by a series of examples. All examples are for calculating past reduction for a taxon with a generation length of 20 years, assessed in 2001 (i.e., for these examples, the "present" is 2001 and "three generations ago" is 1941). All examples of this section are based on data with the same units for all subpopulations; the issue of different units is discussed in the next subsection (*Dealing with uncertainty*).

The worksheet "Multiple populations" in the spreadsheet **CriterionA_Workbook.xls** (mentioned at the start of section 4.5) can be used to calculate reductions using data from multiple populations.

Example 1: Estimates are available for past (three generations ago) and current population sizes.

Subpopulation	**Past**	**Present**
Pacific Ocean	10,000 (1941)	5,000 (2001)
Atlantic Ocean	8,000 (1941)	9,000 (2001)
Indian Ocean	12,000 (1941)	2,000 (2001)
Overall	30,000 (1941)	16,000 (2001)

---

[2] However, see "*Dealing with uncertainty*" below for a discussion of exceptions to this rule.

In this (simplest) case, all past population sizes are added up (30,000) and all present population sizes are added up (16,000), giving an overall reduction of 46.7% [(30-16)/30]. Note that the changes in individual subpopulations are 50% reduction, 12.5% increase and 83.3% reduction. An average of these numbers, weighted by the initial population sizes, gives the same answer [(-0.5*10+0.125*8-0.833*12)/30].

Example 2: Estimates are available for various past population sizes.

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

In this case, the "past" and "present" population estimates are not from the same year for all subpopulations. Thus, it is necessary to calculate reduction for each subpopulation in the same time period. For example, it is necessary to project the population from the "past" census (in the 1930s) to 1941 (three generations ago) as well as from the most recent census (in 1995) to the present.

These calculations depend on the pattern of decline (see section 4.5.1). 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 Ocean" subpopulation has occurred in recent years, the estimate in the 1930s can be assumed to also represent the population in 1941 (three 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 10,000 to 7,000 occurred in 20 years, then assuming a constant rate of decline during this period, annual rate of decline can be calculated as 1.77% $[1-(7{,}000/10{,}000)^{(1/20)}]$, giving a projected reduction of about 10.1% in the six years from the last census (in 1995) to 2001, and a projected 2001 population of 6,290 $(=7{,}000*(7{,}000/10{,}000)^{(6/20)})$. This means a three-generation reduction of 37% (10,000 to 6,290).

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 $[-0.0448=(4{,}000/10{,}000)^{(1/20)}-1]$. Thus, three-generation decline can be estimated as 93.6% $[-0.936=(4{,}000/10{,}000)^{(60/20)}-1]$.

The "Atlantic Ocean" 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]. Note that it is incorrect to calculate a simple (unweighted) average of the 3-generation reduction amounts of the different subpopulations. As mentioned above, reductions of the different subpopulations must be weighted by their initial population sizes (i.e., for A1 and A2, the population size 3 generations ago).

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

Subpop.	**Past**	**Present**	**Notes**	**Population 3 gen. ago (calc*.)**	**Current population (calc*.)**	**Estimated 3-generation reduction**
Pacific Ocean	10,000 (1930s)	7,000 (1995)	Most of the decline in the last 20yr	10,000	6,290	37.1%
Atlantic Ocean	8,000 (1975)		Believed to have been stable	8,000	8,000	0%
Indian Ocean	10,000 (1961)	4,000 (1981)		25,000	1,600	93.6%
Overall				43,000	15,890	63.0%

*calc: calculated based on information in the previous columns

Example 3: Estimates are available for various past population sizes for some subpopulations only.

Subpopulation	**Past**	**Present**	**Reduction**	**Notes**
Pacific Ocean	unknown	5,000 (1990)	50%	suspected reduction over 3 generations
Atlantic Ocean	8,000 (1955)	9,000 (1998)		
Indian Ocean	unknown	2,000 (1980)	70%	inferred reduction over 3 generations

In this case, for some regions, there is no information about the past subpopulation size, but there is a suspected or inferred reduction. In this case, such suspected or inferred values must be averaged, weighted by the population size three generations ago. Since this number is not known, it must be projected using the present estimates and the inferred or suspected reduction amount, using the methods discussed under Example 2. Assuming exponential decline or growth, the table is completed as follows.

Subpop.	**Past**	**Present**	**Reduction**	**Population 3 gen. ago (calc.)**	**Current population (calc.)**	**3-generation change**
Pacific Ocean	?	5,000 (1990)	50% (suspected)	8,807[a]	4,403[a]	50% suspected reduction
Atlantic Ocean	8,000 (1955)	9,000 (1998)		7,699[b]	9,074[b]	17.9% estimated increase
Indian Ocean	?	2,000 (1980)	70% (inferred)	4,374[c]	1,312[c]	70% inferred reduction
Overall				20,880	14,789	29.2% reduction

[a] Annual proportional population change is 0.9885 $[=(1-0.5)^{(1/60)}]$, which is a 1.15% decrease per year. Population change from 1941 until the census in 1990 is 0.5678 $[=0.9885^{(1990-1941)}]$. Thus, population size in 1941 is 8,807 (5,000/0.5678). Population change from the census in 1990 to 2001 is 0.8807 $[=0.9885^{(2001-1990)}]$. Thus, population size in 2001 is 4,403 (5,000*0.8807).

[b] Population change from 1955 to 1998 is 1.125 (=9,000/8,000; 12.5% increase). Thus, annual change is 1.00274, or 0.27% increase per year $[=1.125^{1/(1998-1955)}]$. Population size in 1941 is 7,699 $[=8{,}000/1.00274^{(1955-1941)}]$. Population size in 2001 is 9,074 $[=9{,}000*1.00274^{(2001-1998)}]$.

[c] Annual population change is 0.9801 $[=(1-0.7)^{(1/60)}]$. Population change from 1941 until the census in 1980 is 0.4572 $[=0.9801^{(1980-1941)}]$. Thus, population size in 1941 is 4,374 (2,000/0.4572). Population change from the census in 1980 to 2001 is 0.6561 $[=0.9801^{(2001-1980)}]$. Thus, population size in 2001 is 1,312 (2,000*0.6561).

Example 4: Multiple estimates are available for various past population sizes.

Subpopulation	**Past-1**	**Past-2**	**Past-3**	**Present**
Pacific Ocean	10,000 (1935)	10,200 (1956)	8,000 (1977)	5,000 (1994)
Atlantic Ocean	8,000 (1955)			9,000 (1998)
Indian Ocean	13,000 (1946)	9,000 (1953)	5,000 (1965)	3,500 (1980)

In this case, as in example 2, the “past” and “present” population size estimates are not from the same year for all subpopulations. However, there are estimates for additional years, which provide information for making projections. For example, for the "Pacific Ocean" subpopulation, the annual rate of change has changed from a 0.09% increase in the first period (1935 to 1956) to a 1.15% decrease in the second and a 2.73% decrease in the third period, suggesting an accelerated decline. One option is to assume that the final rate of decline will apply from 1994 to 2001 as well. Another option is to perform a non-linear regression. For example, a 2nd degree polynomial regression on the natural logarithms of the four population estimates predicts population size as $exp(-1328+1.373t -0.0003524t^2)$, where t is year from 1935 to 2001. This equation gives a 1941 population of 10,389 and a 2001 population of 3,942, which correspond to a 62% reduction. The "Indian Ocean" subpopulation shows a different pattern; the annual rate of decline decelerates from 5.12% in the first period to 4.78% in the second and 2.35% in the third period. The same regression method predicts population size as $exp(2881-2.887t+0.0007255t^2)$, giving a 1941 subpopulation of 18,481 and a 2001 subpopulation of 3,538, which correspond to a 80.9% decline (thus, the regression has predicted a slight increase from 1980 to 2001). The completed table is below.

Subpop.	Past-1	Past-2	Past-3	Present (closest to 2001)	Population 3 gen. ago (1941; calc.)	Current population (2001; calc.)	Estimated 3-generation change
Pacific Ocean	10,000 (1935)	10,200 (1956)	8,000 (1977)	5,000 (1994)	10,389	3,942	62.1% reduction
Atlantic Ocean	8,000 (1955)			9,000 (1998)	7,699	9,074	17.9% increase
Indian Ocean	13,000 (1946)	9,000 (1953)	5,000 (1965)	3,500 (1980)	18,481	3,538	80.9% reduction
Overall					36,569	16,554	54.7% reduction

4.5.5 Dealing with uncertainty

In many cases, data from some or even most of the subpopulations (or regions) will be unavailable or uncertain. Even for taxa with very uncertain data, we recommend that the available data be organized in the same way as described above. Section 4.5.1 discusses how to calculate population sizes for the present and three generations ago.

Using uncertain estimates

Uncertain values can be entered as plausible and realistic ranges (intervals). In specifying uncertainty, it is important to separate natural (temporal or spatial) variability from uncertainty due to lack of information. Because criterion A refers to a specific period, temporal variability should not contribute to uncertainty. In other words, the uncertainty you specify should not include year-to-year variation. Criterion A refers to the overall reduction of the taxon, so spatial variability should not contribute to uncertainty. For example, if the reduction in different subpopulations ranges from 10% to 80%, this range ([10,80]%) should not be used to represent uncertainty. Instead, the estimated reduction in different subpopulations should be averaged as described above.

This leaves uncertainty due to lack of information, which can be specified by entering each estimate as an interval, as in the following table.

Subpopulation	**Past**	**Present**
Pacific Ocean	8,000 - 10,000 (1941)	4,000 - 6,000 (2001)
Atlantic Ocean	7,000 - 8,000 (1941)	8,000 - 10,000 (2001)
Indian Ocean	10,000 - 15,000 (1941)	1,500 - 2,500 (2001)

In this case, a simple approach is to calculate the minimum and maximum estimates for the reduction in each subpopulation using the lower and upper estimates[3]. For example, for the "Pacific Ocean" subpopulation, the minimum reduction can be estimated as a reduction from 8,000 to 6,000 (25%) and the maximum reduction can be estimated as 60% (from 10,000 to 4,000). If "best" estimates for past and present population sizes are also available, they can be used to estimate the best estimate for reduction. Otherwise, the best estimate for reduction can be estimated as 44% (9,000 to 5,000), using the midpoints of the intervals for the past and the present population sizes.

If similar uncertainty exists for all subpopulations (as in this example), a simple approach is to add all lower and all upper bounds of estimates. In this case, the total population size would be 25,000–33,000 in the past and 13,500–18,500 in the present. Using the same approach as outlined above, the best estimate of reduction can be calculated as 45% (29,000 to 16,000), with plausible range of reductions from 26% (from 25,000 to 18,500) to 59% (from 33,000 to 13,500).

An alternative method is to use a probabilistic (Monte Carlo) approach. If the uncertainty of past and present population sizes are given as probability distributions, and the correlation between these distributions are known, then the probability distribution for the reduction can be calculated by randomly selecting a pair of past and present population sizes (using the given distributions), calculating the reduction based on this pair, and repeating this with hundreds (or thousands) of randomly selected pairs.

Using data with different units

The examples discussed above assumed that the population data were in the same units (number of mature individuals). In some cases, data from different populations may be in different units (such as CPUE or other indices). In such cases, it is recommended that a separate table be prepared for each data type. If the past and current population sizes are in the same units for any subpopulation, they can be used to calculate (perhaps with extrapolation as discussed above) the reduction for that subpopulation. Such a calculation assumes that the index is linearly related to the number of mature individuals. The assessment should discuss the validity of this assumption, and make the necessary transformation (of the index to one that linearly relates to the number of mature individuals) before reduction is calculated (also see requirement (a) at the start of this section).

It is also important that an effort be made to combine the tables by converting all units to a common one. This is because it is necessary to know the relative sizes of the subpopulations to combine the reduction estimates, unless the subpopulations are known to be similar sizes or have declined by similar percentages. If the percent reduction is similar (within one or two percentage points) for different subpopulations, their relative sizes will not play an important role, and a simple (arithmetic) average can be used instead of a weighted average. If population sizes were known to be similar three generations ago (e.g., the smallest subpopulation was not any smaller than, say, 90% of the largest), again a simple average can be used.

---

[3] This is the method used in RAMAS Red List to calculate reduction based on abundances, when you click the "Calculate" button in the Value editor window for past or future reduction.

If population sizes and reduction amounts differ among subpopulations, then reductions (in percent) based on different units can be combined only if the relative sizes of the subpopulations can be estimated. However, this need not be a very precise calculation. Ranges (intervals) can be used to calculate uncertain results. For example, suppose that the estimates of reduction in two subpopulations are 60% and 80%, and that precise estimates of relative population sizes are not available (because these reduction estimates are based on different indices). In this case, crude estimates of relative sizes can be used. If the relative size of the first subpopulation is estimated to be between 0.40 and 0.70 of the total population, then the overall reduction can be calculated as follows. The high estimate would be (60%*0.4)+ (80%*0.6), or 72%. The low estimate would be (60%*0.7)+(80%*0.3), or 66%. Thus, the overall reduction can be expressed as the interval 66%–72%.

Using data from a few subpopulations

In some cases, reliable data exist from only one or few subpopulations. In such cases, the available data can be used under the following conditions.

1. If the subpopulation for which a reduction estimate is available was by far the largest subpopulation three generations ago, then this estimate can be used for the whole taxon. This process can also be formalized using the methods outlined above. For example, suppose that the largest subpopulation has declined by 60%, and that it had represented 90 to 99% of the mature individuals in the taxon three generations ago. If there is no information on the rest of the subpopulations (representing 1–10% of mature individuals), these subpopulations can be assumed to have declined by 0 to 100% (although, of course, this range does not include all the possibilities, as it excludes the possibility that the other subpopulations have increased). With these assumptions, the low estimate would be 54% (if the rest of the subpopulations had 10% of the individuals, and declined by 0%), and the high estimate would be 64% (if the rest of the subpopulations had 10% of the individuals and declined by 100%). Thus, the overall reduction can be expressed as the interval 54%–64%, which includes the estimate (60%) based on the largest subpopulation, but also incorporates the uncertainty due to lack of knowledge from other subpopulations.

2. If it can be assumed that all (or all the large) subpopulations are declining at the same rate, then the reduction estimated for a subset of the subpopulations can be used for the whole taxon. In this case, it is important to document any evidence that indicates that the rates are the same, and discuss and rule out various factors that may lead to different rates of reduction in different subpopulations.

4.5.6 Fluctuations vs. reduction

The downward phase of a fluctuation will not normally count as a reduction (section 4.5) or a continuing decline (section 4.6); therefore, an observed decline or reduction should not be considered a fluctuation unless there is evidence for this. When fluctuations occur at periods shorter than the assessment period (e.g., annual fluctuations over a 10-year/3-generation period), the methods described in section 4.5.1 can minimize the increase in the uncertainty of the calculated decline (Akçakaya *et al.* 2021).

However, fluctuations with periods similar to or longer than the assessment period may be difficult to distinguish from declines. In such cases, knowing the causes of the population changes (e.g., climatic fluctuations such as El Niño-Southern Oscillation or successional responses to disturbance regimes such as fires or floods) would help attribute the change to a fluctuation. If such verified causal information is not available, these long-term population changes should not be assumed to be part of a fluctuation; they should instead be interpreted as directional changes (population increases or declines).

In rare cases, population changes that occur as a result of cessation of human activities can be considered fluctuations. If a population had previously increased because of a human activity not related to conservation, and that activity has recently changed or stopped, resulting in a decline in the population, that decline can be considered as part of a fluctuation, if there is evidence that the population is returning to a pre-impact level. For example, over-fishing and discarding of fish at sea have artificially led to higher population sizes for some species (Wilhelm *et al.* 2016). As more sustainable fishing practices are adopted, these populations may decline to previous levels. Because the decline is reversing a previous human-caused increase that was not related to conservation, the decline can be considered as part of a fluctuation.

4.6 Continuing decline (criteria B and C)

“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 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, 2012b)

Continuing declines are used in two different ways in the criteria. Continuing declines at any rate can be used to qualify taxa under criteria B or C2. This is because taxa under consideration for criteria B and C are already characterized by restricted ranges or small population size. *Estimated continuing decline* (under criterion C1) has quantitative thresholds, and requires a quantitative estimate, which can be calculated using the same methods as for population reduction (see section 4.5). The concept of continuing decline at any rate is not applicable under criterion C1 (or under criterion A).

Under criteria B1b, B2b, and C2, continuing declines can be observed, estimated, inferred or projected. Although not explicitly mentioned in criteria B or C2, estimated continuing declines are permissible. Under criterion C1, continuing declines can only be observed, estimated or projected. A continuing decline under criteria B or C can be projected, thus, it does not have to have started yet. However, such projected declines must be justified and there must be high degree of certainty that they will take place (i.e., merely 'plausible' future declines are not allowed).

Rates of continuing decline over long generation times (in the same way as reductions) may be estimated from data over shorter time frames. For example, evaluating a taxon under criterion C1 for the Vulnerable category requires estimating a continuing decline for three generations or 10 years, whichever is longer (up to a maximum of 100 years). When extrapolating data from shorter time frames, assumptions about the rate of decline remaining constant, increasing or decreasing, relative to the observed interval must be justified with reference to threatening processes, life history or other relevant factors.

Note that a continuing decline is not possible without a population reduction (which, however, may not be large enough to meet any thresholds under criterion A), but a reduction is possible without a continuing decline: if a reduction has ‘ceased’ under criterion A, there cannot be a continuing decline. However, continuing declines need not be continuous; they can be sporadic, occurring at unpredictable intervals, but they must be likely to continue into the future. Relatively rare events can be considered to contribute to a continuing decline if they happened at least once within the last three generations or 10 years (whichever is longer), and it is likely that they may happen again in the next three generations or 10 years (whichever is longer), and the population is not expected to recover between the events.

A potentially confusing aspect of the criteria is that “estimated continuing decline” under criterion C1 is conceptually very similar to “moving window reduction” under criterion A4. The differences are (i) criterion A4 is always evaluated for three generations/10 years, whereas criterion C1 is evaluated for one, two or three generations, depending on the category, (ii) the thresholds are lower under criterion C1 (e.g., for VU, 10% under criterion C1 and 30% under criterion A4), (iii) criterion C1 also requires small population size, and (iv) under criterion C1, the decline must be observed, estimated or projected, whereas under criterion A4, the reduction can be observed, estimated, inferred, projected or suspected.

If habitat is declining (in area or quality) but abundance is not, this may be because (i) there is a delay in the population's response to lower carrying capacity, perhaps because the population is below the carrying capacity for other reasons (such as harvest), (ii) habitat is declining in areas not currently occupied by the taxon, or (iii) habitat is not correctly identified. In the case of (i), the population will eventually be impacted; in the case of (ii) the loss of recolonization options may eventually impact the population. In both cases, criteria B1b(iii) or B2b(iii) may be invoked even if the population is not undergoing a continuing decline. Incorrect habitat identification (case iii) can be resolved using a more precise definition of "habitat." When determining continuing decline in area, extent and/or quality of habitat (criteria B1b(iii) and B2b(iii)), assessors should define "habitat" in the strict sense, i.e., as the area, characterized by its abiotic and biotic properties, that is habitable by a particular species. In particular, they should avoid using generic classifications such as "forest" that indicate a biotope, a vegetation type, or a land-cover type, rather than a species-specific identification of habitat. In addition, they should document the location of declines in relation to the species' range, and if possible, quantify the proportion of the range affected, the magnitude or rate of the decline, and how the species is responding to the decline.

Note that continuing decline is different from "current population trend", which is a required field in IUCN Red List assessments, but not used when applying the criteria. There is not a simple correspondence between these two terms. The current population trend may be stable or increasing, with a continuing decline projected in the future. If the current population trend is declining, then there is continuing decline, but only if the trend is liable to continue into the future and it is not the declining phase of a fluctuation.

4.7 Extreme fluctuations (criteria B and C2)

“Extreme fluctuations can be said to occur in a number of taxa where population size or distribution area varies widely, rapidly and frequently, typically with a variation greater than one order of magnitude (i.e., a tenfold increase or decrease).” (IUCN 2001, 2012b)

Extreme fluctuations are included in criteria B and C 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.

Population fluctuations may vary in magnitude and frequency (Figure 4.4). For the ‘extreme fluctuations’ subcriterion to be invoked, populations would 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. 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).

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 subpopulations.

If there is regular or occasional dispersal (of even a small number of individuals, seeds, spores, etc.) between all (or nearly all) of the subpopulations, then the degree of fluctuations should be measured over the entire population. In this case, the subcriterion 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 subpopulations 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 subpopulations are totally isolated, the degree of synchrony between the populations is not as important and it is sufficient that a majority of subpopulations each show extreme fluctuation to meet the subcriterion. In this case, if most of the subpopulations show fluctuations of an order of magnitude, then the criterion 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 subpopulations, then the total population size over these connected subpopulations should be considered when assessing fluctuations; each set of connected subpopulations should be considered separately.

Population fluctuations may be difficult to distinguish from directional population changes, such as continuing declines, reductions or increases. Figure 4.4 shows 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: (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 recurring pattern of increases and decreases (Figure 4.4). 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 particular 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. 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. 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.

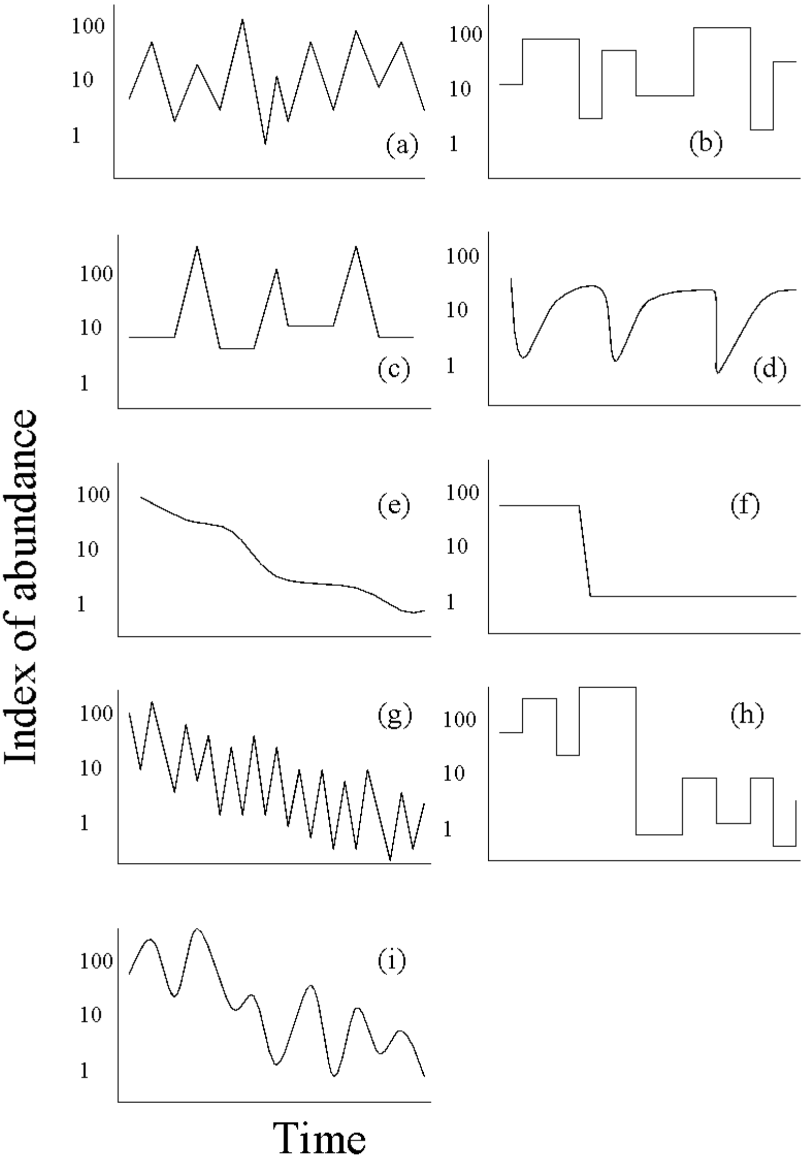


Figure 4.4. Fluctuations without directional change in population size (a–d), population reductions or declines without fluctuations (e, f), population reductions in combination with fluctuations (g-i).

4.8 Severely fragmented (criterion B)

"The phrase 'severely fragmented' refers to the situation in which increased extinction risks to the taxon results from the fact that most of its individuals are found in small and relatively isolated subpopulations (in certain circumstances this may be inferred from habitat information). These small subpopulations may go extinct, with a reduced probability of recolonization." (IUCN 2001, 2012b)

In the IUCN Red List Criteria, the term 'severely fragmented' refers to fragmentation of the population, which often results from, but is different to, habitat fragmentation. The two attributes of subpopulations of a severely fragmented taxon mentioned in the above quote (small and isolated) must be assessed by considering taxon-specific characteristics. However, as stated in the definition (and discussed below), for criterion B, population fragmentation can be inferred from habitat fragmentation. When making this inference, species-specific information (such as dispersal distances and home range sizes) should be used, as discussed below, whenever available; and "habitat" should be defined in the strict sense, i.e., as the area, characterized by its abiotic and biotic properties, that is habitable by the species being assessed. In particular, habitat should not be defined in the generic sense (e.g., as a biotope, a vegetation type, or a land-cover type).

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. Thus, the same degree of habitat fragmentation may not lead to the same degree of population fragmentation for species with different levels of mobility. 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. If natural habitats have been fragmented (e.g., old growth forests and rich fens), this can be used as direct evidence for fragmentation for taxa with poor dispersal ability.

Similarly, fragmentation must be assessed at a scale that is appropriate to population densities of the taxon under consideration. All else being equal, the same level of habitat fragmentation will more likely lead to severe fragmentation for a species with lower population densities, because each habitat fragment will be more likely to have a small number of individuals.

The following criterion can be used to decide whether there is severe fragmentation in cases where data are available on (i) the distribution of area of occupancy (i.e., detailed maps 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.), then: A taxon 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 dispersal kernel of the species (see below).

Criterion B is often used in the absence of any information on population size, density or structure. Therefore, for (1), the area for a viable population (or, the interpretation of "small" in the definition of severely fragmented) should be based on rudimentary and generic estimates of population density, and on the ecology of the taxon. For example, for many vertebrates, subpopulations of fewer than 100 individuals may be considered too small to be viable. For (2), the degree of isolation of patches should be based on dispersal distance of the taxon. For example, subpopulations that are isolated by distances several times greater than the (long-term) average dispersal distance of the taxon may be considered isolated. On the other hand, separation of subpopulations by non-habitat areas (e.g., islands in an archipelago) does not necessarily mean isolation, if the taxon can disperse between the subpopulations.

Note that the existence (or even a large number) of small and isolated patches is not sufficient to consider the taxon severely fragmented. For meeting this criterion, more than half of the individuals (or, more than half of the occupied habitat area) must be in small and isolated patches. On the other hand, a taxon with a single subpopulation can also be severely fragmented, if that subpopulation is too small to be viable (because a single population is by definition isolated). Similarly, a taxon with two subpopulations can be severely fragmented if they are isolated from each other, and both are too small to be viable.

For many taxa, the information on population density and dispersal distance can be based on 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. For bryophytes, it is recommended that in most circumstances, a minimum distance greater than 50 km between subpopulations of taxa without spore dispersal can indicate severe fragmentation, and a distance of between 100 km and 1,000 km for taxa with spores (Hallingbäck *et al.* 2000).

The definition of severe fragmentation is based on the distribution of subpopulations. This is often confused with the concept of "location" (see section 4.11), but is independent of it. A taxon may be severely fragmented, yet all the isolated subpopulations may be threatened by the same major factor (single location), or each subpopulation may be threatened by a different factor (many locations). Also, severe fragmentation does not require an ongoing threat; small and isolated subpopulations of a severely fragmented taxon can go extinct due to natural, stochastic (demographic and environmental) processes.

4.9 Extent of occurrence (criteria A and B)

Extent of occurrence is defined as "the area contained within the shortest continuous imaginary boundary which can be drawn to encompass all the known, inferred or projected sites of present occurrence of a taxon, excluding cases of vagrancy" (IUCN 2001, 2012b).

Extent of occurrence (EOO) is a parameter that measures the spatial spread of the areas currently occupied by the taxon. The intent behind this parameter is to measure the degree to which risks from threatening factors are spread spatially across the taxon's geographical distribution. The theoretical basis for using EOO as a measure of risk spreading is the observation that many environmental variables and processes are spatially correlated, meaning that locations that are close to each other experience more similar (more correlated) conditions over time than locations that are far away from each other. These processes include both human threats (such as diseases, invasive species, oil spills, non-native predators, habitat loss to development, etc.) and natural processes (fluctuations in environmental variables such as droughts, heat waves, cold snaps, hurricanes and other weather events, as well as other disturbance events such as fires, floods, and volcanism). Higher correlation leads to higher overall extinction risk, so that, all other things being equal, a set of populations spread in a small area have higher extinction risk overall than a set of populations spread over a larger area.

EOO is not intended to be an estimate of the amount of occupied or potential habitat, or a general measure of the taxon's range. Other, more restrictive definitions of "range" may be more appropriate for other purposes, such as for planning conservation actions. Valid use of the criteria requires that EOO is estimated in a way that is consistent with the thresholds set therein.

In thinking about the differences between EOO and AOO (area of occupancy; discussed in section 4.10), it may be helpful to compare species that have similar values for one of these spatial metrics and different values for the other. All else being equal, larger EOOs usually result in a higher degree of risk spreading (and hence a lower overall risk of extinction for the taxon) than smaller EOOs, depending on the relevant threats to the taxa. For example, a taxon with occurrences distributed over a large area is highly unlikely to be adversely affected across its entire range by a single fire because the spatial scale of a single occurrence of this threat is narrower than the spatial distribution of the taxon. Conversely, a narrowly distributed endemic taxon, with the same

AOO as the taxon above, may be severely affected by a fire across its entire EOO because the spatial scale of the threat is larger than, or as large as, the EOO of the taxon.

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.

If EOO is less than AOO, EOO should be changed to make it equal to AOO to ensure consistency with the definition of AOO as an area within EOO.

"Extent of occurrence can often be measured by a minimum convex polygon (the smallest polygon in which no internal angle exceeds 180 degrees and which contains all the sites of occurrence)" (IUCN 2001, 2012b). The IUCN Red List Categories and Criteria state that EOO may exclude "discontinuities or disjunctions within the overall distribution of the taxa". However, for assessments of criterion B1, exclusion of areas forming discontinuities or disjunctions from estimates of EOO is strongly discouraged. Exclusions are not recommended for criterion B1, 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. Inappropriate exclusions of discontinuities or disjunctions within the overall distribution of a taxon will underestimate EOO for the purpose of assessing criterion B and consequently will underestimate the degree to which risk is spread spatially for the taxon.

When there are such discontinuities or disjunctions in a species distribution, the minimum convex polygon (also called the convex hull) yields a boundary with a very coarse level of resolution on its outer surface, resulting in a substantial overestimate of the range, particularly for irregularly shaped ranges (Ostro *et al.* 1999). The consequences of this bias vary, depending on whether the estimate of EOO is to be used for assessing the spatial thresholds in criterion B or whether it is to be used for estimating or inferring reductions (criterion A) or continuing declines (criteria B and C). The use of convex hulls is unlikely to bias the assessment of EOO thresholds under criterion B, because disjunctions and outlying occurrences often do contribute to the spatial spread of risk (see above). This is also true for "doughnut distributions" (e.g. aquatic species confined to the margins of a lake) and elongated distributions (e.g., coastal species). In the case of species with linear elongated distributions, minimum convex polygon may lead to an overestimate of extinction risk. Nevertheless, given the paucity of practical methods applicable to all spatial distributions, and the need to estimate EOO consistently across taxa, minimum convex polygon remains a pragmatic measure of the spatial spread of risk.

However, the bias associated with estimates based on 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 generalization 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 and 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. For α-hulls the estimate of area and trend in area also converges on the correct value as sample size increases, unless other errors are large.

This does not necessarily hold for convex hulls. Kernel estimators may be used for the same purpose but their application is more complex.

To estimate an α-hull, the first step is to make a Delauney triangulation of the mapped points of occurrence (Figure 4.5). The triangulation is created by drawing lines joining the points, constrained so that no lines intersect between points. The outer surface of the Delauney triangulation is identical to the convex hull.

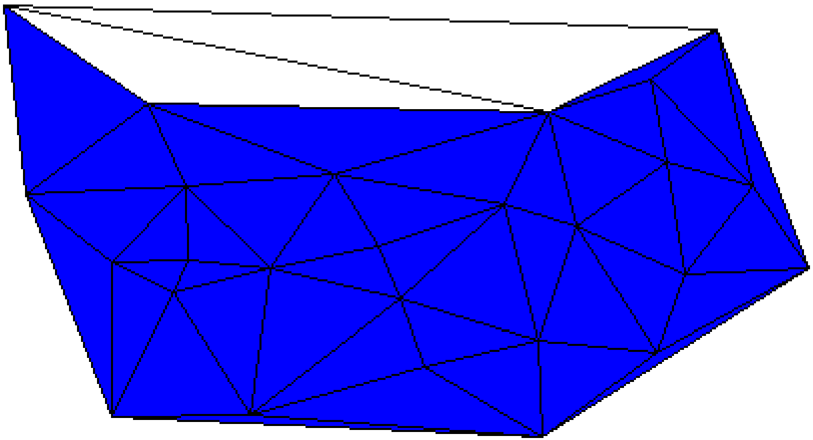


Figure 4.5. Illustration of α-hull. The lines show the Delauney triangulation (the intersection points of the lines are the taxon's occurrence locations). The sum of the areas of darker triangles is EOO based on the α-hull. The two lighter coloured triangles that are part of the convex hull are excluded from the α-hull.

The second step is to measure the lengths of all of the lines, and calculate the average line length. The third step is to delete all lines that are longer than a multiple (α) of the average line length. (This product of α and the average line length represents a "discontinuity distance".) The value of α can be chosen with a required level of resolution in mind. The smaller the value of α, the finer the resolution of the hull. Experience has shown that an α value of 2 is a good starting point for some species (however, the value to use for specific cases of assessing reductions in EOO should be based on a compromise between minimizing the potential bias associated with incomplete sampling of outlying occurrences and minimizing the departure from a convex hull). This process results in the deletion of lines joining points that are relatively distant, and may subdivide the total range into more than one polygon. The final step is to calculate the extent of occurrence by summing the areas of all remaining triangles. When this exercise is repeated to estimate EOO from a second temporal sample of points (and hence assess change in EOO), the same discontinuity distance between points should be used as a threshold for deleting lines (rather than the same value of α). This will reduce bias due to variation in sampling effort between the two surveys and the bias due to changing average line length with more or fewer occurrences.

Extent of occurrence and area of occupancy are measures of the current distribution, i.e. they should not include areas where the species no longer exists. On the other hand, these measures should not only include the actually known sites, but also inferred or projected sites (see section 4.10.7). For instance, sites can be inferred from presence of known appropriate habitat, but where the species has not yet been searched for. In doing so, it will be important to judge to what extent the taxon has been looked for. Incorporating inferred sites results in a range of plausible values, which may give a range of plausible Red List Categories (see sections 3.1 on *Data availability, inference and projection*, and 3.2 on *Uncertainty*).

4.10 Area of occupancy (criteria A, B and D)

Area of occupancy (AOO) is a scaled metric that represents the area of suitable habitat currently occupied by the taxon. Area of occupancy is included in the criteria for two main reasons. The first role of AOO is as a measure of the 'insurance effect' (Keith *et al.* 2018), whereby taxa that occur within many patches or large patches across a landscape or seascape are 'insured' against risks from spatially explicit threats. In such cases, there is only a small risk that the threat will affect all occupied patches within a specified time frame. In contrast, taxa that occur within few small patches are exposed to elevated extinction risks because there is a greater chance that one or few threats will affect all or most of the distribution within a given time frame. Thus, AOO is inversely related to extinction risk. Species at high risk because of their small AOO are often habitat specialists. Secondly, there is generally a positive correlation between AOO and population size. The veracity of this relationship for any one species depends on spatial variation in its population density (Gaston 1996). Nonetheless, AOO can be a useful metric for identifying species at risk of extinction because of small population sizes when no data are available to estimate population size and structure (Keith 1998).

As with EOO, in the case of migratory species, AOO 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 valid use of the criteria and maintain consistency of Red List assessments across taxa it is essential to scale estimates of AOO using 2 × 2 km grid cells. Estimates of AOO are highly sensitive to the spatial scale at which AOO is measured (Figure 4.6 below, Hartley and Kunin 2003, Nicholson *et al*. 2009). Thus, it is possible to arrive at very different estimates of AOO from the same distribution data if they were calculated at different scales (see 4.10.1 "Problems of scale" and Figure 4.6 below). The resolution (grid size) that maximizes the correlation between AOO and extinction risk is determined more by the spatial scale of threats than by the spatial scale at which AOO is estimated or shape of the taxon's distribution (Keith *et al.* 2018). The thresholds of AOO that delineate different categories of threat in criteria B2 and D2 are designed to assess threats that affect areas in the order of 10 – 2,000 km^2, and therefore assume that AOO is estimated at a particular spatial scale. These Guidelines require that AOO is scaled using 2 × 2 km grid cells (i.e., with area of 4 km^2) to ensure that estimates of AOO are commensurate with the implicit scale of the thresholds. Use of the smallest available scale (finest grain) to estimate AOO (sometimes erroneously called "actual area" or "actual AOO") is not permitted, even though mapping a species' distribution at the finest scale may be desirable for purposes other than calculating AOO. It should be noted that the scaling estimates of AOO to a standard spatial grain in criteria B2 and D2, and scaling of rates of population decline by generation length in criterion A, are both essential procedures to promote consistency in Red List assessments. The scale requirement only applies to calculation of AOO because of its role as a measure of insurance effects on extinction risk, rather than a precise measure of occupied habitat area (Keith *et al.* 2018). Habitat maps with higher resolutions can be used for other aspects of a Red List assessment, such as calculating reduction in habitat quality as a basis of population reduction for criterion A2(c) or estimating continuing decline in habitat area for B2(b), as well as for conservation planning.

Recognizing the role of AOO and the importance of valid scaling, IUCN (2001, 2012b) includes the following text: "Area of occupancy is defined as the area within its 'extent of occurrence' (see 4.9 above), which is occupied by a taxon, excluding cases of vagrancy. The measure reflects the fact that a taxon will not usually occur throughout the area of its extent of occurrence, which may contain unsuitable or unoccupied habitats. 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. The size of the area of occupancy will be a function of the scale at which it is measured, and should be at a scale appropriate to relevant biological aspects of the taxon, the nature of threats and the available data (see below). To avoid inconsistencies and bias in assessments caused by estimating area of occupancy at different scales, it may be necessary to standardize estimates by applying a scale-correction factor. It is difficult to give strict guidance on how standardization should be done because different types of taxa have different scale-area relationships."

4.10.1 Problems of scale

Red List assessments based on the area of occupancy (AOO) may be complicated by problems of spatial scale. Estimating the quantity of occupied habitat for taxa with markedly different body sizes, mobility and home ranges intuitively requires different spatial scales of measurement. Nevertheless, many of the major threats that impact those same taxa operate at common landscape and seascape scales. For this reason, the Red List Criteria specify fixed range size thresholds to identify taxa at different levels of extinction risk. The use of fixed range size thresholds is also important for pragmatic reasons to maintain the parsimony of the Red List Criteria. Use of different thresholds for different groups of taxa would greatly amplify the complexity of the criteria and guidelines, as well as the risks of inconsistent applications.

The need to scale estimates of AOO consistently follows logically from the adoption of fixed AOO thresholds in the Red List criteria and the sensitivity of AOO estimates to measurement scale. "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, 2012b).

The following sections first describe a simple method of estimating AOO, then specify the appropriate reference scale, and finally we describe a method of standardization (or scaling) for cases where the available data are not at the reference scale.

4.10.2 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 taxon (see Figure 2 in IUCN 2001, 2012b), and then tallying the total area of all occupied cells:

$$\text{AOO} = \text{no. occupied cells} \times \text{area of an individual cell} \qquad \text{(equation 4.1)}$$

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, but here we use area). 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 because it is more difficult for different assessors to produce consistent estimates using such approaches.

If different grid locations (starting points of the grid) result in different AOO estimates, the minimum estimate should be used.

4.10.3 The appropriate scale

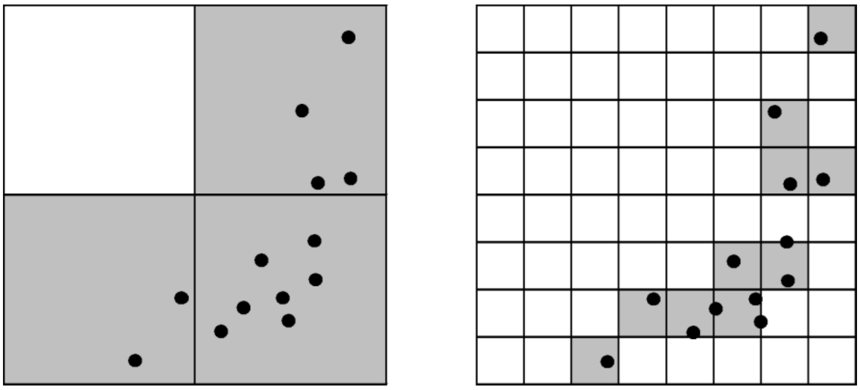
In all cases, 4 km^2 (2 × 2 km) cells are recommended as the reference scale for estimating AOO to assess criteria B2 and D2. If an estimate was made at a different scale, especially if data at different scales were used in assessing species in the same taxonomic group, this may result in inconsistencies and bias (Keith *et al.* 2018). Scales of 3.2 × 3.2 km grid size or coarser (larger) are inappropriate because they do not allow any taxa to be listed as Critically Endangered (where the threshold AOO under criterion B is 10 km^2). Scales finer (smaller) than 2 × 2 km grid size tend to list more taxa at higher threat categories than the definitions of these categories imply. Assessors should avoid using estimates of AOO at other scales. 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 (see below).

If AOO can be calculated directly at the reference scale of 4 km^2 (2 × 2 km) cells, you can skip sections 4.10.4 and 4.10.5. If AOO cannot be calculated at the reference scale (e.g., because it has already been calculated at another scale and original maps are not available), then the methods described in the following two sections may be helpful.

4.10.4 Scale-area relationships

The biases caused by use of range estimates made at different scales may be reduced by standardizing estimates to a reference scale that is appropriate to the thresholds in the criteria. This and the following section discuss the scale-area relationship that forms the background for these standardization methods, and describe such a method with examples. The method of standardization depends on how AOO is estimated. In the following discussion, we assume that AOO was estimated using the grid method summarized above.

The standardization or correction method we will discuss below relies on the relationship of scale to area, in other words, how the estimated AOO changes as the scale or resolution changes. Estimates of AOO may be calculated at different scales by starting with mapped locations at the finest spatial resolution available, and successively doubling the dimensions of grid cells. The relationship between the area occupied and the scale at which it was estimated may be represented on a graph known as an area-area curve (e.g., Figure 4.6). The slopes of these curves may vary between theoretical bounds, depending on the extent of grid saturation. A theoretical maximum slope = 1 is achieved when there is only one occupied fine-scale grid cell in the landscape (fully unsaturated distribution). A theoretical minimum slope = 0 is achieved when all fine-scale grid cells are occupied (fully saturated distribution).



grid length	grid area	AOO
1	1	10
2	4	24
4	16	48
8	64	64
16	256	256
32	1,024	1,024

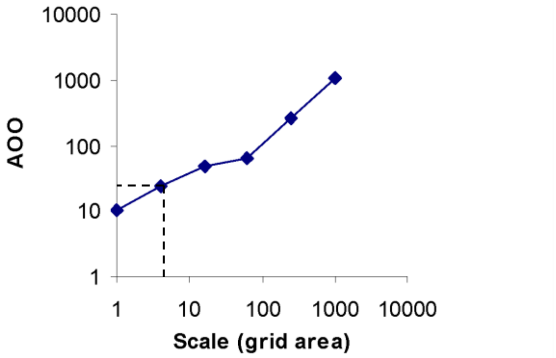


Figure 4.6. Illustration of scale-dependence when calculating area of occupancy. At a fine scale (map on right) AOO = 10 x 1 = 10 units^2 (based on Equation 4.1). At a coarse scale (map on left) AOO = 3 x 16 = 48 units^2. AOO may be calculated at various scales by successively doubling grid dimensions from estimates at the finest available scale (see Table). These may be displayed on an area-area curve (above).

4.10.5 Scale correction factors

Estimates of AOO may be standardized by applying a scale-correction factor. Scale-area relationships (e.g., Figure. 4.6) 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 the reference scale (i.e., 2 × 2 km grid size) that is appropriate to the area of occupancy thresholds in criterion B2. 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 B2.

Example: Scaling Up

Assume that estimates of AOO are available at 1 × 1 km grid resolution shown in Figure 4.6 (right) and that it is necessary to obtain an estimate at the reference scale represented by a 2 × 2 km grid. This may be done cartographically by simply doubling the original grid dimensions, counting the number of occupied cells and applying equation 4.1. When the reference scale is not a geometric multiple of the scale of the original estimate, it is necessary to calculate an area-area curve, as shown in Figure 4.6, and interpolate an estimate of AOO at the reference scale. This can be done mathematically by calculating a scale correction factor (C) from the slope of the area-area curve as follows (in all equations below, "log" means logarithm to base 10):

$$C = \log(AOO_2/AOO_1) / \log(Ag_2/Ag_1) \qquad \text{(equation 4.2)}$$

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 2 as follows:

$$AOO_R = AOO_1 * 10^{C*\log(AgR / Ag1)}, \text{ or } \quad AOO_R = AOO_2 * 10^{C*\log(AgR / Ag2)} \qquad \text{(equation 4.3)}$$

In the example shown in Figure 4.6, estimates of AOO from 1 × 1 km and 4 × 4 km grids may be used to verify the estimate AOO at the reference scale of 2 × 2 km as follows:

$C = \log(48/10) / \log(16/1) = 0.566$, and using equation 4.3 with this value of C, the AOO estimate at the larger scale (AOO_2=48), and the grid sizes at the larger and reference scales (Ag_R=4; Ag_2=16), the AOO estimate at the reference scale is calculated as:

$$AOO = 48 * 10^{0.566*\log(4/16)} = 22 \text{ km}^2$$

Note that this estimate differs slightly from the true value obtained from grid counting and equation 1 (24 km^2) because the slope of the area-area curve is not exactly constant between the measurement scales of 1 × 1 km and 4 × 4 km.

Example: Scaling Down

Scaling down estimates of AOO is more difficult than scaling up because there is no quantitative information about grid occupancy at scales finer than the reference scale. Scaling therefore requires extrapolation, rather than interpolation of the area-area curve. Kunin (1998) and He and Gaston (2000) suggest mathematical methods for this. A simple approach is to apply equation 4.3 using an approximated value of C.

An approximation of C may be derived by calculating it at coarser scales, as suggested by Kunin (1998). For example, to estimate AOO at 2 × 2 km when the finest resolution of available data is at 4 x 4 km, we could calculate C from estimates at 4 × 4 km and 8 × 8 km as follows.

$$C = \log(64/48) / \log(64/16) = 0.208$$

However, this approach assumes that the slope of the area-area curve is constant, which is unlikely to hold for many taxa across a moderate range of scales. In this case, AOO at 2 × 2 km is overestimated because C was underestimated.

$$AOO = 48 * 10^{0.208*\log(4/16)} = 36 \text{ km}^2.$$

While mathematical extrapolation may give some guidance in estimating C, there may be qualitative information about the dispersal ability, habitat specificity and landscape patterns that could also provide guidance. Table 4.1 gives some guidance on how these factors may influence the values of C within the range of scales between 2 × 2 km and 10 × 10 km grid sizes.

Table 4.1. Characteristics of organisms and their habitat that influence the slope of the scale-area relationship, and hence the scale-correction factor, C, within the range of spatial scales represented by 2 × 2 km and 10 × 10 km grid cells.

Biological characteristic	Influence on C	
	small (approaching 0)	large (approaching 1)
Dispersal ability	Wide	localized or sessile
Habitat specificity	Broad	Narrow
Habitat availability	Extensive	Limited

For example, if the organism under consideration was a wide-ranging animal without specialized habitat requirements in an extensive and relatively uniform landscape (e.g., a species of camel in desert), its distribution at fine scale would be relatively saturated and the value of C would be close to zero. In contrast, organisms that are either sessile or wide ranging but have specialized habitat requirements that only exist in small patches within the landscape (e.g., migratory sea birds that only breed on certain types of cliffs on certain types of islands) would have very unsaturated distributions represented by values of C close to one. Qualitative biological knowledge about organisms and mathematical relationships derived from coarse-scale data may thus both be useful for estimating a value of C that may be applied in equation 4.3 to estimate AOO at the reference scale. Uncertainty in the value of C can be represented through the use of interval or fuzzy arithmetic to propagate uncertainty through the assessment as described in section 3.2.

Finally, it is important to note that if unscaled estimates of AOO at scales larger than the reference value are used directly to assess a taxon against thresholds in criterion B, then the assessment is assuming that the distribution is fully saturated at the reference scale (i.e., assumes C = 0). In other words, the occupied coarse-scale grids are assumed to contain no unsuitable or unoccupied habitat that could be detected in grids of the reference size (see Figure 4.7).

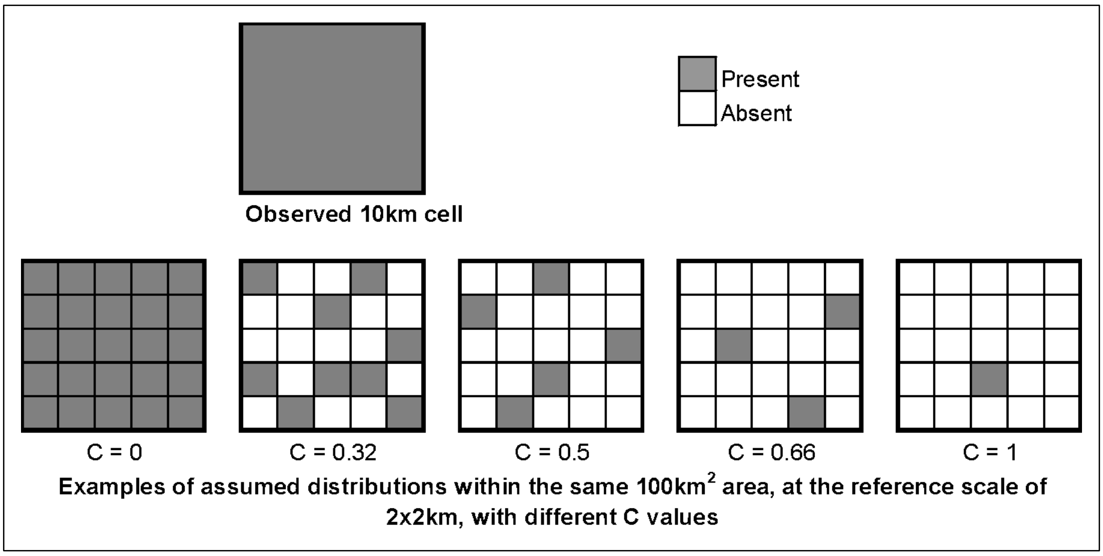


Figure 4.7. Demonstration of the consequences of different assumed C values. The available map is at 10 x 10 km resolution, so a presence observed at this scale corresponds to 25 cells at the reference scale of 2 x 2 km. Assuming C = 0 (i.e., using the unscaled estimate directly as AOO) assumes that all of these 25 cells are occupied. At the other extreme, a value of C = 1 assumes that only one 2 x 2 km cell is occupied.

4.10.6 "Linear" habitat

There is a concern that grids do not have much ecological meaning for taxa living in "linear" habitat such as in rivers or along coastlines. Although this concern is valid, for the purpose of assessing taxa against criterion B, it is important to have a measurement system that is consistent with the thresholds, and that leads to comparable listings. If AOO estimates were based on estimates of length x breadth of habitat, there may be very few taxa that exceed the VU threshold for criterion B2 (especially when the habitats concerned are streams or beaches a few metres wide). In addition, there is the problem of defining what a "linear" habitat is, and measuring the length of a jagged line. Thus, we recommend that the methods described above for estimating

AOO should be used for taxa in all types of habitat distribution, including taxa with linear ranges living in rivers or along coastlines.

4.10.7 AOO and EOO based on habitat maps and models

Both AOO and EOO may be estimated based on "…known, inferred or projected sites of present occurrences…" (IUCN 2001). In this case, 'known' refers to confirmed extant records of the taxon; 'inferred' refers to the use of information about habitat characteristics, dispersal capability, rates and effects of habitat destruction and other relevant factors, based on known sites, to deduce a very high likelihood of presence at other sites; and 'projected' refers to spatially predicted sites on the basis of habitat maps or models, subject to the three conditions outlined below.

Habitat maps show the distribution of potential habitat for a species. They may be derived from interpretation of remote imagery and/or analyses of spatial environmental data using simple combinations of GIS data layers such as land-cover and elevation (Brooks *et al*. 2019), or by more formal statistical habitat models (e.g., generalized linear and additive models, decision trees, Bayesian models, regression trees, etc.). These habitat models are also referred to as ecological niche models, species distribution models, bioclimatic models and habitat suitability models. 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. They cannot be used directly to estimate a taxon's AOO or EOO because they often map an area that is larger than the occupied habitat (i.e., they also map areas of potential habitat that may presently be unoccupied). However, they may be a useful means of estimating AOO or EOO indirectly, provided the three following conditions are 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 *potential* habitat must be interpreted to produce an estimate of the area of *occupied* habitat.
iii) For AOO, the estimated area of occupied habitat derived from the map must be scaled to the reference scale (see section 4.10). For EOO, the occupied habitat areas must be used to estimate the area of the minimum convex polygon (see section 4.9).

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 potential habitat, irrespective of altitude. The spatial resolution of habitat resources also affects how well maps can represent habitat. For example, specialized 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 or modelling at coarse scales. Any application of habitat maps to Red List assessments 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 potential habitat. A critical evaluation of condition (i) should include both biological and statistical considerations. For example, the selection of predictor variables should be based on knowledge of the biology of the species and not simply fitted statistically from a pool of candidate variables that are conveniently available. Statistically, appropriate methods of model evaluation should be employed (e.g., cross validation). See section 12.1.12.

Habitat maps may accurately reflect the potential habitat, but only a fraction of potential habitat may be occupied (condition ii). Conversely, depending on survey effort, the set of 'known' occurrences may underestimate the area of occupied habitat (Anderson 2023; see section 4.10.8).

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 or EOO, and will therefore need to be adjusted (using an estimate of the proportion of habitat occupied) to produce a valid estimate. 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. To determine what portions of predicted potential habitat should be identified as 'projected' sites that may be used to estimate AOO and EOO, assessors should consider which sites are very likely to be occupied based on: predicted habitat suitability values; ecologically relevant characteristics of the locality; the taxon's dispersal capability; potential dispersal barriers; physiological and behavioural characteristics of the taxon; proximity to confirmed records; survey intensity; the effect of predators, competitors or pathogens in reducing the occupied fraction of available habitat; and other relevant factors.

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 section 4.10.5). In other words, the area of potential habitat (also called area of habitat, AOH, or extent of suitable habitat, ESH) measured at a finer scale (higher resolution) than 2 × 2 km grid cells, even after correction for occupancy (due to a taxon not occupying all of the suitable habitat identified), cannot be used directly to compare against AOO thresholds, and certainly not against EOO thresholds. For AOO the area needs to be measured at the reference scale (see section 4.10.5), and for EOO the area must be used to calculate the minimum convex polygon that includes all the identified habitat areas (see section 4.9).

In those cases where AOO is less than the area of potential habitat, the population may be declining within the habitat, but the habitat may show no indication of change. Hence this method could be both inaccurate and non-precautionary for estimating reductions in population change.

However, if a decline in mapped habitat area is observed (and the map is a reasonable representation of potential 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. Thus, if estimates of AOO are not available, then the observed decline in mapped habitat area can be used to invoke "continuing decline" in criterion B, and the rate of such decline can be used as a basis for calculating a lower bound for population reduction under criterion A. Observed decline in mapped area can be used to invoke "continuing decline" in criterion C2 if the relationship between habitat area and the number of mature individuals was known (and positive).

4.10.8 Effect of sampling effort and detectability on estimates of AOO

Estimates of AOO may be sensitive to sampling effort, as may estimates of EOO, the number of locations and the number of subpopulations. Inevitably, a taxon may not have been detected everywhere that it occurs, either because it has cryptic life forms, short-lived detectable life stages, is hard to identify (and few capable experts available), or because it occurs in inaccessible or poorly surveyed regions. For conspicuous taxa occurring in well-sampled areas, it may be reasonable to assume that most occurrences have been detected and AOO may be estimated by tallying the area of 2 × 2 km grid cells in which observation records are located using Equation 4.1. For other taxa that may have many unrecorded occurrences, however, this assumption and the resulting tally will underestimate AOO.

Underestimation of AOO will affect the outcome of Red List assessments under criterion B2, e.g. if the estimated AOO is less than, or close to, 2,000 km^2, the lower threshold of the VU category. In such cases, assessors may not be able to justify the assumption that AOO is estimated accurately from a simple intersection of current records with a standard 2 × 2 km grid, and an alternative assumption must be made in support of a more accurate estimate.

Assessors should follow section 3.2 to deal with uncertainty in estimates of AOO for potentially threatened taxa that have poorly sampled distributions. A plausible lower bound of AOO would be no smaller than that based on an intersection of current records with a 2 × 2 km grid, but could be larger. A plausible upper bound of AOO would be no larger than that based on an intersection of potential habitat (given that it is well known) with a 2 × 2 km grid, but will usually be smaller because the taxon may not occupy all of its suitable habitat. Both of these bounds must be considered (e.g., by entering a range of values in SIS); assuming that AOO is equal to either the lower or the upper bound is not consistent with the idea of a precautionary but realistic attitude to uncertainty. Habitat maps and models may inform plausible estimates of AOO based on guidance in section 4.10.7, especially if issues of detectability and prevalence are taken into account (Anderson 2023).

An important step in the approach outlined in section 4.10.7 is to estimate the proportion of potential habitat that is occupied at the time of the Red List assessment. This should be based on explicit assumptions referring to information on survey effort and success, and ecological factors such as predation, competition, disease, etc. that may limit occupancy within potential habitat. Assessors should describe this information and explain how it supports their estimate of the proportion of potential habitat that is occupied by the taxon.

Finally, where the plausible upper and lower bounds of AOO span the full range of categories from Least Concern to Critically Endangered, the species should be assigned to the Data Deficient category (section 3.2), unless other criteria apply.

4.10.9 Complementarity of AOO, EOO and number of locations

It should be understood that AOO, EOO and the number of locations are all spatial metrics that measure different (though sometimes overlapping) aspects of risk-spreading or insurance against spatially explicit threats. Therefore, all three measures should be estimated and assessed against the criteria where available data permit. As mentioned in section 4.9, to understand the relationships between these spatial metrics, it may be helpful to think of species that have similar values for one of these metrics and different values for the other. Suppose two species with similar life histories have the same EOO, but different values for AOO, perhaps because one has more specialized 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.

4.11 Location (criteria B and D)

"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 subpopulations. Where a taxon is affected by more than one threatening event, location should be defined by considering the most serious plausible threat." (IUCN 2001, 2012b)

In the Red List criteria, "location" refers to a threat-based area, and is different from the general notions of location and locality. The number of locations, AOO and EOO are metrics that measure different (though sometimes overlapping) aspects of risk-spreading or insurance against spatially explicit threats (see section 4.10.9). Fewer locations means that larger parts of a species' range are subject to the same threat, resulting in less risk-spreading, more correlated (synchronized) declines due to threats, and therefore greater extinction risk.

Justification for the number of locations used in Red List assessments should consider all areas whether they are under threat or not (see below), and, for areas that are under threat, should include reference to the most serious plausible threat(s). For example, where the most serious plausible threat is habitat loss due to development, a location is an area where a single development project can rapidly eliminate or severely reduce the population. The time frame should be short (e.g., within a single generation or three years, whichever is longer, but not any longer than is possible to project the threats and their impacts on the species).

When there are several threats, locations should be based on the one that has the maximum product of probability and consequence (in terms of percentage reduction in population).

Where the most serious plausible threat is habitat loss that occurs gradually and cumulatively via many small-scale events, such as clearance of small areas for small-holder grazing, a location can be defined by the area over which the population will be eliminated or severely reduced within a single generation or three years, whichever is longer. Where 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 the size of jurisdictions (within which similar regulations apply) or on the 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 subpopulations occur within an area that may be threatened by one such event, they must be counted as a single location. Conversely, if a single subpopulation covers an area larger than may be affected by any single event, it must be counted as more than one location.

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

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.

When parts of the distribution are not affected by any threat, the following options will be appropriate under different circumstances: (a) number of locations is not used (i.e., the subcriteria that refer to the number of locations consequently are not met), especially if the unaffected area is more than half the taxon's range; (b) number of locations in the unaffected areas is set to the number of subpopulations in those areas, especially if there are several subpopulations; (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 term "location" cannot be used and the subcriteria that refer to the number of locations will not be met.

4.12 Quantitative analysis (criterion E)

"A quantitative analysis is defined here as any form of analysis which estimates the extinction probability of a taxon based on known life history, habitat requirements, threats and any specified management options. Population viability analysis (PVA) is one such technique. Quantitative analyses should make full use of all relevant available data. In a situation in which there is limited information, such data as are available can be used to provide an estimate of extinction risk (for instance, estimating the impact of stochastic events on habitat). In presenting the results of quantitative analyses, the assumptions (which must be appropriate and defensible), the data used and the uncertainty in the data or quantitative model must be documented." (IUCN 2001, 2012b)

Quantitative analyses are used for assessing taxa under criterion E. Guidelines for applying criterion E are discussed in section 9. It is important to note that the risk-based thresholds of criterion E should not be used to infer an extinction risk for a taxon assessed as VU, EN and CR under any of the criteria A to D.

5. Guidelines for Applying Criterion A

The A criterion is designed to highlight taxa that have undergone a significant reduction in the near past, or are projected to experience a significant reduction in the near future. Methods of calculating reductions are explained in section 4.5.

The rationale for criterion A is that, all other things being equal, the probability of extinction is greater when the decline rate is high (Mace *et al*. 2008). The obvious mechanism is that if declines are not stopped, the population will go extinct, regardless of current population size. Even if a population is not currently declining, prior declines indicate risk of extinction. One reason is that if a population responded to a threat with a large decline, a similar decline can happen in the future in response to a similar threat. Further declines do not have to be immediate (criterion A does not require continuing decline). Another reason is that having declined to densities far below those at which it existed or evolved with, the species may be vulnerable to new threats or other changes in its environment, even if the population is not currently declining (see section 5.5 for examples).

Criterion A is based only on population reduction. The reason the IUCN criteria (except for E) consider symptoms of endangerment (such as decline, small population, restricted distribution, fragmentation, etc.) singly or a few in combination, instead of altogether, is that in the vast majority of cases reliable data on all of them do not exist for the same species. For example, although decline rates can be estimated based on an index of abundance (e.g., CPUE) and are relatively common, unbiased estimates of population size are rare, esp. for abundant species. Another reason criterion A considers only reduction is that, when a population is declining with a substantial rate, extinction risk is more sensitive to the rate of decline than to the population size (Lande *et al.* 2003). Finally, there are many examples of abundant species that have become extinct or nearly extinct. Such species could have been identified as threatened only by a criterion based only on declines (Stanton 2014). So, from both practical and theoretical points of view, it is necessary to have a criterion based only on rate of decline, in addition to one (criterion C) that is based on both population size and rate of decline.

Reductions under criterion A are considered over 10 years or three generations (whichever is longer, but up to a maximum of 100 years for future reductions). Scaling reductions with generation length is necessary because species with longer generation length recover more slowly from declines, although they may decline just as fast (the rate of population increase is limited by biological constraints whereas rates of human-induced declines are not). Therefore, the same *annual* rate of decline would put a longer-lived species at a higher risk of extinction. Scaling with generation length corrects this disparity.

Reductions for criterion A are calculated over 3 generations, because 1- or 2-generation reductions can be difficult to distinguish from fluctuations. Although the 3-generation requirement makes calculation of reduction challenging for long-lived species, it is essential for avoiding the underestimation of the extinction risk of these species. Ideally, reductions would be calculated from data that span 3 or more generations, but incomplete data or data from shorter time series can be used to calculate the 3-generation reduction (see section 4.5.1).

The criterion is split into the criteria A1, A2, A3 and A4.

- Criterion A1 deals with reductions in the past 10 years or three generations (whichever is longer) and is applicable to taxa in which the reduction is clearly reversible AND the causes of the reduction are understood AND have ceased (see discussion below), based on (and specifying) any of (a) to (e), as discussed above.
- Criterion A2 also deals with reductions in the past 10 years or three generations (whichever is longer) but for taxa where the reduction may not be reversible OR its causes may not have ceased OR may not be understood, based on (and specifying) any of (a) to (e) under A1.
- Criterion A3 deals with population reductions projected, inferred or suspected to be met in the future 10 years or three generations (whichever is longer, but up to a maximum of 100 years), based on (and specifying) any of (b) to (e) under A1.
- Criterion A4 deals with reductions observed, estimated, inferred, projected or suspected over any 10 year or three-generation time period (up to a maximum of 100 years into the future), where the time period must include both the past and the future, and where the causes of the reduction may not have ceased OR may not be understood OR the reduction may not be reversible, based on (and specifying) any of (a) to (e) under A1.

Under criterion A, a specific quantitative threshold indicating the population reduction must be met to qualify for one of the categories of threat. Under criterion A1, these thresholds are 90% (CR), 70% (EN) and 50% (VU). Under criteria A2, A3 and A4, these thresholds are 80% (CR), 50% (EN) and 30% (VU). These different rates reflect the understanding that taxa in which the causes of reduction are clearly reversible AND understood AND ceased are less at risk from extinction than those where the causes of reduction may not have ceased OR may not be understood OR may not be reversible. In order to use A1, three conditions must be met. (1) The reduction must be reversible. For example, the population size must not be so low that factors such as Allee effects make it impossible or unlikely to recover. It is the condition that must be reversible, not the cause of the deteriorated state. For example, loss of habitat may be irreversible even if the action that caused the loss has ceased. In contrast, a reduction in a forest-dependent species caused by logging could be considered reversible if changed management practices are leading to recovery of this species. (2) The causes of the reduction (the threatening factors) must be identified and their actions must be understood. Thus, it is not sufficient to simply list the threatening factors; it is also necessary to understand the scale and mechanism of their action (e.g., the magnitude and spatial distribution of overfishing, or the relationship between pollution and the population reduction). (3) The threatening factors must have ceased (e.g., overfishing has stopped). Examples of taxa that might qualify under criterion A1 are fish species that have suffered declines under exploitation but where the cause of reduction (e.g., over-exploitation) has ceased. This criterion may also be applicable to situations where the population is still being exploited, at lower levels of exploitation that do not cause additional population reductions. If any of the three conditions (reversible and understood and ceased) are not met in a substantial portion of the taxon's population (10% or more), then A2 should be used instead of A1.

5.1 The basis of reductions

Listing a taxon under criterion A requires specifying whether the reduction is based on (a) direct observation (A1, A2 and A4 only), (b) an index of abundance appropriate to the taxon, (c) a decline in area of occupancy, extent of occurrence and/or quality of habitat, (d) actual or potential levels of exploitation, and/or (e) the effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites.

The difference between direct observation (a) and index of abundance (b), as well as the value of distinguishing between them, lies in the assumptions to be met to provide valid estimates of population size. While “direct observation” requires only statistical assumptions (e.g., random sampling), indices of abundance require assumptions related to the biology of the species. For example, for a marine turtle species, use of “nesting females” to examine population change assumes that the proportion of mature individuals that breeds each year, and the number of visits to breeding sites per female per year are reasonably constant (or at least vary randomly) among years. If these assumptions are true, then “nesting females” is an appropriate index of mature individuals.

Direct observation (a) is the most relevant measure and, all things being equal, should be preferred. However, other measures may be used if they result in more reliable or more consistent (i.e., covering the three-generation period more comprehensively) estimates of population size through time; for example, for species that are difficult to detect, direct counts may entail large sampling errors and be biased (i.e., systematically under or overestimate the change in population size). Alternatively, an index based on easily detectable traces (e.g., tracks, droppings, etc.) or resources that the taxon depends on exclusively may provide more reliable estimates of population reduction. Similarly, for a species that is censused very infrequently, or responds to habitat loss with a time lag, habitat change may be a more comprehensive estimate of reduction than direct observation (see section 5.8 on the relationship between habitat change and population change).

All applicable bases for reduction should be listed. Even if the reduction is calculated based on the best available data, for example, from direct observation, if others (such as decline in area of occupancy) are also observed, estimated, inferred or suspected, these should also be specified.

The IUCN criteria use the terms "observed", "estimated", "projected", "inferred", and "suspected" to detail the nature of the evidence (including aspects of data quality) used for specific criteria. It is important to note that, for a given data source, not all combinations can form the basis for use of criterion A (Table 5.1). Examples below detail the types of data that might be used to calculate population reduction for criterion A.

Table 5.1. The relationship between the nature of evidence (data qualifiers) and the basis of reduction for criterion A.

	Basis of reduction for criterion A:				
	a	**b**	**c**	**d**	**e**
Nature of evidence (see section 3.1 for detailed information)	direct observation	index of abundance (e.g. CPUE)	AOO, EOO, habitat quality	actual or potential exploitation (e.g. landings, road kill)	introduced taxa, hybridization, pathogens, pollutants, competitors, parasites
observed (all counted - census)	A1, A2, A4	n.a.	n.a.	n.a.	n.a.
estimated (statistical assumptions)	A1, A2, A4	A1, A2, A4	n.a.	n.a.	n.a.
projected (extrapolated into future)	A4	A3, A4	n.a.	n.a.	n.a.
inferred (estimated from indirect evidence on variables of same type)	n.a.	A1, A2, A3, A4	n.a.	A1, A2, A3, A4	A1, A2, A3, A4
suspected (estimated from indirect evidence on variables of different type)	n.a.	n.a.	A1, A2, A3, A4	A1, A2, A3, A4	A1, A2, A3, A4

n.a. : not applicable

A population reduction can be *observed* if the data used to deduce the decline are from a census in which a direct count of all known individuals of a population is made. This can be used in criteria A1 or A2. For criterion A4, where the time frame for assessing reductions spans both the past and present, only the portion of a reduction in the past can be *observed*. The portion of the population trend in the future must be under another qualifier (e.g., *projected*).

A population reduction can be *estimated* from census data, as above, or from an index of abundance (e.g., Catch Per Unit Effort, density, number of nesting females; abundance based on mark-recapture data). Indices of abundance rely on statistical assumptions (e.g. about how the sampling scheme implemented relates to the number of mature individuals) and/or assumptions related to the biology of the species, i.e. how the index relates to the variable being estimated to calculate a population reduction (mature individuals).

A population reduction can be *projected* if it is extrapolated from census data or an index of abundance, either from the present into the future (criterion A3), or from past and present into the future (criterion A4). For example, a decline may be estimated for a population over two generations, and projected for a further generation into the future (criterion A4).

A population reduction can be *inferred* if it is calculated from indirect evidence of variables of the same general type. For example, population reduction in number of mature individuals calculated from a decline in catch data from fisheries, hunting data, or road kill (criterion A2d) could all be classed as *inferred*. Inference may also involve extrapolating an observed or estimated reduction from a known subpopulation to calculate an inferred reduction for another subpopulation of the same species. For example, an observed decline in population size from a forest fragment could be inferred to be the same for a subpopulation in a similar sized fragment that has not been censused, but which is perceived to be under the same threats. Inference may also be made from decline in EOO, or based on a reduction in habitat quality or extent. In this case we might expect the number of mature individuals of a habitat specialist species to have a closer association to the reduction in habitat extent than a non-habitat specialist.

A population reduction can be *suspected* if, based on circumstantial evidence, the relationship can be made based on a factor related to population abundance or distribution. The relevance of the factor as a proxy for number of mature individuals must be reasonably supported. Records of traditional ecological knowledge or anecdotal data may, for example, be used to calculate a suspected reduction over a given time period, if a population used to be seen regularly, but is now rarely observed.

5.2 The use of time caps in criterion A

Generation length is used in criterion A as a way of scaling the time frame over which reductions are measured with the life history of the taxon. Short-lived, faster-reproducing taxa have to suffer higher annual mortality rates than long-lived, slower-reproducing taxa to meet the same quantitative threshold (e.g., 80% reduction) over a set time period (e.g., 10 years). To put it another way, long-lived taxa might be unlikely ever to meet quantitative decline thresholds over a fixed time period, yet could be facing many years of population decline per recruitment opportunity. The three-generation time period is used to scale the decline rate threshold for the species' life history. This important scalar allows criterion A to be applied to a wide range of taxa. A minimum time cap of 10 years is specified because, although some taxa will have three-generation periods of less than 10 years, 10 years is the shortest time period of relevance to conservation planning and action. A maximum time cap has been introduced for assessments based on projections into the future, as it is felt that the distant future cannot be predicted with enough certainty to justify its use as a way of assessing whether a taxon is threatened. A maximum time cap is not applied to assessments based on past reductions, as it is felt that for long-lived taxa, it is important to use data for three generations, if it is available.

5.3 How to apply criterion A4

In order to decide whether a taxon can be listed under criterion A4, a "moving-window" reduction must be calculated. It is not possible to determine whether criterion A4 is applicable only by looking at the qualitative pattern of the decline, or by calculating only past or only future reductions.

To calculate a "moving window" reduction, first create a time series of past population sizes and future projections. Then, calculate 3-generation reduction for all time frames that include at least one past year and at least one future year. The length of all those time frames (windows) must be three generations or 10 years (whichever is longer), but cannot extend more than 100 years into the future. Finally, find the maximum of these reductions, which is the number to use in criterion A4. Whether a taxon is listed under criterion A4 or not, of course, depends on whether it qualifies under any of the other criteria.

In cases where reliable past data are available only for time periods of less than three generations, and/or reliable future predictions can only be made for less than three generations into the future, the 3-generation window to use in criterion A4 can be set as the time period for which reliable data and predictions are available.

In general, if a taxon is listed under criteria A2 and A3, it will also be listed under criterion A4. However, this is not always the case, and the category of threat determined using a "moving window" can exceed that calculated from past and future declines. Therefore, species should always be evaluated against criterion A4 as well as criteria A2 and A3. For a simple example of the use of criteria A2, A3 and A4, see the worksheet "A1-A4" in the spreadsheet **CriterionA_Workbook.xls** mentioned in section 4.5.

5.4 Reduction followed by short-term stabilization or increase: The 'ski-jump' effect

Some widespread, long-lived taxa show very large long-term declines as well as recent increases, and their population sizes are well above the thresholds for critical population size and distribution (under criteria B to D). This pattern has been termed the 'ski-jump' effect and affects any long-lived taxa that have declined in the past and are now stable or increasing. The question often asked is whether the long term historical declines or the more recent increases should take precedence in the assessment of threat in such taxa. However, the question is misleading; the IUCN criteria do not allow precedence among the criteria, or emphasizing one criterion over another. The correct interpretation is to assess the taxon against all the criteria. The point of criterion A is that long-term trends may indicate an underlying cause whereas recent trends may be temporary.

When applying criterion A to taxa showing these patterns, a few points should be remembered. (1) If the reduction is clearly reversible AND the causes of the reduction are understood AND have ceased, then the higher thresholds of criterion A1 (90% for CR, 70% for EN and 50% for VU) apply, which may lead to a down-listing of the taxon that would reflect the fact that it is currently stable or increasing. (2) Uncertainty in the data (particularly long-term historical data) if properly incorporated into the assessment may affect the outcome of the listing (see section 3.2). (3) If it is projected, inferred or suspected that populations will decline to the thresholds under criterion A, the taxon can be listed under criteria A3 or A4.

5.5 Historical reduction followed by long-term stabilization: Severely depleted populations

Some taxa (particularly marine taxa) show persistence at very low fractions of their unexploited equilibrium or carrying capacity. The current size of a population relative to historical levels can be calculated by estimating the reduction from the earliest year for which data are available to the current year (see section 4.5 for methods for estimating reductions). Such estimates, or other information, may show that a population is severely depleted relative to its unexploited equilibrium or carrying capacity. In some cases, taxa may be severely depleted, but show no detectable declines, so they may not qualify under criteria A1 or A2 because their declines occurred more than three generations ago, and they may be too widespread and abundant to qualify under any other criteria, reflecting the fact that they do not have a high extinction risk at present. Nevertheless, they may be more cause for concern because they are more susceptible to unforeseen catastrophic events and marine taxa may be harvested as bycatch in other fisheries. Such taxa are not currently being assessed as threatened under the criteria A1 and A2, although they may still qualify under criteria A3, A4, B, C, D or E.

Taxa in this situation may be assessed under criteria A3 or A4 based on projected or suspected population declines in the future, provided there is sufficient evidence for the threats faced by the taxon or the likely decline rate of the taxon to warrant such a listing. These range from biological or ecological factors (e.g., depensation or sex ratio effect thresholds especially in species adapted to high population density), to threat and detection factors (e.g. increased economic value increasing with rarity, technological innovation, or sudden removal of management measures). Such assessments against criteria A3 or A4 should be undertaken where the status of the species depends on conservation or management measures that are projected, suspected or inferred to become less effective over three generation lengths. Specific examples from marine taxa include: Queen Conch (*Strombus gigas*) and abalone (*Haliotis* spp.), which have minimum density requirements for reproduction (e.g., Hobday *et al.* 2001, Stoner *et al.* 2012); Gag (*Mycteroperca microlepis*), which may experience sperm limitation under heavy female sex ratio skew (Coleman *et al.* 1996); Nassau Grouper (*Epinephelus striatus*), which experienced a sudden collapse due to hyperstability or possible depensation (Sadovy and Domeier 2005); Totoaba Croaker (*Totoaba macdonaldi*), which underwent intense exploitation after a sudden increase in the value of the swim bladder (Sadovy and Cheung 2003); and Nassau Grouper in the Bahamas, which underwent a temporary removal of protection due to an economic downturn (Lam 2009).

The category Near Threatened could also be used if a taxon nearly qualifies as Vulnerable under criteria A3 or A4. It must be remembered however that the IUCN Red List Criteria are designed to identify taxa that exhibit symptoms of endangerment, and not simply depletion or conservation priority. The problem of assessing these taxa is also related to the scaling issues discussed under the definition of area of occupancy (section 4.10), which affects the application of criterion B. If an appropriate taxon-specific scaling factor is used, severely depleted marine taxa may qualify as threatened under criterion B.

5.6 Fisheries

5.6.1 Fisheries management and extinction risk

Taxa that are the targets of fisheries may show a decline in population size due to intentional management action. Under the Red List Criteria, such taxa could be assigned a threatened status under criterion A (declining population). Concern has been expressed that such a listing might not reflect extinction risk, especially if the decline is a consequence of a management plan designed to achieve a goal such as the maximisation of sustainable yield from a fishery.

It is important to note that criterion A measures declines over the last three generations, not from the original, unexploited stock. Thus, a well-managed stock should trigger the IUCN Criterion A thresholds only during the first three generations after the commencement of exploitation. Indeed, a species that is sustainably fished to achieve, for example, maximum sustainable yield (which could be at a biomass that is ~90% of the original biomass for a shark through to ~30% of the original biomass for a highly productive tuna) should have a current decline rate of zero. In addition, fisheries that are being managed sustainably would be assessed against the higher thresholds of criterion A1 (50% over three generations for VU), making it less likely that they will be classified as threatened.

There should not be a large number of fish stocks for which there would be a 50% reduction in population size over the most recent three generations due to commencement of regulated exploitation. This is because there are few stocks that were close to their unexploited state three generations ago. Rather, most major fisheries started more than three generations ago (Sethi *et al.* 2010). Even for these few stocks, a reduction of 50% should last only a few years (perhaps up to one generation) until the population approaches the target level and the decline rate decreases. If declines continued, there would be reason for concern; in this case a new assessment, against all five criteria, may indicate that the taxon is still threatened.

5.6.2 Technical aspects of using criterion A for fisheries

Percentage reductions in the number of mature individuals can be estimated in a number of ways, including ‘an index of abundance appropriate to the taxon’. In the case of exploited fishes, catch per unit effort (CPUE) may be used. This measure should be used with caution because changes in CPUE may underestimate population declines. This may occur, for example, if the population aggregates even at small sizes so that catches remain high with the same level of effort, even if the size of the population is declining. It may also occur if increases in fishing efficiency are not fully taken into account. It is therefore preferable to assess exploited fish taxa using the results of fishery-independent survey techniques.

Assessments of taxa under criterion A1 need to justify that the threat (e.g., overexploitation) has ceased and the taxon is being managed sustainably. This can be based on the ratio of the average level of fishing mortality (F) to the fishing mortality corresponding to maximum sustainable yield (MSY), i.e., $F/F_{\mathrm{MSY}} < 1$, for the greater of one generation or five years. Other methods could be used to justify the use of criterion A1 instead of A2. However, care needs to be taken to consider the chance that unsustainably managed species are incorrectly judged to be sustainable.

5.7 Long-lived taxa

The generation length of some species (e.g., some trees) can exceed 100 years. It is difficult to estimate population declines from a point in time before which the species populations or even the species itself may have been recorded. It is important to emphasize the point that the most significant declines, which are useful to record and which may be possible to reverse, are probably those that have been caused over the last 100 years.

5.8 Relationship between loss of habitat and population reduction

Under criterion A, a reduction in population size may be based on a decline in area of occupancy, extent of occurrence and/or quality of habitat. 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. For example, a bird species may not be reduced by 50% if 50% of its habitat is lost (perhaps because it will colonize 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, a coral reef fish may be reduced by more than 50% if 50% of its habitat is lost through fishing with explosives (perhaps because spawning areas have been destroyed).

The sensible use of inference and projection is encouraged when estimating population reductions from changes in habitat. For example, if a forest species' extent of occurrence has been 70% clear cut in the last five years it might be justified to suspect a 50% decline in the population over the past ten years. The species would therefore qualify as Endangered A2c**.**

In all cases, an understanding of the taxon and its relationship to its habitat, and the threats facing the habitat is central to making the most appropriate assumptions about habitat loss and subsequent population reduction. All assumptions about this relationship, and the information used should be included with the assessment documentation.

Available population data may 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. If it is decided that the abundance data are adequate to determine trends, the taxon should be listed under criterion A2. The implications of a possible lagged response in abundance to loss of habitat should, however, be considered when evaluating the taxon under criterion A3. For example, if population reduction in the last three generations is 30% based on abundance data, which are adequate to determine trends, then the species should be listed as VU A2, even if habitat loss in the same period was 60%. However, if a lagged response in abundance to loss of habitat is likely (i.e., the impact of habitat loss at present may lead to a future reduction in the number of mature individuals), then the population may be expected to decline further in the future (even if habitat loss has stopped), so an EN A3 or EN A4 listing should be considered as well, if the 60% loss of habitat is suspected to lead to 50% or more reduction in the number of mature individuals.

6. Guidelines for Applying Criterion B

Criterion B has been designed to identify populations with restricted distributions that are also severely fragmented or have few locations, undergoing a form of continuing decline, and/or exhibiting extreme fluctuations (in the present or near future). It is important to pay particular attention to criterion B, as it is the most commonly misused criterion. To qualify for criterion B, the general distributional threshold must first be met for one of the categories of threat, either in terms of extent of occurrence (EOO) or area of occupancy (AOO). The taxon must then meet at least TWO of the three options listed for criterion B. The options are (a) severely fragmented or known to exist in no more than x locations, (b) continuing decline, or (c) extreme fluctuation (Table 2.1). Therefore, if a taxon has met the distributional requirement for the Endangered category and option (c) extreme fluctuation, but none of the other options, it would not qualify as Endangered (or Vulnerable) under criterion B. To qualify, it would also have to meet either (a) or (b). An example of the proper use of criterion B is Endangered: B1ab(v). This means that the taxon is judged to have an extent of occurrence of less than 5,000 km^2, the population is severely fragmented or known to exist at no more than five locations, and there is a continuing decline in the number of mature individuals.

Subcriterion (a) requires severe fragmentation and/or limited number of locations. The numbering in the criteria does not allow distinguishing between these two conditions. We recommend that assessors make this distinction by explicitly specifying in their documentation: (1) whether the taxon is severely fragmented, and (2) the number of locations.

Some of the problems encountered when applying criterion B are dealt with elsewhere in this document, i.e. definitions of "subpopulations" (section 4.2), "location" (section 4.11), "continuing decline" (section 4.6), "extreme fluctuations" (section 4.7), "severely fragmented" (section 4.8), "extent of occurrence" (section 4.9) and "area of occupancy" (section 4.10). The different types of information used in criterion B need not be based on the same area at the same time of the year. For example, for a migratory species, AOO can be based on its distribution during the breeding season (because, e.g., the taxon occupies a smaller area during breeding, and AOO is the "smallest area essential at any stage to the survival of existing populations of a taxon"), and the locations can be based on the distribution and threats in the non-breeding season (because, e.g., they are "the most serious plausible threats" to the taxon).

7. Guidelines for Applying Criterion C

Criterion C has been designed to identify taxa with small populations that are currently declining or may decline in the near future. For criterion C, the small population threshold must be met as well as one of the two subcriteria that describe decline. For example, to qualify for Endangered under criterion C, the population must be estimated to number less than 2,500 mature individuals, and to either (1) have an estimated continuing decline in the number of mature individuals of at least 20% within five years or two generations (whichever is longer, up to a maximum of 100 years) or (2) have a continuing decline in the number of mature individuals and either (a) a restricted population structure or (b) extreme fluctuations in the number of mature individuals (see Table 2.1 for details).

Few taxa have data on both population size and decline rates at the necessary resolution to apply subcriterion C1. There is also some overlap between criteria A and C1, the difference being that criterion C applies only to small populations, the time frame over which the decline is measured is shorter (except for the Vulnerable category) and the decline rate thresholds are lower, because the populations are already small.

Criterion C2a has two subcriteria (i and ii), focusing on seemingly opposite conditions. These subcriteria take into account the fact that the distribution of a taxon's total population into either many subpopulations, or a single (or very few) subpopulation(s) could both lead to higher extinction risk, for different reasons. On the one hand, a taxon that is divided into many subpopulations may be severely fragmented (as defined in section 4.8), with many of the subpopulations having a small population size and a very high probability of extinction. On the other hand, a single subpopulation is like putting all eggs in one basket: a single subpopulation cannot recover from a local extinction by recolonization, or from a catastrophic decline by the rescue effect. Which of these is more important depends on subpopulation sizes and other factors. Criterion C2a covers both of these situations: (i) is for the first case, where even the largest subpopulation is quite small, and (ii) is for the second case, where almost all or all individuals are in the same subpopulation. A species that meets the general conditions for criterion C2a (i.e., has a small, declining population) is likely to be affected by one of these two conditions if they occur.

It may seem that a species with a single subpopulation (or with almost all individuals in the largest subpopulation) may not have increased risk of extinction, if it also has a wide range. However, this would be true only if the different parts of the range fluctuated and declined independently of each other. But if this were the case, then these different "parts" would likely not be connected (otherwise they would be in synchrony), so they should not be considered a single subpopulation. Thus, in order to apply criterion C2a correctly, it is important to identify subpopulations correctly (see section 4.2).

Some of the problems encountered when applying criterion C are dealt with elsewhere in this document, i.e. definitions of "subpopulations" (section 4.2), "mature individuals" (section 4.3), "continuing decline" (section 4.6), calculation of declines (section 4.5), and "extreme fluctuations" (section 4.7).

8. Guidelines for Applying Criterion D

This criterion identifies very small or restricted populations. A taxon qualifies for criterion D if the population of mature individuals (see section 4.3) is smaller than the threshold set for each of the categories of threat. Under the Vulnerable category there are two options, D1 and D2. A taxon qualifies for Vulnerable D1 if the population size is estimated to number fewer than 1,000 mature individuals (defined in section 4.3). Criterion D1 is provided for taxa that may not be declining, but are characterized by an acute restriction in their number of mature individuals, thereby rendering them particularly susceptible to stochastic events as well as to threats. A taxon qualifies for Vulnerable D2 if the area of occupancy is very restricted (typically less than 20 km^2) or exists at typically five or fewer locations, and if there is a plausible natural or anthropogenic threat. Criterion D2 is provided for taxa that may not be declining, but are characterized by an acute restriction in their area of occupancy or in their number of locations thereby rendering them particularly susceptible to a plausible threat.

The subcriterion D2 under Vulnerable was intended to be used for taxa with very small distributions. However, the thresholds for area of occupancy and the number of locations, although given as indicators (i.e., typically less than 20 km^2 or typically five or fewer locations), are frequently interpreted literally, which is not appropriate. Some people have argued that the subcriterion is too inclusive and results in massive over-listing, while others argue that it is too exclusive (e.g., many marine species) and so leads to under-listing. It must be emphasized that the restricted area of occupancy under criterion D2 is defined such that the population is prone to the effects of human activities or stochastic events in an uncertain future, and is thus capable of becoming Critically Endangered or even Extinct in a very short time period (e.g., within one or two generations—or within three to five years, if this is longer—after the threatening event occurs). The numerical thresholds are given more by way of example and are not intended to be interpreted as strict thresholds.

The focus of subcriterion D2 is not the area or the location count (for which many taxa could qualify), but the risk that the taxon could suddenly become Critically Endangered or Extinct (i.e., if the plausible threat is realized, then the species will within a very short time qualify for listing in one of these categories under, for example, criterion A or B). So, simply meeting the suggested (or any other) threshold for AOO or number of locations is not sufficient. It is necessary that this restriction makes the species capable of becoming CR or EX within a very short time, because of the effects of human activities or stochastic events. There must be a substantial possibility of these activities or events actually occurring. Thus, 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), events unlikely to cause extinction (e.g., because the species has survived many hurricanes, or is likely to adapt to global warming, etc.), or events unlikely to take place rapidly enough to result in a CR or EX listing in a very short time would not qualify for listing under criterion D2. The stochastic events or human activities that lead to this listing must be specified in the justification for listing (see example below). If the taxon is highly restricted, and there are plausible threats that can cause the species to become VU or EN in a short time, then the taxon should be considered for listing as NT.

8.1 Taxa known only from the type locality

If a taxon is only known from its type locality and there is no information on its current status or possible threats, the taxon should be listed as DD. If there are no plausible threats, and the area is relatively well known, Least Concern is appropriate, unless criteria A, B or C is met. If people have searched for the taxon, both at the type locality and at a reasonable number of other potential localities, and no more than 50 mature individuals are estimated, then the taxon would be listed as Critically Endangered D (the surveys must cover an appropriate time interval for the taxon). If any significant or plausible threats can be identified, then a full assessment will be necessary to determine the most appropriate classification (e.g., Critically Endangered under criteria B or C, or Vulnerable under criterion D2). If, despite searches, the taxon has not been recorded since the collection of the type specimen, and there are threats in the area, a listing of Critically Endangered (Possibly Extinct) or Extinct may be appropriate (see section 11 for guidance on how to make this determination).

8.2 Example of applying criterion D

A very rare bird species is described from two female specimens collected in 1851 and an observation in 1905 on a single island. The species was thought to be extinct in 1970, however, islanders reported that it may still exist, and in 1972 three birds were reported by an experienced bushman. It is thought that this unobtrusive and easily overlooked species may survive in two separate locations popular with trekkers and bird watchers. Very little is known about this species, but it is safe to estimate, given the limited sightings many years ago and the likelihood that bird watchers would have seen it, that the population contains less than 50 mature individuals. Therefore this species is listed as Critically Endangered: D.

8.3 Example of applying criterion D2

A bird species is confined to only four predator-free islands in close proximity, where it is common and its populations are considered stable. The historical range of this species was reduced as the result of the introduction of predators such as cats, rats *Rattus* spp. and a predatory bird. Birds attempting to colonize a neighbouring island are killed by cats and the predatory bird. The accidental introduction of alien species to the predator-free islands could easily cause local extinction. Thus, the number of locations is estimated as four (because it is unlikely that such introductions would occur on more than one island at any given time), and the species is classified as VU under criterion D2.

9. Guidelines for Applying Criterion E

To qualify under the E criterion a quantitative analysis such as a Population Viability Analysis (PVA) must be conducted to determine a species' probability of extinction over a given time period. For example, Critically Endangered E, would mean that the taxon has at least a 50%

probability of going extinct in the wild in the next 10 years or three generations (whichever is longer).

9.1 What is extinction?

Extinction is defined as population size reaching zero. Population size, for the purpose of defining extinction, is the number of all individuals of the taxon (not only mature individuals). In some cases, extinction can be defined as population size reaching a number larger than zero. For example, if only females are modelled, it is prudent to define extinction as one female (instead of zero) remaining in the population. More generally, an extinction threshold greater than zero is justified if factors that were not incorporated into the analysis due to a lack of information (for example, Allee effects, sex structure, genetics, or social interactions) make the predictions of the analysis at low population sizes unreliable.

For criterion E, extinction risk must be calculated for up to three time periods:

- 10 years or three generations, whichever is longer (up to a maximum of 100 years)
- 20 years or five generations, whichever is longer (up to a maximum of 100 years)
- 100 years

For a taxon with a generation length of 34 years or longer, only one assessment (for 100 years) is needed. For a taxon with a generation length of 20 to 33 years, two assessments (for three generations and 100 years) are needed. For a taxon with a generation length less than 20 years, all three assessments are needed.

9.2 Which method can be used?

One of the commonly used techniques of quantitative analysis is population viability analysis (PVA), which is a collection of methods for evaluating the threats faced by populations of species, their risks of extinction or decline, and their chances for recovery, based on species-specific data and models. For an introduction to PVA, see Boyce (1992), Burgman *et al*. (1993), Morris and Doak (2003). Types of models used in a PVA will be discussed below.

In some cases, criterion E can be used without a full PVA, using instead a quantitative analysis that does not necessarily include demographic information. For example, if a species is restricted to a small area, it may be possible to estimate the probability of the destruction of its entire remaining habitat. Such estimations may be based on past weather records, or other information about trends and locations of past habitat loss. It is important to remember, however, that such estimates can only be considered as lower bounds on the risk of extinction as it would have been estimated using a PVA. This is because a PVA incorporates such stochastic effects on habitat as well as other factors such as demographic variability, and other threats such as direct exploitation. Whatever the method used, the analysis must be numerical (i.e., a qualitative assessment such as "high probability of extinction" is not sufficient).

Which method is appropriate depends on the availability of data and the ecology of the taxon. The model structure should be detailed enough to use all the relevant data, but no more detailed. Assessments that use all the available and relevant data are more reliable than those that ignore part of the relevant information. However, including more detail than can be justified by the quality of the available data may result in increased uncertainty.

If the only available data are presence-absence information from a number of locations, occupancy models can be used (see Sjögren-Gulve and Hanski 2000, Mackenzie *et al.* 2017). If census information from a number of years is available, then a scalar (unstructured; count-based) dynamic model can be used (see Dennis *et al*. 1991, Burgman *et al*. 1993, Morris and Doak 2003). If data are available for various age classes or stages (e.g., juvenile and adult), then a structured model can be used (see Akçakaya 2000). If detailed data are available at the individual level (for example, pedigree data), then an individual-based (agent-based) model can be used (see Lacy 2000, DeAngelis and Diaz 2019). If data on the spatial distribution are available, a metapopulation model or other spatially explicit model should be considered (note that scalar, structured and individual-based models can all be spatially structured).

The second important consideration in selecting a model is the ecology of the species. The model structure and assumptions should be realistic with respect to the ecology of the species. The documentation should list all the assumptions (even the most obvious ones) related to model structure, parameters and uncertainties. In cases where the available data and the ecology of the species allow more than one type of model, comparative modelling (e.g., Brook *et al*. 2000, Kindvall 2000) and other types of validation (McCarthy *et al*. 2001) may strengthen the conclusions.

9.3 Are there sufficient data?

The types of data that can be used in an assessment include spatial distributions of suitable habitat, local populations or individuals, patterns of occupancy and extinction in habitat patches, presence-absence data, habitat relationships, abundance estimates from surveys and censuses, vital rate (fecundity and survival) estimates from censuses and mark-recapture studies, as well as temporal variation and spatial covariation in these parameters. Not all of these types of data are required for any one model. For more information about data needs of particular types of PVA models, see the references mentioned above.

When there is not sufficient data, or when the available information is too uncertain, it is risky to make a criterion E assessment with any method, including PVA. In order to decide whether the available data are sufficient to make a criterion E assessment, we suggest the following procedure. First, select a model structure based on the discussion in the previous section. Then, estimate the model parameters (see below), incorporating the uncertainties in the data. A simple way to do this is to make a best estimate for each parameter, as well as an “optimistic” and a “pessimistic” estimate. The more uncertain a parameter is, the wider the difference will be between the “optimistic” and the “pessimistic” estimates. Use these estimates to create a range of models, which should give a range of extinction risk estimates. The range of these estimates indicates whether the results are useful (and, hence, whether there is enough data). See also “Incorporating uncertainty” (section 9.5) below.

Remember that criterion E does not require very specific predictions. Even very uncertain results may be useful. For example, if the minimum estimate for the risk of extinction in 100 years is 10%, then the taxon is at least Vulnerable, regardless of the most pessimistic predictions. The criteria also allow incorporating uncertainty in the form of a range of categories presented in the documentation, while a single category should always be specified in the Red List (see Annex 1 of IUCN 2001, 2012b). So, for example, if the generation length is 10 years, and the extinction risk is 20–60% in 100 years, 10–30% in 50 years, and 5–10% in 30 years, the taxon could be classified as (VU–EN) in the documentation, while either has to be chosen for the Red List.

9.4 Model components and parameters

It is very important that model parameters are estimated without bias. However, it is difficult to provide detailed guidelines on parameter estimation because the components and parameters of a model depend on its structure. Thus, although we provide some general guidelines and specific examples in this section, these are not comprehensive.

9.4.1 Density dependence

Density dependence is the relationship between demographic parameters (such as survival, fecundity, population growth rate, etc.) and the size or density of the local population. The relationship can be negative (also called compensation), with demographic parameters decreasing as density increases, or it may be positive (also called depensation), with demographic parameters decreasing as density decreases. The former type of density dependence may result, for instance, from overcrowding and interspecific competition, and the latter may result from Allee effects, social structure, and inbreeding depression. Both types of density dependence have important effects on extinction risks, so models should address both. In other words, whether the model includes or excludes these types of density dependence, the choice should be justified.

Compensation is especially important to include in cases where habitat loss is a threat. Depensation can be incorporated by setting an extinction threshold greater than zero (see above).

Because density dependence affects demographic parameters such as survival and fecundity, estimates of these rates should include description of the population sizes or densities during the time period when the data for these estimates were obtained.

9.4.2 Temporal variability

Because the criteria are in terms of probabilities, it is essential that all relevant forms of variability are included in the assessment. Thus, the following types of variability should be considered: environmental fluctuations (in the form of random changes in one or more model parameters), demographic stochasticity, expected future trends in the average values of model parameters (e.g., as a result of deteriorating habitat), genetic stochasticity, random changes in the sex ratio, and low-probability, high-impact events (disturbances or catastrophes).

In modelling environmental fluctuations, the estimates of the variances of model parameters should include only temporal variation; variation due to demographic stochasticity, measurement error, spatial variation, etc. should be subtracted. For example, if survival rates are based on census data, binomial variance representing demographic stochasticity can be subtracted from total observed variance (Akçakaya 2002); if the survival rates are based on a mark-recapture analysis, methods described by Gould and Nichols (1998) and White *et al*. (2002), or in the help file of Program MARK can be used to remove demographic/sampling variance.

If catastrophes are included in the model, only data from non-catastrophe years should be used when estimating the mean and variance of the model variable (such as survival, fecundity, or carrying capacity) that the catastrophe affects.

When probabilistic results are based on simulations, the number of replications or iterations determines the precision of these results. In most cases, the randomly sampled model parameters are statistically representative if the number of replications is in the 1,000 to 10,000 range.

9.4.3 Spatial variability

If different subpopulations of the taxon are spatially separated or have different demographic rates, these should be incorporated by making the model spatially explicit. Modelling such a taxon with a single-population model may underestimate the extinction probability. When multiple populations are included in the model, the correlation among the different populations is an important factor; ignoring it (i.e., assuming all populations to be independent) may underestimate the extinction probability.

9.5 Incorporating uncertainty

We suggest that all parameters be specified as ranges (or as distributions) that reflect uncertainties in the data (lack of knowledge or measurement errors). In addition, uncertainties in the structure of the model can be incorporated by building multiple models (e.g., with different types of density dependence). There are various methods of propagating such uncertainties in calculations and simulations (Ferson *et al*. 1998). One of the simplest methods is to build best-case and worst-case models (e.g., Akçakaya and Raphael 1998). A best-case (or optimistic) model includes a combination of the lower bounds of parameters that have a negative effect on viability (such as variation in survival rate), and upper bounds of those that have a positive effect (such as average survival rate). A worst-case or pessimistic model includes the reverse bounds. The results from these two models can be used as upper and lower bounds on the estimate of extinction risk, which in turn can be used to specify a range of threat categories (see Annex 1 of IUCN 2001, 2012b).

9.6 Documentation requirements

Any Red List assessment that relies on criterion E should include a document that describes the quantitative methods used, as well as all the data files that were used in the analysis. The document and accompanying information should include enough detail to allow a reviewer to reconstruct the methods used and the results obtained.

The documentation should include a list of assumptions of the analysis, and provide explanations and justifications for these assumptions. All data used in estimation should be either referenced to a publication that is available in the public domain, or else be included with the listing documentation. The uncertainties in the data should be documented.

Methods used in estimating model parameters and in incorporating uncertainties should be described in detail. Time units used for different model parameters and components should be consistent; the periods over which parameters are estimated should be specified.

10. Guidelines for Applying the Categories DD, NT and NE

10.1 When to use the category Near Threatened

To qualify for the Near Threatened category, the taxon should be close to qualifying for the Vulnerable category. The estimates of population size or range size should be close to the Vulnerable thresholds, especially when there is a high degree of uncertainty, or possibly meet some of the subcriteria. This may be combined with biological susceptibility and threat.

The category Near Threatened is not specified by its own criteria, but instead by the proximity of a species to the criteria for the category Vulnerable. One way of determining whether the taxon is close to qualifying for Vulnerable is to follow the uncertainty guidance given in section 3. If the range of plausible categories include both LC and VU (or EN), the taxon can be classified as NT, unless the best estimate is VU (or EN). (If all categories from LC to CR are equally plausible, the taxon should be classified as DD.)

For taxa listed as Near Threatened on the IUCN Red List, assessors are asked to indicate as part of the justification, which criteria were nearly met. For example, NT listing would be justified in the following cases (in each case, any criteria not specifically mentioned are not met and are not nearly met):

- Based on data uncertainties, LC and VU are equally plausible categories.
- Based on data uncertainties, LC, VU, and EN are equally plausible categories (in this case, both NT and VU can be considered as the listing category).
- Population has declined by an estimated 20-25% in the last three generations.
- The taxon meets the area requirements under criterion B for threatened (EOO <20,000 km^2 and/or AOO <2,000 km^2) and is declining, but the population is not severely fragmented, occurs at many more than 10 locations, and there are no extreme fluctuations.
- The taxon meets the area requirements under criterion B for threatened (EOO <20,000 km^2 and/or AOO <2,000 km^2) and is severely fragmented, but the population is not declining, occurs at many more than 10 locations, and there are no extreme fluctuations.
- The taxon is declining and occurs at ten locations, but has an EOO of 30,000 km^2 and/or an AOO of 3,000 km^2, which are uncertain estimates.
- The taxon is declining and severely fragmented, but has an EOO of 30,000 km^2 and/or an AOO of 3,000 km^2, which are uncertain estimates.
- The taxon is declining and severely fragmented, but has an EOO of 22,000 km^2 and/or an AOO of 2,200 km^2, which are highly certain estimates.
- Population has declined by an estimated 10% in the last three generations, and is continuing to decline, and has about 15,000 mature individuals.
- The taxon exists in a single subpopulation of about 15,000 individuals and is declining.
- The population has about 1,500 mature individuals.
- The best estimate of population size is 2,000 mature individuals, but this estimate is very uncertain, and as low as 1,000 mature individuals cannot be ruled out.
- The taxon exists at three sites, occupying an area of 12 km^2; the population is being harvested but is not declining; there are no current threats, but there are plausible events that may cause the species to decline, but these are unlikely to make the species Extinct or Critically Endangered in a short time.
- Population has declined by 40% in the last three generations, but the decline has stopped, and the causes of the decline have been understood.

The following are examples of species that should not be listed as NT (or any of the categories of threat), unless other criteria apply:

- Based on data uncertainties, LC is the only plausible category.
- Population has declined by an estimated 10% in the last three generations, and there are more than 20,000 mature individuals.
- Population has declined by an estimated 30% as part of fluctuations.
- The taxon meets the area requirements under criterion B for CR (EOO <100 km^2 and/or AOO <10 km^2), but is not declining, not severely fragmented, there are no extreme fluctuations, and there are no obvious threats.
- The taxon is long-lived and slow growing, but does not meet any criteria A-E.

- The population has more than 2,000 mature individuals.
- The taxon exists at three sites, occupying an area of 30 km^2; the population is not declining; there are no current threats, and the species is very unlikely to become Extinct or Critically Endangered in a short time.

Previously (prior to 2021), dependence on conservation measures had been used to categorize taxa as NT that otherwise did not meet NT guidelines (see IUCN 2001; Annex 3). However, taxa in any Red List Category can be conservation-dependent. Conservation dependence (or conservation reliance) of taxa is more appropriately assessed as part of an IUCN Green Status of Species assessment (IUCN 2021), and should no longer be used to assess taxa as NT on the IUCN Red List.

10.2 Not Evaluated and Data Deficient

Listing in the categories of Not Evaluated (NE) and Data Deficient (DD) indicates that no estimation of extinction risk has been made, though for different reasons. NE indicates that no attempt to evaluate the current status of the taxon has been made. DD indicates that the taxon was evaluated using available data, which were found to be insufficient to place the taxon into a category. Taxa listed in these categories should not be treated as if they were not threatened.

10.3 When to use Data Deficient

If a taxon is known, but there is no direct or indirect information about its current status or possible threats, then it is obviously Data Deficient (DD). A Data Deficient listing does not imply that a taxon is not threatened.

The issue becomes more complex when there is very little information known about a taxon, but the available information indicates that the taxon may be threatened. The question then becomes how far is it acceptable to take inference and projection? This is discussed in greater detail in sections 3.1 and 3.2 (Data availability, inference and projection, and uncertainty).

When data are very uncertain, the category of Data Deficient may be assigned. However, in this case the assessor must provide documentation showing that this category has been assigned because data are inadequate to determine a threat category. If the data are so uncertain that both CR and LC are plausible categories, the taxon can be listed as DD. If plausible categories range from NT to threatened categories, DD is not the appropriate category; in this case, see section 3.2 about guidance to select the most plausible category while documenting the uncertainty. It is important to recognize that taxa that are poorly known can often be assigned a threat category on the basis of background information concerning the deterioration of their habitat and/or other causal factors; therefore, the liberal use of Data Deficient is discouraged.

Data Deficient species may be flagged with one or both of the following tags, although most DD species would not need either:

1. **Unknown provenance**. The taxon is known only from one or more specimens with no or extremely uncertain locality information, so that it is not possible to make any further inference about its status.

   Examples:

A hypothetical hummingbird known from a single trade-skin purchased in the 1900s in Bogotá, and speculated to have been collected on the East Andes or possibly the Central Andes of Colombia, within a few hundred kilometres of the capital. However, some "Bogotá" specimens came from as far away as Ecuador. Since no other specimen is known, it is assumed to be (or have been) a relict species of restricted range.

A hypothetical freshwater fish known only from the type collection made in 1930 one day west of Andapa which is somewhere along the northeast coast of Madagascar. This species has not been collected again since the 1930s, largely because its exact type locality is not known. There are therefore no data available upon which to base statements about the quality of its habitat or the size of its population, but it is assumed to be (or have been) a relict species of restricted range.

A hypothetical hemi-epiphytic aroid plant is endemic to Ecuador. It is known only from the type collection made in an unknown locale by a well-known botanist a century ago. The lack of information prevents any evaluation of the species' conservation status and this is further compounded by taxonomic problems with many species of the same genus described by the same botanist.

2. **Taxonomic uncertainty explains lack of information**. The paucity of data may be a consequence of taxonomic uncertainty, i.e. the lack of information on distribution, status, ecology and threats is because there are very few specimens and/or records, and this may be because the taxon represents aberrant individuals, hybrids, rare colour morphs, or subspecies of other species. This explanation is as or more likely than the possibility that the taxon is genuinely rare, threatened or has been inadequately searched for. It is important to note that this tag should not be used for taxa that simply have uncertainty around their taxonomy. Such taxa should not be classified as Data Deficient simply because of this uncertainty: they should either be regarded as good species and assessed against the Red List Criteria, or not assessed for the Red List. The process of determining the list of taxa to be assessed should be separated from the process of assessing extinction risk (see section 2.1 on taxonomy).

   Examples
   A hypothetical island bird species was named relatively recently on the basis of two specimens collected in the 1930's in a single location. The specimens are juveniles, and it was speculated that they may refer to juveniles of a related species, although differences in some morphological features make this unlikely. Nevertheless, the lack of any further information on distribution, population size, trends, ecology and threats, mean that the IUCN Red List Criteria cannot be applied, and the species is consequently classified as Data Deficient.
   A hypothetical bird species is known from one specimen collected in northeast Kalimantan in the early 1900's and another from Sumatra in the 1930's, plus reports in 1992 in Brunei. It has been speculated to be of hybrid origin, or a rare morph, although it is possible that it may be a genuinely rare habitat specialist that is occasionally forced to search other areas for food. With no further information, this uncertainty makes Data Deficient the most appropriate category.

   For further discussion and examples, see Butchart and Bird (2009).

Where a species name is widely accepted as containing multiple taxa that may deserve species-level recognition (a 'species complex') AND there is insufficient information (direct or indirect) to apply the Red List Categories and Criteria, the 'species complex' should be listed as Data Deficient. If the complexity and uncertainty of the taxonomic status plausibly explains the lack of information, then the assessment should be tagged as 'Taxonomic uncertainty explains lack of information'.

10.4 When not to use Data Deficient

Data Deficient classification implies that the taxon has been assessed against all criteria. All DD assessments require documentation of available data, sources of uncertainty and justification for why each of the five criteria cannot be applied (and, if applicable, the tags discussed in the previous section). If all of the five criteria have not been considered, DD cannot be used (the taxon must be categorized as NE).

In many cases, uncertainty in the data precludes placing the taxon in one of the other categories (LC to EX). However, not being able to place the taxon into a single category is, by itself, not a sufficient reason for a DD assessment. As discussed above, if the data are so uncertain that both CR and LC are plausible categories, the taxon can be listed as DD. If, however, plausible categories range from NT to threatened categories, DD is not the appropriate category. In this case, the assessor must select the most plausible category. If it is not possible to identify the most plausible category, the assessor must select one of the categories, based on their level of risk tolerance. For example, if LC, NT, and VU are considered to be equally plausible categories, the taxon may be categorised as NT. In all cases, the justification text must specify all categories that were considered plausible, as well as the degree of risk tolerance (see section 3.2.3). If assessors cannot decide on the level of risk tolerance, the mid category should be selected. It is important to note that, if uncertainty is specified at the parameter level (using the Red List Criteria Calculator in SIS), then the range of plausible categories and the most plausible category would be automatically selected, in accordance with the specified level of risk tolerance. See also section 3.2 about guidance to select the most plausible category while documenting the uncertainty; section 3.1 on data availability, inference and projection, and section 5.8 on inferring population reduction based on habitat loss.

In some cases, the data uncertainty has a spatial component; for example, there may be some data from one part of the range, but none or little from the other parts. In such cases, the assessors should try to avoid a DD listing by considering different plausible assumptions about how representative the threats are from known areas and use these assumptions to form uncertainty intervals for the parameters used (such as mature individuals, locations, subpopulations, etc.).

In other cases, the uncertainty may have a temporal component: the information may be more uncertain in the more distant past and/or about the more distant future. In such cases, the assessors should try to avoid a DD listing by using criterion A4 to minimize uncertainty. Considering a 3-generation window that includes both the more recent past and the more near future would focus the assessment to a period where data uncertainties are smaller.

11. Guidelines for Applying the Extinct Categories and Tag

11.1 The extinct categories (EX and EW)

The category of Extinct is used when ‘there is no reasonable doubt that the last individual has died’. However, extinction—the disappearance of the last individual of a species—is very difficult to detect. Listing of a species as Extinct requires that exhaustive surveys have been undertaken in all known or likely habitat throughout its historical range, at appropriate times (diurnal, seasonal, annual) and over a timeframe appropriate to its life cycle and life form. Thus, a key aspect of the definition of Extinct is "exhaustive surveys" (further guidance on this is in section 11.3).

Listing as Extinct has significant conservation implications, because protective measures and conservation funding are usually not targeted at species believed to be extinct. Therefore, a species should not be listed in the Extinct (EX) or Extinct in the Wild (EW) categories if there is any reasonable possibility that they may still be extant, in order to avoid the 'Romeo Error' (Collar 1998), where any protective measures and funding are removed from threatened species in the mistaken belief that they are already extinct. This term was first applied to the case of Cebu Flowerpecker *Dicaeum quadricolor*, which was rediscovered in 1992 after 86 years without a record (Dutson *et al*. 1993), having been written off as extinct at least 40 years earlier on the presumption that none of its forest habitat remained on the island of Cebu (Magsalay *et al*. 1995). An evidentiary approach to classifying extinctions is appropriate to encourage continuing conservation efforts until there is no reasonable doubt that the last individual of a species has died. However, if assessments of EX or EW are too evidentiary, then extinction rates based on the Red List are likely to be under-estimated. In other words, there are costs to making both types of errors (listing an extant species as EX and failing to list an extinct species as EX) and benefits of making the correct listing (Akçakaya *et al.* 2017). These issues are addressed by:

i) defining a 'Possibly Extinct' tag for species listed as CR but that are likely to be extinct (section 11.2);
ii) using methods that calculate the probability that the species is extinct, and comparing this probability to recommended thresholds (section 11.3); and
iii) using the probability that a species is extinct in calculating the number of extinct species and extinction rates (section 11.4).

It is strongly recommended that the methods and thresholds described in section 11.3 are applied to any species that has not been recorded since the last assessment, or is suspected to have become extinct.

Extinct in the Wild is defined as existing only in cultivation, in captivity or as a naturalized population (or populations) well outside the past range. "Cultivation" and "captivity" are not necessarily restricted to confinement. To be consistent with the definition of a "wild" subpopulation (see section 2.1.4 on managed subpopulations), EW should also be used if none of the subpopulations are wild. Thus, if the only surviving subpopulations of a taxon are not confined, but are nonetheless subject to intensive, individual-level management interventions as discussed in section 2.1.4, that taxon should be listed as EW. This category can also be applied when plant or fungal taxa are represented only by viable propagules (e.g., seeds or spores) in adequate storage facilities, if effective protocols have been developed for the taxon to ensure there is the potential for these propagules to develop into viable reproductive offspring and to undertake species recovery *in situ*.

11.2 'Possibly Extinct' tags for Critically Endangered taxa

Although an evidentiary approach to classifying extinctions is appropriate, this approach biases analyses of recent extinctions when based only on those species classified as Extinct or Extinct in the Wild (when individuals survive only in captivity). For example, the number of recent extinctions documented on the IUCN Red List is likely to be a significant underestimate, even for well-known taxa such as birds. The tag of 'Possibly Extinct' has therefore been developed to identify those Critically Endangered species that are, on the balance of evidence, likely to be extinct, but for which there is a small chance that they may be extant. 'Possibly Extinct in the Wild' correspondingly applies to such species known to survive in cultivation or captivity. Note that 'Possibly Extinct' and 'Possibly Extinct in the Wild' are tags, and **not** Red List Categories.

Relevant types of evidence supporting a listing as Extinct include (Butchart *et al.* 2006):

- for species with recent last records, the decline has been well documented.;
- severe threatening processes are known to have occurred (e.g., extensive habitat loss, the spread of alien invasive predators, intensive hunting, etc.);
- the species possesses attributes known to predispose taxa to extinction, e.g. flightlessness (for birds); or
- recent surveys have been apparently adequate and appropriate to the species' detectability, but have failed to detect the species.

Such evidence should be balanced against the following considerations (Butchart *et al.* 2006):

- recent field work has been inadequate (any surveys have been insufficiently intensive/extensive, or inappropriately timed; or the species' range is inaccessible, remote, unsafe or inadequately known);
- the species is difficult to detect (it is cryptic, inconspicuous, nocturnal, nomadic, silent or its vocalisations are unknown, identification is difficult, or the species occurs at low densities);
- there have been reasonably convincing recent local reports or unconfirmed sightings; and
- suitable habitat (free of introduced predators and pathogens if relevant) remains within the species' known range, and/or allospecies or congeners may survive despite similar threatening processes.

Similar considerations apply when assigning a taxon to either the Extinct in the Wild or Critically Endangered (Possibly Extinct in the Wild) categories. These considerations are implemented in the methods for calculating the probability that a species is extinct, and comparing this probability to recommended thresholds (as discussed in section 11.3). All assessments of taxa that might be extinct should follow the approach described in sections 11.3 and 11.4.

The documentation for each taxon assessed as Extinct, Extinct in the Wild, Critically Endangered (Possibly Extinct) and Critically Endangered (Possibly Extinct in the Wild) should explicitly justify the application of the Extinct categories and 'Possibly Extinct' tags. The documentation must summarize the lines of evidence for and against extinction, describe surveys carried out to search for the species and specify the date and relevant details of the last confirmed record. A completed data template (described below) can be used for this purpose. The status of all taxa assigned 'Possibly Extinct' tags should preferably be reviewed at five-year intervals.

There is sometimes difficulty in choosing the correct criteria for species listed as CR(PE) or CR(PEW). If the species disappeared from known sites within the last ten years or three generations (whichever is longer), then listing under criterion A2 is the preferable option. If the species is known from a single location with EOO less than 100 km^2 or AOO less than 10 km^2, then listing as CR B1ab(i,ii,v) or B2ab(i,ii,v) are possibilities. However, there are many species for which extinction is a possibility, but for which the declines or disappearances took place more than 10 years or three generations ago (whichever is longer), and for which the EOO and AOO are too large for listing as CR, and/or at least two subcriteria for CR B are not met. In such instances, the species should be listed as CR C2a(i), CR C2a(ii), and/or CR D, whichever seems more plausible. Such an assessment therefore implies an estimated population size of fewer than 250 mature individuals (for C2) or 50 mature individuals (for D). Even though it is impossible to know whether or not such an assumption is correct, it is a reasonable one for a species that could be Extinct.

11.3 Assigning taxa to EX or CR(PE)

Extinction of a taxon is often difficult to confirm, yet there are costs associated with the wrong listing (listing an extant taxon as EX, or failing to list an extinct taxon as EX) as well as benefits to making the correct listing. This section describes an approach to making these listings as consistently as possible, by quantifying how "exhaustive" surveys for the taxon have been, which is a key aspect of the definition of Extinct. The approach involves two models (the Threats Model and the Records and Surveys Model) that calculate the probability that a taxon is extinct, $P(E)$, and comparing this probability to thresholds that were determined based on a cost-benefit framework (Akçakaya *et al.* 2017). The following sections describe these two models, their parameters, and recommendations for interpreting their results. In these sections, everything mentioned about EX also applies to EW, and everything mentioned about CR(PE) also applies to CR(PEW).

To use the models described here, download the data template **EX_data.xlsx**, the instructions document **EX_instructions.pdf**, and the R script **RecordsSurveysModel.R**, which are available at https://www.iucnredlist.org/resources/ex-probability.

Assessors can use other approaches to estimate $P(E)$, as long as the alternative approaches incorporate the factors and parameters about threats, records, and surveys that are discussed below, and therefore quantify the extent to which the surveys had been exhaustive.

11.3.1 The Threats Model

The Threats Model (Keith *et al.* 2017) estimates the probability that the taxon is extinct, $P(E)$, based on qualitative and, where available, quantitative information about the severity, duration and scope of threats and their interaction with the life history traits that determine the species' susceptibility to these threats.

To use this model, estimate two subjective probabilities, based on expert knowledge of the threats faced by the species:

1. *P(local)*, the probability that the combination of threats affecting the species occurred for a sufficient duration and was sufficiently severe that they caused local extinction;
2. *P(spatial)*, the probability that the threats occurred over the entire range of the species.

Estimating ***P(local)*** requires assessors to draw on the history of the impacts of threats on populations of the target taxon. A relevant historical observation, for example, would be that the taxon disappeared from an area shortly after the introduction of an invasive alien predator. It may also draw on examples where the threats have caused ecologically similar or phylogenetically related taxa to become extinct at a particular location. Inferences about which taxa are ‘ecologically similar or related’ may be based on life history (e.g., life cycle structure, dependence on hosts, body size, diet), habitat ecology (e.g., microhabitat type, breeding sites) and/or phylogeny.

Estimating ***P(spatial)*** requires assessors to evaluate two components: (i) the likelihood that the threats (with sufficient severity and duration to have caused local extinction) operated throughout the entire range of the taxon (i.e., distribution of habitat and/or individuals, as appropriate); and (ii) the certainty with which the range limits are known. Relevant considerations for the first component include whether the threats operated in such a pattern as to have caused extinction throughout the taxon's range. This may be influenced by the spatial occurrence of different threats, dispersal dynamics, migration patterns and patch dynamics, as well as species life-history traits and cultural factors that influence species susceptibility to threats (see Keith *et al.* 2017 for further discussion). Relevant factors to consider for the second component (range limits) include taxonomic uncertainty, reliability of records and whether potential habitat outside the confirmed range has been adequately searched. These uncertainties are incorporated into estimates of *P(spatial)* by setting upper and lower bounds taking into account plausible maximum and minimum extents of the taxon's range.

For both *P(local)* and *P(spatial)*, estimate a plausible lower bound (minimum), an upper bound (maximum) and a mid-value (best estimate). See the instructions document for a general guide in estimating these probabilities.

11.3.2 The Records and Surveys Model

The Records and Surveys Model (Thompson *et al.* 2017) is an iterative model to estimate the probability that the taxon is extinct, $P(E)$, based on a time series of records of the taxon, and the timing, comprehensiveness and adequacy of any targeted surveys designed to detect the taxon after the last known record. A record is any evidence that the taxon was extant in a given year. Surveys are dedicated or passive (opportunistic) but unsuccessful efforts to find the taxon (i.e., surveys do not result in a record). For each year, enter a maximum of 1 record or 1 survey.

For each record, estimate p(ci), the probability that the taxon is correctly identified as extant. This probability depends on the type and quality of evidence, similarity of the individual recorded to taxa with which it could potentially be confused, circumstances of the record and the skill and experience of the recorder. Before estimating p(ci), it may be helpful to create a default table of probabilities for each of the common types of records available for the taxa you are assessing, to act as a guide and to ensure consistency, rather than being prescriptive (see the instructions document for an example).

For this and any other quantity described in this section, estimate a plausible lower bound (minimum), a plausible upper bound (maximum) and a mid-value (best estimate).

For each survey, estimate the following three quantities:

(1) ε (epsilon), the proportion of the taxon's habitat within its likely entire range that was surveyed (or covered by passive surveillance). If there were several dedicated surveys within a year in different areas of the range, make only one entry, with the total proportion of the taxon's habitat surveyed across all the surveys. Even when the range of a species is very uncertain, it may be possible to estimate ε with sufficient certainty to allow estimation of $P(E)$. If, for instance, the northern and southern range limits of a species' range are unknown, but the survey intensity is similar regardless of latitude, it would be possible to estimate ε with a higher certainty than the range itself. The full range of potential plausible habitat should be considered when estimating ε. For example, outlier records (e.g., at greater depths, altitude, or drier climates) may give insights into the potential occurrence of the taxon in a broader range of environments than indicated by the majority of previous records, particularly if some of these environments are hitherto poorly explored. The reliability and precision of outlier records should also be considered when estimating ε.

(2) p(r), the probability that the taxon, or recent evidence of it, would have been recorded in the area that was surveyed, if it were present. This depends on aspects of detectability, including body size, behaviour (e.g., activity and movement patterns, shyness, tendency to skulk, phenology, vocalization, sociality), degree of crypsis, local abundance, and accessibility to or searchability of its habitat and microhabitat.

(3) p(i), the probability that the taxon, or recent evidence of it, could have been reliably identified in the survey if it had been recorded. This depends on the verifiability of the record; that is, the likelihood that the recorded taxon could be distinguished from a similar taxon (e.g., a congener) given its distinctiveness (e.g., in appearance, morphology, vocalizations, behaviour), and the identification skill of the observers. Assessors must consider all signs of recent evidence (e.g., scat, spoor, nests, owl pellets, woodpecker bark peelings, shells, etc.) and all life-stages at the time of the survey; for example, the mature life-form may be highly distinctive, but the juvenile/seed/ larval/dormant life-stages may be extremely difficult to distinguish from similar taxa.

For p(r) and p(i), it may be helpful to create a default table of probabilities for taxa with different characteristics in the taxonomic group you are assessing (see examples in the instruction document).

11.3.3 Interpreting the model results

After completing the data entry, follow the instructions in the files mentioned above (the data file, the instructions document, and the R script). The results of both models will be displayed in a graph such as Figure 11.1, which includes $P(E)$ estimated by the two models (the square marker), the bounds of the estimates (the error bars), and lines indicating the thresholds of $P(E)$ for considering a species CR(PE) or EX (the red lines). The following thresholds are recommended:

CR(PE), if $P(E) \geq 0.5$ and <0.9
EX, if $P(E) \geq 0.9$

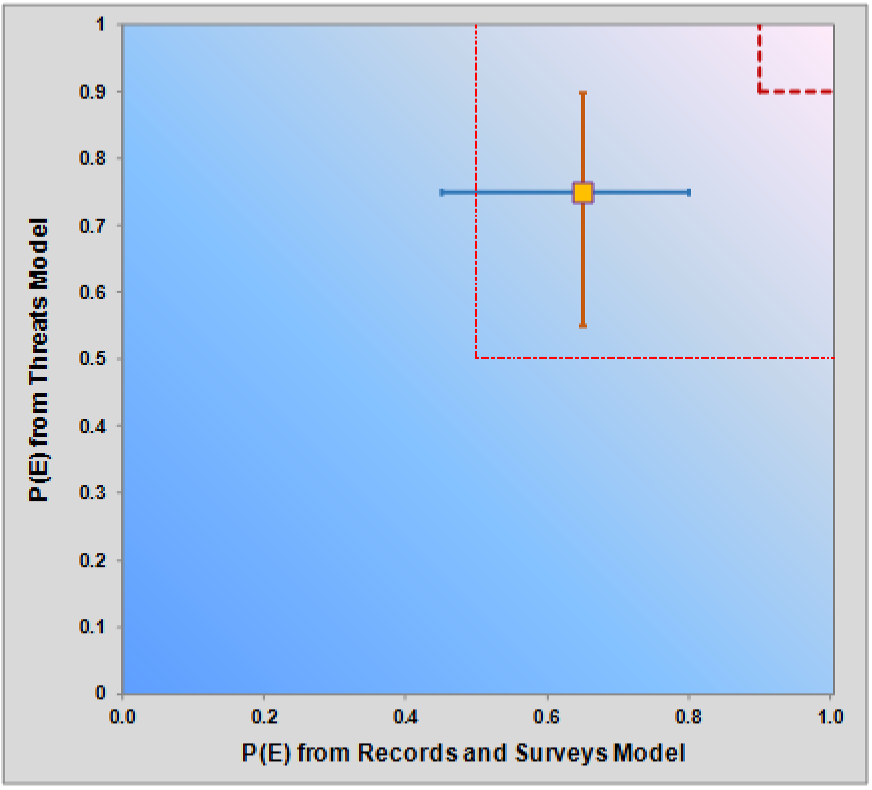


Figure 11.1. Graphical display of *P*(*E*), the probability that the species is extinct, based on the two models. The square marker shows the best estimates and the error bars show the uncertainty bounds, based on the Threats Model (y axis) and the Records and Surveys Model (x-axis). The thick red lines indicate the thresholds of *P*(*E*) for listing a species as EX, and the thin red lines indicate the thresholds for listing as CR(PE).

The recommended thresholds are based on considerations of the costs of making the wrong call and benefits of making the right one, and on consideration that the costs are not the same for the different types of errors (e.g., listing an extant species as EX versus not listing an extinct species as EX). Akçakaya *et al.* (2017) presents a detailed discussion of these considerations. In addition, these thresholds were tested for birds (Butchart *et al.* 2018) and for a small number of amphibians and invertebrates.

Nevertheless, the recommended thresholds should be considered as guidelines, rather than rules. For example, Butchart *et al.* (2018) give examples of species for which they judge the probability of extinction to have been under-estimated by these methods. One reason this could happen is that the records may not be independent as assumed by the Records and Surveys Model. A possible mechanism of dependence among records might be that, if a record is known publicly, it may bias the judgment of less-experienced observers, or increase their likelihood of claiming a record.

When interpreting the results for possible listing of the species as CR(PE) or EX, results of both models, as well as the uncertainties of the results should be considered. For instance, if both methods give *P*(*E*) estimates with lower bounds above 0.9, then there is strong indication that the species should be listed as EX. Conversely, if both methods give *P*(*E*) estimates with upper bounds below 0.5, then there is strong indication that the species should be considered extant.

When the two methods give substantially different results, but have similar amounts of uncertainty, we recommend that the decision is based on the method that gives the lower value for *P*(*E*). In other words, listing as EX, for instance, requires *both* methods to give $P(E)>0.9$. This corresponds to "Method 1" in Butchart *et al.* (2018; see Figure 1).

If the two models have substantially different amounts of uncertainty, the assessors may consider giving more weight to the model with narrower uncertainty bounds. Such consideration can be guided by calculating a weighted average of the two $P(E)$ estimates, where the weights are the reciprocal or complement of the uncertainty range (i.e., 1/*range* or 1–*range*, where *range* is $P(E)_{max}–P(E)_{min}$; see the 'Results' worksheet).

The spreadsheet of input estimates and the output extinction probabilities calculated using these methods should be documented and referenced (if published) or submitted (as Supplementary Information) as part of Red List assessments for the relevant taxa.

11.4 Calculating the number of extinct species and extinction rates

Analyses that calculate the number of extinct species (globally, in a region, or in a taxonomic group) or extinction rates (proportion of species that have gone extinct) should consider estimates of $P(E)$, the probability that a species is extinct. If $P(E)$ can be estimated for all species, the number of extinct species should be estimated as the sum of these probabilities— rather than simply summing the numbers of species listed as EX or CR(PE)—so that the estimated number of extinct species is independent of the thresholds of $P(E)$ for EX and CR(PE). See Akçakaya *et al.* (2017, Table 3) for a demonstration of this calculation.

If $P(E)$ is not calculated for some species that are listed as EX or CR(PE), then the above calculation should be made by assigning a weight to each such species, and summing those weights. The weights should be the average $P(E)$ for species in the same taxonomic group with the same listing, for which $P(E)$ has been calculated. If there are no (or very few) species in the group for which $P(E)$ has been calculated, then the weights should be 0.95 and 0.70, for species listed as EX and CR(PE), respectively. These weights are based on the midpoint of the range of $P(E)$ for each category based on the recommended thresholds (see above).

12. Guidelines for Threatening Processes

As discussed in an earlier section (2.3), the criteria aim to detect symptoms of endangerment rather than causes (see also Mace *et al*. 2008). Consequently, they are applicable to any threatening process that results in symptoms such as population decline, small population sizes, and small geographic distributions. A taxon may be classified as threatened even if a threatening process cannot be identified. Regardless of the nature of threats, assessments must follow IUCN (2001, 2012b) and these guidelines to ensure valid application of the criteria. However, different threats, especially new or poorly understood processes such as global climate change may require further guidance in the application of definitions and criteria.

The purpose of this section is to provide such specific guidance. In this version, we focus on global climate change; future versions will provide further guidance on how the criteria may be interpreted to assess taxa affected by other threats. It is important to note that the guidance in this section is not an alternative to previous sections.

One aspect of a Red List assessment involves listing the major threats in the required documentation, as described in IUCN (2001, 2012b; Annex 3), using a standard classification scheme available at www.iucnredlist.org/technical-documents/classification-schemes. A separate set of guidelines are currently being developed for evidence-based identification and quantification of threats. The guidance given in this section does not relate to this process of identifying threats; instead, the focus here is on the application of the Red List Categories and Criteria.

12.1 Global climate change

There has been concern that the Red List Criteria may not be adequate for assessing species threatened with climate change. This is because many species that are projected to undergo substantial range contractions in the future have short generation lengths. Consequently, there are concerns that the assessment time frames are too short for the inferred population declines to trigger the relevant IUCN Red List Criteria, which consider declines over a three-generation period (see section 12.1.1). However, recent studies show that the IUCN Red List Criteria can identify species vulnerable to extinction due to climate change. In a study involving North American reptiles and amphibians, Pearson *et al.* (2014) showed that extinction risk due to climate change can be predicted by information available now, such as current occupied area and population size, much of which is used in the IUCN Red List Criteria.

Stanton *et al.* (2015) defined "warning time" as the time between when a species is first identified as threatened and when it goes extinct, assuming no conservation action. Using the same species and climate projections as Pearson *et al.* (2014), they showed that IUCN Red List Criteria can identify species that would go extinct because of climate change without conservation action, and can do so with decades of warning time. In an independent study, Keith *et al.* (2014) reached the same conclusion for a short-lived Australian amphibian. Although these studies show the ability of the IUCN Red List Criteria to identify species vulnerable to extinction because of climate change, they also show that warning times may be short in data-poor situations, and if conservation action is started only when a species is listed at the highest IUCN threat category (Critically Endangered). Therefore, there is a need to develop further guidance for using the IUCN Red List system, especially in data-poor situations and for timely policy responses to exploit the maximum warning time available for species on extinction trajectories in response to changes in climate. As new research increases understanding of the impacts of climate change on species, the results will be used to improve these guidelines. Below, guidance is provided on a number of relevant issues, based on research available in 2015.

12.1.1 Time horizons

An important issue in the application of the criteria to species impacted by climate change concerns the time horizons over which the assessments are made.

The time horizons used in the criteria serve several purposes. First, the generation time is used as a surrogate for turnover rates within populations and as a biologically relevant scaling factor that corrects for the variation in rates at which different taxa survive and reproduce. Second, the time horizon is set to a minimum of 10 years because measuring changes over shorter time periods is difficult and does not reflect time scales for human interventions. Third, the time horizon is set to a maximum of 100 years into the future, because of the uncertainties in predicting population sizes for a long time from the present day (Mace *et al*. 2008).

The global climate is projected to continue to change for several centuries (IPCC 2013; Chapter 12). The effects on biological systems will certainly continue for a long time. Thus, for many species, especially short-lived ones, Red List assessments are based on time horizons much shorter than the long periods over which we now expect the world's climate and its effect on species will change. This by itself may not make climate change fundamentally different: other threats, such as habitat loss may also continue for a long time.

However, the nature of change in biological systems caused by climate change is thought to be different than changes caused by other threats. Thuiller *et al*. (2005), for example, argued that, "the recognized time scales for assigning species IUCN Red List Categories are not suited to evaluating the consequences of slow-acting but persistent threats," suggesting that the projected climate change impacts are thought to be of a more deterministic nature than other threats. In addition, some amount of climate change-related impact is irreversible (already committed) because of the lag between greenhouse gas emissions and climate change (and subsequent biological change).

While stochastic events (catastrophic fires, ENSO events, etc.) that contribute to the variability and hence the risk of extinction of populations clearly operate at different time scales than climate change, there are other processes that also are slow-acting and persistent. For example, it is debatable whether threats such as habitat loss and fragmentation linked to land use change are any less persistent or any more uncertain than climate change. Although climate change may be persistent, the predictions are also very uncertain. For example, IPCC (2013) makes most of its predictions only until 2100 because general climate models tend to produce very different outputs towards the end of the 21st century.

The criteria recognize that some threats may be irreversible (as explicitly noted in criterion A). For example, in many cases, habitat loss brought about by urban sprawl is not reversible. Various threats may involve time lags similar to that of climate change. For example, human populations have a momentum, and thus there is often a lag between a change in the human population growth rate and resulting changes in human pressures on natural systems.

Thus, the assessment of species with short generation times is not fundamentally different under climate change and under other threats. Although short-lived species may not be listed under criterion A initially, if they are affected by climate change they will be listed (likely under criteria B or C) as their ranges and populations change as a response to climate change. They can also be listed under criterion E (see below).

In summary, many of the issues related to time horizons are not specific to global climate change. Although future versions of this document may provide further guidance on this issue, for the time being, the horizons for each of the criteria should continue to be applied as they are currently specified, regardless of the nature of the threatening factor, including global climate change.

12.1.2 Suggested steps for applying the criteria under climate change

There are a number of challenges in applying the criteria to species impacted by global climate change, which have resulted in several misapplications of the criteria. A common mistake is making arbitrary changes to thresholds or time horizons specified in the IUCN Red List Criteria (see Akçakaya *et al.* 2006 for examples and details). An important characteristic of the Red List is that threat categories are comparable across taxonomic groups. For this important standard to be maintained, it is essential that the thresholds and time periods used in the criteria are not altered (see section 12.1.1).

To assess species that might be impacted by climate change, the following steps are recommended (Figure 12.1), as available data and information about the species permit.

1. Assessors are encouraged to think systematically through the potential mechanisms of the impact of climate change on the species (see section 12.1.3 below). The identification of likely mechanisms of impact will help with defining key variables used in Red List assessments in the context of climate change. This diagnostic process may be aided by development of diagrammatic models.

2. Assessors should identify and estimate or infer the values of all the parameters in the Red List criteria relevant to the mechanisms of taxon change under climate change identified in Step 1. These parameters include "very restricted distribution" and "plausibility and immediacy of threat" (section 12.1.4), "number of locations" (12.1.5), "severely fragmented populations" (12.1.6), "extreme fluctuations" (12.1.7), "continuing decline" (12.1.8), and "population reductions" (12.1.8). Inferences about such variables can lead to listing under criteria A, B, D2 or C2 (Figure 12.1).

3. To incorporate future climate impacts on species more explicitly, assessors are encouraged to make inferences about the magnitude of future population reduction (criteria A3 and A4) and whether continuing decline (criteria B and C2) will occur due to climate change (see section 12.1.8). Such inferences can be aided by developing models of (a) bioclimatic habitat or (b) population dynamics (see sections 12.1.9, 12.1.10, and 12.1.12). The identification of likely mechanisms of impact will also help with developing such models. The output of such models can lead to listings under criteria A, C1 or E (Figure 12.1).

4. Finally, the results of the bioclimatic models can be used to determine the spatial structure of stochastic population models, which are then used to estimate probability of extinction for assessment under criterion E (discussed in detail in section 12.1.11). This allows assessors to explicitly incorporate effects of future habitat shifts and habitat fragmentation, future increases in climate variability (hence in extreme fluctuations), and dispersal limitations and barriers. The output of such models can lead to listings under criteria A or E (Figure 12.1). However, this approach requires substantial amounts of demographic information that may not be available for most species.

Assessors should first complete Steps 1 and 2, and then complete as many of the remaining steps as the available data and expertise allow. In the following sections, we discuss mechanisms of impact of climate change, applications of various definitions and criteria, and use of different types of models for estimating population reductions and continuing declines. Although we discuss particular criteria in this section, this does not mean that these are the only applicable ones. As with any other threat, the taxon should be assessed against all the criteria as available data permit.

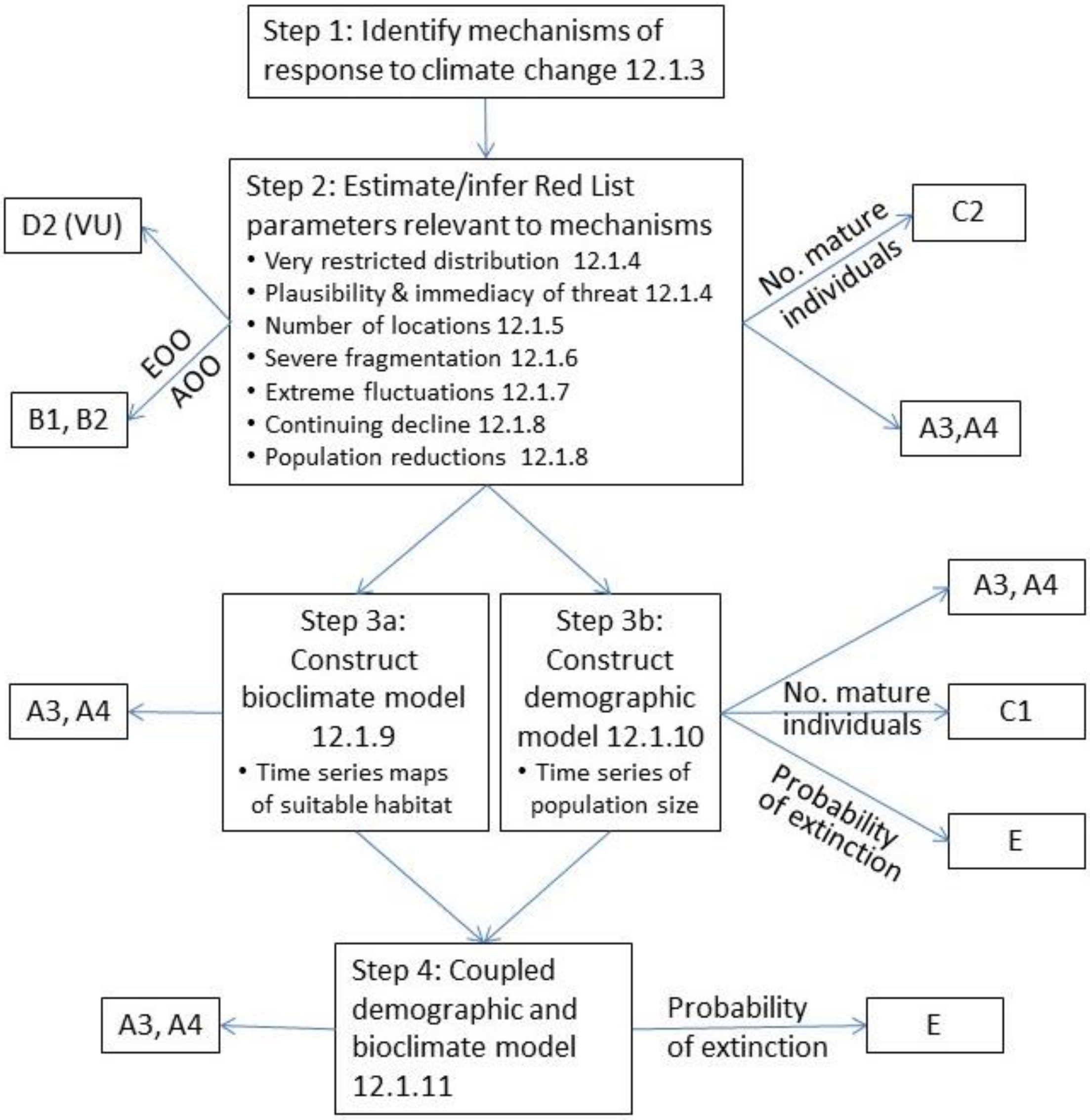


Figure 12.1. Protocol for assessing extinction risks under climate change using the IUCN Red List Criteria for threatened species (IUCN 2001). Letters and numbers in marginal boxes refer to respective Red List Criteria. Numbers within central boxes refer to relevant sections of text in these Guidelines. Any assessment must address all plausible threats (not just climate change) and should also evaluate eligibility for listing under criteria A1, A2 and D1 (not shown).

12.1.3 Mechanisms

Climate change can affect populations via many mechanisms; thinking about how this will occur for a given taxon can clarify the parameters and criteria relevant for a Red List assessment. Relevant parameters for assessments under climate change include “very restricted distribution”, “plausibility & immediacy of threat”, “number of locations”, “severe fragmentation”, “continuing decline”, “extreme fluctuations”, and “population reductions”. The relevant criteria for future effects of climate change include A3, A4, B1, B2, C1, C2, D2 (VU), and E (Figure 12.1).

The effects of climate change on taxa are analysed quantitatively through two main groups of symptoms: changes in the taxon’s distribution and changes in the demography of the taxon which is then included in population models. While range changes have been the most studied symptom of species decline due to climate change (Pearson *et al.* 2002), changes in demography can also lead to reductions in population abundance even when species distributions are projected to increase under climate change. This is because births, deaths, emigration and immigration drive population dynamics and these are demographic factors not necessarily directly linked to habitat and range size (Thuiller *et al.* 2014). Demographic factors that could be affected by climate change include vital rates (e.g., survival, growth, fecundity, and dispersal), species interactions, phenology, population responses to disturbance, and deposition and production of calcareous structures and tissues (e.g., in corals) (Foden *et al.* 2013). Hence, when considering population declines driven by climate change, it is important to consider the main mechanisms by which this is likely to occur as this will highlight the most appropriate criteria for assessment under this threat.

Changes in habitat can occur under climate change because climate is a predictor of habitat suitability for many taxa. Changes in precipitation and temperature across space can shift, fragment, contract or increase species ranges, leading to changes in EOO and AOO and the degree of fragmentation. The ability of a population to track shifts or increases in suitable habitat will depend on its dispersal capabilities (Foden *et al.* 2013). However, changes in climatic variables can also expose organisms to conditions outside their range of physical tolerance, resulting in reduced survival and fecundity, leading to reductions in population size (Deutsch *et al.* 2008). In the case of corals, increased ocean temperatures or changes in pH can reduce or prevent development of calcareous tissues, thus reducing survival and growth rates. Increased temperatures can change predator-prey relationships, or food webs, by altering organisms' behaviour such as movement and exposure times, with potential ramifications to both the predator and prey or consumer and resource (Gilman *et al.* 2010). Phenology, or the timing of life-cycle processes, can shift by climate change such that a mismatch occurs between, say, the timing of flowering and the presence of pollinators (Memmott *et al.* 2007). And changes in the intensity and frequency of environmental events, such as fire, drought, or floods can reduce populations that have evolved under a different regime (Dale *et al.* 2001). For example, obligate seeding plants that rely on seedbanks for post-fire seed regeneration will undergo population declines if fire frequency is increased, because fewer seeds will be added to seedbanks between successive fires.

12.1.4 Very restricted distribution and plausibility and immediacy of threat (VU D2)

Taxa that have very restricted distributions and become susceptible under climate change to a threat that is plausible and liable to cause the entire population to rapidly become Critically Endangered or even Extinct in the Wild will be eligible for listing as Vulnerable under criterion D2. However, classification under criterion D2 is only permissible if the effects of climate change are such that the taxon is capable of becoming Critically Endangered or Extinct in a very short time period after the effects of the threat become apparent.

Application of this criterion requires only knowledge of the species' distribution and an understanding of the severity and immediacy of impacts of a plausible threat. For example, a sessile terrestrial organism that is susceptible to salt would qualify for listing as VU D2 if it had a very restricted distribution in a coastal location that is projected to become more exposed to salt water or saltspray as a consequence of projected rises in sea-level and/or increased frequency of coastal storms. More detailed examples are given below.

Example 1. A species that currently does not meet the area thresholds under criterion B may be classified as VU D2 if bioclimate models (see section 12.1.12) predict that a range reduction could correspond to a population reduction of 80% or more (and other information indicates that there are few locations; see above). In this case, the start of the decline may not occur soon, but the decline is plausible, and once it begins it is expected to cause a population reduction in a very short period of time (e.g., within one to two generations or 10 years) so that the species will be classified as CR A3c, so it now meets VU D2.

Example 2. A species of coral currently has a restricted area of occupancy (less than 20 km^2) but does not satisfy the criteria for classification under criterion B. Climate change models predict increases in ocean temperatures, greater than the typical seasonal variation, across the entire range of the species. This temperature increase is expected to cause coral bleaching such that the area of occupancy will be reduced to less than 10 km^2 within 10 years of the start of bleaching. It is highly uncertain when the temperature increase or the onset of the bleaching will occur, but there is a reasonable chance that it will occur in the future. Once the bleaching starts, the species will meet CR B2ab within a short time, so it now meets VU D2.

Example 3. A small mammal with an AOO >500 km^2 occurs in a single location (see example of Species 3 in section 12.1.5) where it is dependent on snow cover (for insulation and predator avoidance during the winter). Climate change is expected to increase the probability of a series of years with no or inadequate snow cover. If this occurs, the species is expected to decline by 80% or more within 1–2 generations due to mortality from exposure and predation. Although having a number of years with no snow cover is a stochastic process and cannot be exactly predicted, in this case the climate models indicate that it is a plausible event. The species meets VU D2 because this plausible event, once it occurs, will cause the species to be listed as CR.

Example 4. A species has AOO <20 km^2 but is not declining or under any specific threat or experiencing extreme fluctuations. It is expected that future climate change will affect this species, but the effects are expected to cause gradual and slow decline, which will not trigger any criteria for CR or cause extinction within three generations. Thus, this species does not meet VU D2.

Example 5. A fish species known only from a single oceanic archipelago, where it occurs from 1 to 30 m depth. It lives in small recesses on slopes and walls of rocky reefs. In this region, localized declines, including the complete loss of at least one other endemic fish species, have occurred after strong El Niño-Southern Oscillation (ENSO) events that result in shallow waters that are too warm and nutrient poor for extended periods of time. The frequency and duration of ENSO events in this region appears to be increasing. Given the restricted distribution of the species and its specialized shallow water habitat, oceanographic environmental changes, such as those associated with future ENSO events, may cause the extinction of this species in a short period of time (as has happened for a similar species). Thus, it meets VU D2.

12.1.5 Definition of "Location" under climate change (B1, B2, D2)

Using the number of locations in Red List assessments requires the most serious plausible threat(s) to be identified (see section 12.1.3). In some cases, the most serious plausible threat will be climate change, but it may not be correct to assume that species threatened by climate change occupy a single location. In general, it is not possible to identify climate change as the main threat (for purposes of defining locations) without knowing something about how the effects of climate change are likely to be played out through the proximate causes or direct threats. For most species susceptible to climate change, climate change itself (e.g., increasing temperatures or changes in precipitation) is not the direct threat. Rather, the process through which climate change is expected to affect species involves a large variety of threats or proximate causes—such as changes in fire frequency, hydrology, species interactions, habitat suitability, diseases—that affect the species vital rates (these proximate causes can be inferred using knowledge of species ecology and predicted changes in relevant climatic variables). Thus, even when the ultimate cause of endangerment is climate change, the locations occupied by a species should be defined (and counted) in terms of these direct threats. Climate change should only be used to define the number of locations when it is the direct threat (e.g., where survival rates are reduced by thermal stress and are likely to be the principal direct cause of population declines or when suitable habitat is reduced due to changes in temperature and precipitation).

In some cases, climate change may threaten different parts of a species' range through different proximate factors, or not affect some parts at all (for example, part of the range may be expanding). In such cases, the most serious plausible threats should be used to define locations in different parts of the species range in accordance with section 4.11 (options a–d).

Examples of estimating the number of locations for species susceptible to climate change:

Species 1 is restricted to a single climatic zone affected by severe storms that cause episodes of high mortality. The frequency of severe storms in the region is projected to increase by at least 20% over the next 100 years. A single severe storm is unlikely to affect the entire range of the species, but two severe storms could cover the entire range. The species is correctly estimated to occur at two locations based on severe storms as the proximate threat (the minimum number of independent storms that could affect its entire range). It would be incorrect to interpret the species as occupying a single location based on the single climatic zone occupied in which severe storm frequency is projected to increase.

Species 2 is restricted to three coastal freshwater wetlands potentially affected by saltwater incursion associated with sea level rise. Two of the wetlands occur on the same floodplain, one at a low-lying site 0.5 metres above sea level, and another perched on the upper floodplain five metres above sea level. The third wetland also occurs at five metres above sea level, but in another region where there is a very large inter-tidal range. Sea level is projected to rise, on average by 1.0 metre by year 2100. The low-lying wetland will certainly be affected by sea level rise. The nearby perched wetland is very unlikely to be affected by sea level rise. The third wetland could be affected by saltwater incursion during extreme spring tides under projected future climate, but this is uncertain. Incursion by saltwater is the most serious plausible threat at the low-lying (first) site and the distant (third) site with the high inter-tidal range. These two sites could be interpreted as a single location if they are both threatened by the same regional sea-level rise. However, if sea level rise could lead to different outcomes at the two sites they could be interpreted as two separate locations. For example, the same amount of sea level rise may inundate the first wetland but only sporadically affect the third wetland, causing different types of impacts at the two wetlands (total habitat loss in one and temporary population reduction in the other). If the independence of threat outcomes at the two wetlands is uncertain, then a bounded estimate of [1-2] locations is appropriate (see section 3). The second wetland is very unlikely to be affected by sea level rise, and hence the most serious plausible threat for this wetland is not sea level rise. If this site is subject to other threats, the most serious plausible one will govern how many locations are represented at that site. For example, if the entire wetland is threatened by polluted runoff, then it should be counted as a single location and the total number of locations for the species is [2-3]. Alternatively, if the second wetland is not threatened, then the number of subpopulations at that site could be used as a proxy or the number of locations may not be applicable to the assessment of the species (i.e., the subcriteria for number of locations cannot be met, see section 4.11).

Species 3 is restricted to the highest altitudes of two mountain ranges separated by a plain of 100 km. The two mountain ranges have a seasonal cover of winter snow that extends above a similar threshold altitude (1,800 m above sea level), although the summits of their mountains are at different elevations. Seasonal snow cover affects breeding success by providing insulation during cold winters. The extent of snow cover is projected to decline stochastically over the next 30 years. The most serious plausible threat is the risk of a year in which there will be very low or no winter snow cover, which causes an episode of very high mortality in the species population. The chance of this occurring in the same year on both mountain ranges is about 30%, based on correlation of minimum snow extent over previous years. Despite their geographic separation, the two mountain ranges are interpreted as a single location for the species because they may be affected by the same 'low-snow cover' event.

12.1.6 Severe fragmentation (B1, B2, and C2)

If a taxon is not currently severely fragmented (see section 4.8), this cannot be used to meet the severe fragmentation subcriteria (e.g., criterion B1a) even if there is evidence to infer that it may become so under future climates. However, projected future fragmentation can be used to infer continuing decline, if certain conditions are met. Continuing decline is recent, current or projected future decline (see section 4.6). Severe fragmentation can for some species lead to local extinctions of subpopulations inhabiting the smallest habitat fragments. If the population density and the projected distribution of fragments justify a prediction of increasing rate of local extinctions in the near future, this may be used to infer continuing future decline in population size.

The same conditions may also allow inferring population reduction under criterion A3, but this requires a quantitative prediction. Suppose that a bioclimatic model (see section 12.1.12) predicts that EOO of a taxon will decline by 20% in the next three generations due to climate change. Assuming that the population reduction will be at least as large as the EOO reduction (but see section 12.1.8), this can be used to infer a 20% population reduction but would not by itself meet the VU threshold for A3. However, suppose that a population dynamic model predicts that populations smaller than a certain size have 50% risk of extinction. If the bioclimatic model also predicts that 40% of the population will be in fragments that support populations of this size or smaller, then we can infer that the population will undergo a further 20% reduction due to increased local extinction of smaller populations. Combined with the 20% reduction due to range contraction, this result can be used to infer a total of 40% population reduction, listing the species as VU A3.

12.1.7 Extreme fluctuations (B1, B2, and C2)

One of the predictions of many climate models is an increase in the frequency of extreme weather events (such as droughts, heat waves, etc.). This may increase population fluctuations to extreme levels (see section 4.7). If a taxon is not currently experiencing extreme fluctuations but is predicted to do so in the future as a result of climate change, this prediction cannot be used to meet the extreme fluctuation subcriteria (e.g., B1c). However, a projected future increase in population fluctuations can be used to infer continuing decline if certain conditions are met. Continuing decline is recent, current or projected future decline (see section 4.6). Extreme fluctuations can for some species lead to an increase in rate of local extinctions of subpopulations (especially if combined with severe fragmentation; see above). If the population sizes and the projected increase in fluctuations justify a prediction of increasing rate of local extinctions in the near future, this may be used to infer continuing future decline in population size.

A prediction of future extreme fluctuations can also contribute to a VU D2 listing if projected local extinctions could cause it to meet the criteria for CR in a very short period of time (see above).

12.1.8 Inferring population reduction and continuing decline (A3, A4, B1, B2, C2)

Criteria A3 and A4 may be applied if a population reduction of a given magnitude may be inferred from relevant evidence. Unless there are quantitative models enabling projections of suitable habitat or population size under future climates, the evidence base will be indirect or circumstantial (section 3.1). For example, if there is evidence of a strong relationship between temperature and survival or temperature and breeding success, and there are projections of future temperatures that suggest that they will rise rapidly enough to reduce the number of mature individuals by at least 30% within the next 10 years or three generations, whichever is longer, then this information may be used to apply criterion A3. Similar inferences may be used to infer the direction of trends in the number of mature individuals, which may be used to infer continuing declines under criteria B1, B2 and C2.

12.1.9 Inferring reductions from bioclimatic models (A3, A4)

Bioclimate models are often used to predict changes in a taxon’s range as defined by climatic variables. Such models are also known as species distribution models (SDM) or ecological niche models (ENM) that use climatic variables (among others) as predictor variables (see section 12.1.12 for detailed guidance on developing these models). Note that bioclimate models also use predictor variables other than climatic ones (e.g., land-cover variables). The results of bioclimate models will be a series of habitat suitability maps. In order to infer population reduction (for use in criteria A3 or A4) from these maps, it is necessary to calculate the expected population size (or a relative population index) from the current map and from the map for the time step that corresponds to three generations in the future. If climate data are not available for the year that corresponds to three generations in the future, it should be created by interpolation from the available layers.

Even if the current population size of the taxon is known, the same method of estimation should be used for both the "current" and the "future" maps. This is because the quantity of interest is the proportional change in population size, and using the same methods removes some of the effects of the assumptions involved in making this conversion from habitat suitability (HS) to population size.

The relationship between population reduction and habitat loss may not be linear (see section 5.8). However, in the absence of more specific information, it is an allowable assumption. With this assumption, the conversion from habitat suitability (HS) to population size will involve summing all the HS values in each map and calculating the proportional change in three generations. One important correction to this calculation is to use a threshold value of HS, to exclude from calculation of proportional reduction any areas that are unlikely to support a population because of low suitability. For other important corrections and recommended methods, see the detailed guidance on developing bioclimate models in section 12.1.12.

Assessments with criterion A3 require projections for the next 3 generations or 10 years, whichever is longer (up to a maximum of 100 years). Shorter projections with models can be combined with past reduction, for assessments under Criterion A4. For instance, one-generation past reduction can be combined with two-generation reduction based on model outputs. This may result in a narrower range of uncertainty than a three-generation future reduction based on model outputs, especially for long-lived species with reliable data on recent population trends.

In addition to bioclimate models, correlative analyses of population size or density as a function of environmental factors can also be used to infer population reductions. For example, the 2015 assessment of the Polar Bear (*Ursus maritimus*) used statistical relationships between sea ice and population size, combined with projected future decrease in sea ice, to calculate the range of plausible future 3-generation population reduction amounts (Wiig *et al.* 2015).

Projected change in habitat can also be used to infer continuing decline in EOO, AOO, or habitat quality (e.g., criterion B1b(i,ii,iii)).

12.1.10 Inferring reductions from demographic change

As noted in section 12.1.3, climate change may lead to population reductions or continuing declines through a range of demographic mechanisms. Understanding these can help to project the direction and rate of population response. The tools that are used to inform these projections will depend on the mechanism of response. In this section we briefly review the principal mechanisms, alert assessors to appropriate means of inference and suggest suitable tools to inform projection.

Some mechanisms are based on a direct ecophysiological relationship between a climate variable and one or more vital rates of the population. For example, in some taxa quantified relationships exist between fecundity and particular temperature variables for which projections can be derived from the outputs of Global Climate Models (GCMs, e.g., Kearney and Porter 2009). Other vital rates including survival, growth and dispersal may be affected. A range of plausible scenarios can be constructed from uncertainty in both the species response and the climate projection to estimate plausible bounds of population reduction. This method of projection will usually involve some assumptions about rates of adaptation to new environmental conditions (Hoffmann and Sgrò 2011). In some cases, there may be sufficient data to use demographic models for this purpose.

Some mechanisms involve a relationship between calcification rates and ocean acidity for organisms with calcified body parts (e.g., corals, molluscs) (Orr *et al.* 2005). Hence projections of ocean acidification (with characterisation of uncertainty in trends) should permit inferences about the continuing declines (criteria B and C) and projections of population reduction over required time frames (criterion A). This should be based on defensible assumptions about rates of adaptation and should generate bounded estimates to represent the uncertainty in the projections.

A wide range of taxa have life history processes and vital rates that respond to regimes of fire, flood or storms, and hence may undergo population reductions depending on how disturbance regimes respond to climate change. It is possible to generate projections for indices of change in the frequency, intensity and season of such disturbance events from GCMs (e.g., Milly *et al.* 2002, Clarke *et al.* 2012, Zhao *et al.* 2015). Such projections, in combination with models of the species responses to the disturbance should support inferences about continuing declines and bounded estimates of population reduction over required time frames. Changes in the frequency of heat waves and other extreme weather events could be treated in a similar manner where they are key drivers of declines.

A fourth mechanism of response to climate change involves changes to species interactions. These are challenging to predict, but it may be plausible to project the direction of change, as a basis for inferring continuing declines, if the mechanisms are reasonably well understood. Examples include population changes of a target species inferred from projected increases in the area of spatial overlap between the habitat of the target taxon with those of its competitors, predators or disease vectors. Another example involves continuing declines inferred from phenological decoupling of mutualistic or facilitation interactions, or conversely phenological changes that result in increased exposure to competitors, predators or diseases.

Quantitative estimates of population reduction may be derived for many of these estimates using stochastic population models (e.g., Akçakaya *et al.* 2004). The parameterisation of these models may be adjusted to reflect projected trends in vital rates under a range of future climate scenarios based on regionally skilled GCMs (see section 12.1.12 for guidance on the selection of these). All applications of such models should justify the parameter settings and selection of scenarios used in projection. Recent developments allow the coupling of stochastic demographic models to species distribution models projected to produce a time series of habitat suitability maps under future climate scenarios (e.g., Keith *et al.* 2008). Alternative modelling approaches are developing to achieve similar goals (e.g., Cabral *et al.* 2013). These not only allow projections of future population reductions for assessment of criteria A3 and A4 but may produce estimates of extinction risk over required time frames for assessment under criterion E (see section 12.1.11).

12.1.11 Estimating extinction risk quantitatively with coupled habitat and population models (E)

Because of its time horizon for VU of 100 years (regardless of generation time), criterion E can be used to list species with short generation times that are predicted to be threatened by climate change. However, the difficulties with using criterion E (see section 9) are increased when climate change is the main threat, because of the need to take into account multiple types of stochastic and deterministic changes in the taxon's environment, demography and habitat that are caused or exacerbated by climate change.

New approaches that link outputs of GCMs to species habitat models and metapopulation models can be used to estimate risks of extinction (Keith *et al*. 2008, Anderson *et al*. 2009, Brook *et al*. 2009, Cabral *et al.* 2013) when adequate data are available for developing both bioclimate models (see section 12.1.12) and population models (see section 9). Preliminary findings from these studies showed that extinction risks under climate change are subject to complex dependencies between species life history, distribution patterns and landscape processes (Keith *et al*. 2008). Even in cases where adequate data exist to parameterize coupled SDM-population type models, addressing multiple sources of uncertainties associated with demographic model parameters, choice of GCMs and emission scenarios through the use of sensitivity analyses can aid the interpretation of complex model outcomes (Naujokaitis-Lewis *et al*. 2013, Prowse *et al*. 2016).

It is very important not to ignore other threats, which may interact with, or supersede, climate change impacts when predicting species vulnerability to climate change. Approaches that focus on climate change alone may therefore lead to underestimation of extinction risks (Brook *et al*. 2009).

12.1.12 Using bioclimate models

Some of the guidance in the preceding sections refers to variables that may be calculated from outputs of bioclimate models. Such models are also known as species distribution models (SDM) or ecological niche models (ENM) that use climatic variables as predictor variables. This section will summarize methodological guidance in the use of these models for the purposes of Red List assessments. It is important to note that the use of these models is not necessary for all assessments of species threatened with climate change. Future versions of this document may include guidelines for other types of predictive modelling (such as eco-physiological models) that may be useful for Red List assessments.

Bioclimate models have been widely applied to explore potential impacts of climate change on species distributions (for reviews of this field see: Guisan and Zimmerman 2000, Guisan and Thuiller 2005, Heikkinen *et al*. 2006, Franklin 2010, Peterson *et al*. 2011; for a practical introduction see Pearson 2007). These models commonly utilize associations between environmental variables and known species' occurrence records to identify climatic conditions within which populations can be maintained. The spatial distribution that is suitable for the species in the future can then be estimated under future climate scenarios. Advantages and disadvantages of this modelling approach have been widely debated in the literature, and multiple uncertainties make it essential that the model outputs are carefully interpreted (Pearson and Dawson 2003, Hampe 2004, Araújo and Guisan 2006, Thuiller *et al*. 2008).

Bioclimate models may provide useful information for Red Listing by identifying species that are more or less likely to experience contractions in the area of suitable climate space in the future and by estimating the degree to which potential distributions in the future might overlap with current observed distributions. The guidelines here are intended as a list of methodological issues that must be carefully considered in applications of these models for red listing under climate change. It is important that methodologies are well justified within the context of any particular study, and with respect to the biology of the taxon being assessed. Assessments that rely on bioclimate models will be reviewed by the Standards and Petitions Committee (SPC), so sufficient detail must be provided to allow the SPC to determine if the model follows these guidelines. To facilitate this determination, Red List assessments that use bioclimate models must include a supplementary information document with detailed information about the model, with the headings and subheadings as given in the **Bioclimate Model Template** file (download this file from the IUCN Red List website: https://www.iucnredlist.org/resources/supporting-information-guidelines).

The guidance provided in this section, and the detailed information and justification that is required, may suggest a higher burden placed on assessments that use models. Although a high burden is not the intention, the guidance does reflect the need to ensure that assessments made with complex methods that involve many parameters and assumptions are defensible. The guidance also intends to balance this need with the need to ensure that assessments do not exclude future declines or extinctions because of the technical difficulty of using these models. Assessors who are not sure whether the intended modelling exercise meets the minimum standards are encouraged to seek clarification from the SPC through the Red List Unit.

Results of bioclimatic models can be used in various ways to help with species assessments under the Red List Categories and Criteria. These uses include inferring population reduction under criterion A3 and continuing decline (see section 12.1.9), linking bioclimate and demographic models for criteria E (section 12.1.11), inferring continuing decline from projected increases in fragmentation (see section 12.1.6), and projecting plausible threats for use in criterion D2 (see section 12.1.4). Although the interpretation of the results from these models for Red List assessments relies on several assumptions, they do allow a tentative solution to the problem of incorporating the long-term impacts of climate change. Several alternative modelling approaches are being developed to explore the relationship between climate change and species endangerment (see section 12.1.11), which will allow more comprehensive guidelines for assessing the risk of extinction due to climate change.

12.1.12.1 Species data (the response variable)

In most cases, the response variable is a combination of (i) species presence or occurrence points and (ii) background or (pseudo-) absence points (discussed below). If a different response variable (e.g., population density, or "occurrence" points sampled from polygons representing the known range of the species) is planned to be used, the details of the methods should be shared with, and approved by, the Red List Unit prior to the analyses. These guidelines focus on models where the response variable is presence-background, presence-pseudoabsence, or presence-absence. Methods that use only presence data (and do not use background points, pseudo-absence points, or absence points) are not acceptable (see below).

Quality of species occurrence data

Bioclimate models rely on observed occurrence records for characterizing species limits of tolerance to climate predictors, so it is essential that these data are of good quality. Confidence in the accuracy of georeferencing and species identifications of occurrence records should be high. It is important that georeferencing of occurrence records is accurate to a degree that is relevant to the resolution of the environmental variables (e.g., accuracy should be within a few tens of metres if the resolution of analysis is 1 km^2). Similarly, the time of the occurrence records should be within the time range represented by the predictor variables (or each occurrence record should be associated with the value of the predictor variables at the time of that record). Ideally, occurrence records should be associated with vouchered specimens and/or should have been identified by experts in the taxonomic group of interest. Data extracted from distributed databases (e.g., GBIF, HerpNET) should be carefully checked for accuracy, coverage, and sampling intensity prior to use. Questionable records should be removed, and the checks performed on the data documented. In many cases, occurrence records are spatially biased (i.e., not randomly distributed across the species' range). This may be because they are opportunistic records, or even when gathered using systematic methods (e.g. records of human-wildlife conflicts such as collisions with vehicles) they are the result of non-homogenous sampling efforts. In such cases, thinning (or resampling, weighting) should be used to reduce bias and obtain a set of occurrence records that are more representative geographically and environmentally (Phillips *et al*. 2009, Aiello-Lammens *et al*. 2015).

Capturing entire species ranges and avoiding model extrapolation

It is necessary to include occurrence records from throughout the species' range, even in the case of regional assessments, in order to avoid artificially truncating response curves when modelling the species' niche (Elith and Graham 2009, Thuiller *et al*. 2004). Excluding occurrences from outside the region of interest reduces the ability of the model to infer the full 'climate envelope' of the species. If, for instance, the current environmental conditions of a set of occurrence points in an area outside the region correspond to future projected conditions in some part of the region, then excluding those points from the model decreases the model's ability to predict areas correctly within the region that may become suitable in the future. Therefore, models based on data from part of the range (for example, only one country within a multi-national species' range) will generally be unacceptable. Caution must also be exercised when extrapolating model results under future climate scenarios (i.e., extrapolating in environmental space beyond the range of data used to build the model; Pearson *et al*. 2006). Extrapolation should be avoided where possible (e.g., Pearson *et al*. 2002), or else the behaviour of the model (i.e., the shape of response curves when extrapolating) should be known and well justified (see also the section on *Model complexity*, and Figure 12.2 below).

Background/pseudo-absence in the species distribution data

Model results are sensitive to the extent of the study region from which background or pseudo-absence samples are taken. It is therefore important to select an appropriate study region. In general, background and pseudo-absence records should be selected from areas in which the species' range is likely to be in equilibrium with the predictor variables, and should not be selected from areas where the species is absent due to non-climatic factors, such as dispersal limitation or inter-species competition (because such records provide a false-negative signal that will lead to poorer characterization of the species’ climatic requirements; Anderson and Raza 2010). Where possible, selection of the extent of the study region should therefore take into account factors including the dispersal capacity of the species and distributions of competitors. Acceptable methods include selecting background and pseudo-absence points from the specific ecoregion(s) where the species is present, or from an area that is within a specific distance from occurrence points. In the latter case, the distance used can have a substantial impact on the results. As a very general rule-of-thumb, the selection distance should be comparable to (but can be somewhat larger than) the 3-generation dispersal distance. In addition to a criterion of being *within* a specific distance of (or in the same bioregion as) the occurrences, selection of pseudo-absence points (but ***not*** background points) can also use a criterion excluding areas very close to occurrence points, so that the same set of environmental conditions are not associated both with a presence and an absence.

12.1.12.2 Environmental variables (the predictor variables)

Selection of environmental predictor variables

Predictor variables need to be carefully selected. It is important to select variables that are expected to exert direct influence on the distributions of the species (e.g., minimum temperature of the coldest month, maximum temperature of the warmest month, spring precipitation) through known eco-physiological mechanisms, and avoid indirect variables (e.g., elevation; see below) (e.g., Guisan and Zimermann 2000). Often, there are several candidate variables for modelling the distributions of species, but they tend to be correlated amongst each other. When this is the case, it is often advisable to investigate these correlations and select a reduced number of uncorrelated variables (to avoid problems of collinearity; Araújo and Guisan 2006, Dormann *et al*. 2013), unless it is demonstrated that the method used is insensitive to collinearity. Note that the number of predictor variables should not exceed (or even come close to) the number of species occurrence records that are used. As a general rule, no more than one predictor variable for every five observations should be used. Some methods (e.g., Maxent, Phillips *et al*. 2006; Boosted Regression Trees, Elith *et al*. 2008) select a parsimonious number of variables automatically, in which case the above rule would not apply. One reason to aim for parsimony in variable selection is to avoid overfitting of the models, thus increasing generality.

Global climate models and future emission scenarios

Climate models, commonly referred to as Global Climate Models (GCMs) or Earth System Models (ESMs), are key tools for modelling Earth's climate. GCMs represent complex relationships between different components of the climate system, including atmosphere, oceans, land surface, and ice, which are coupled to simulate Earth's climate. GCMs are used to develop projections of possible future climates under various assumptions such as changing future greenhouse gas emissions, land-use change and levels of climate mitigation. As of 2023, the newest generation of GCMs are based off of the sixth Coupled Model Intercomparison Project (CMIP6) and a synthesis of the results from these models is included in the IPCC 6th Assessment Report.

Given uncertainties around future rates of greenhouse gas emissions and associated mitigation measures, projections of future climates are based on various scenarios. Each of these scenarios makes different assumptions about future greenhouse gas emissions, land-use and other driving forces. Assumptions about future technological and economic developments are built into families of 'storylines', each describing alternative pathways for the future. Scenarios featured in the latest assessment are called Shared Socioeconomic Pathways (SSPs). SSP scenarios range from SSP1-2.6 (an optimistic scenario that assumes strong emission reductions through mitigation measures) through SSP2-4.5 (a moderate scenario roughly consistent with current trends) to SSP3-7.0 (where CO2 emissions roughly double by 2100).

In addition to uncertainties associated with the modelling algorithm and other factors (summarized in a later section), bioclimatic models projected to future climate conditions need to account for uncertainty associated with choice of GCMs and scenarios. It is recommended that assessments use projections of at least two scenarios (e.g., selected from among the three mentioned above), and at least two GCMs, in order to uncertainty in climate model projections. Guidance for GCM selection from CMIP5 are provided by Sanderson *et al.* (2015) and from CMIP6 by ISIMIP (the Intersectoral Impact Model Intercomparison Project). It is recommended that a bioclimatic model should be projected using each GCM/scenario combination individually, rather than using the ensemble mean projection (where an average across each of the GCMs is taken and then a multi-model mean projection is used to project future distributions), because the latter can result in loss of important variation as a result of averaging (Mahony *et al*. 2022). Furthermore, as emission scenarios are revised in future, the relevant Red List assessments based on them should be revised.

Note that it may be necessary to interpolate climatic variables, if they are not available for the time points required for a Red List assessment. For assessments using criterion A3, these time points include a year as close to the assessment year as possible ("current"), and the year that is 3-generations or 10 years (whichever is larger) after the "current" year, but no more than 100 years after it ("future"). Thus, for example, for a species with generation length of 10 years, assessed in 2024, climate data are needed for 2024 and 2054. Depending on the species generation length, this may require interpolation. Other time horizons are not acceptable for criterion A3 (see section 12.1.1), but shorter time horizons can be combined with past reduction for assessment under criterion A4 (see below). If generation length is uncertain, multiple time periods need to be used for each assessment, resulting in a range of outcomes (such as a range of projected reductions in range area).

Non-climatic variables and land-use masks

In addition to climate, non-climatic factors such as current and future land-use also constrain the distribution of species. Other non-climatic variables that constrain species distributions include soils, hydrology, topographic variables (e.g., slope; but see below for aspect and elevation), and human impact variables other than land-use (human footprint, distance to roads, etc.). Using non-climate variables is especially crucial for species whose bioclimatic envelope is predicted to shift through human-dominated landscapes. Assessments that rely on climate data alone are prone to over-predict areas of suitable habitat because climate may be suitable, but land cover may be unsuitable (Pearson *et al*. 2004). A land use map can be used as a mask to exclude such unsuitable areas from current and projected habitat. However, if land-use and climatic variables are likely to interact, then the land-use variables should be included in the model together with the climatic variables, rather than used as a mask (Stanton *et al.* 2012). Even if future land-use is not known, using the current land-use as a static variable in the model, or as a land-use mask, can increase the accuracy of the projections (Stanton *et al*. 2012), especially if land-use has a different spatial distribution in areas that will become more climatically suitable in the future than in the species' current range (e.g., if the areas that will become climatically suitable for the species in the future are dominated by more intensive human land-use compared to the current range). This is especially important for terrestrial species in lowlands, whose ability to track climate is hampered by habitat loss and fragmentation due to human activities such as agricultural practices (Lenoir *et al*. 2020).

If using aspect or other variables measured on a circular scale, assessors should not use a variable with a range of 0 to 359 degrees. Instead, one or both of two separate variables should be used, such as Northness=cosine(A*π/180) and Eastness=sine(A*π/180), where A is aspect in degrees, with 0 corresponding to North.

Variables such as elevation, latitude or longitude may serve as useful proxies for current climatic conditions, but they hinder the accuracy of future predictions because the relationships between these and climatic variables may change in the future. In particular, including elevation in the model is likely to result in the underestimation of the projected effects of future climate change. Models should not include elevation, latitude, or longitude, other than in exceptional cases. Elevation may be used if the model is only used to project the current range spatially, and not used for the purposes of projecting changes in the species distribution due to climate change.

Choosing an appropriate spatial resolution

Bioclimatic models have been fitted with data of varying resolutions, for instance ranging from 1-ha cells in Switzerland (Randin *et al*. 2009), to 2-degree latitude-longitude cells at a global level. There is commonly a trade-off between the geographical extent of the study area and the resolution of the data: studies across large areas are likely to use data at coarser resolutions than studies across smaller regions. Similarly, it is often necessary to use data at finer resolution when modelling the bioclimate envelope of restricted range species, whereas wide-ranging species may be effectively modelled using data at coarser resolutions. Also, when modelling species across regions with low spatial heterogeneity (e.g., flat terrain), coarser resolution data are less of a problem than when models are used across areas of high heterogeneity (e.g., rugged terrain). It is important to bear in mind, however, that analyses at coarse resolutions may not account for microclimates that may be important for species persistence (Pearson *et al*. 2006, Trivedi *et al*. 2008, Randin *et al*. 2009).

12.1.12.3 Model building and evaluation

Model selection

Many bioclimatic modelling techniques exist and most are acceptable to use in Red List assessments. However, it is important that assessments of species range changes are based on established methodologies that have been used and verified by several independent research groups. Note that modelling approaches based only on presence data (without background, pseudo-absence, or absence data) are not acceptable. Such approaches include the "climate-envelope" model BIOCLIM from the 1990s (not to be confused with the BioClim variables; see Booth *et al*. 2014), DOMAIN (Carpenter *at al*. 1993), and any approach that relies only on the values of climatic variables observed in the occurrence locations. Sometimes these are referred to as "rectilinear" approaches, but the shape of the response curve is not the reason they are unacceptable. They are unacceptable because they cannot eliminate variables that are not important for the species.

Model complexity

Model complexity refers to the functional forms of the response curves for each predictor variable and the interactions between the variables (as well as the number of predictor variables; see above). The functional forms can be linear, quadratic, or more complex, involving hinge and threshold features. Although it has been shown that agreements between predicted and observed distributions are often greater with models allowing complex response curves (e.g., Elith *et al*. 2006), the more complex forms should be used sparingly, if at all, because they could lead to overfitting and biologically implausible response shapes towards the margins of the data range (Merow *et al*. 2014).

The selection of functional forms, and the inclusion of interactions (i.e., product of two variables) should be guided by the knowledge of the species' biology (specifically, its response to the environmental variable). For example, quadratic forms can be justified for variables that have a narrow range of optimal values for the species, above and below which the species' performance declines.

Also important is the assumption about the functional forms beyond the current range of the predictor variables (this is related to "clamping" vs. not "clamping"). The assessors should state and justify their assumption about the species' response if and when the future value of a predictor variable is beyond its current range of values (see Figure 12.2). Possible assumptions include (a) that the response variable has a specific value (such as zero), (b) that the response variable has the value at the end of its current range, and (c) that the response variable has whatever value is given by the function (i.e., the function is extrapolated into the future range of the predictor variable without clamping or any other modification). Note that even if no variable is projected to have future values that are beyond the current range of values, combinations of parameters may be beyond the current range of the combinations of the variables (Zurell *et al*. 2012). If such novel climatic conditions or novel combinations are projected, they should be mentioned, and their potential effects discussed.

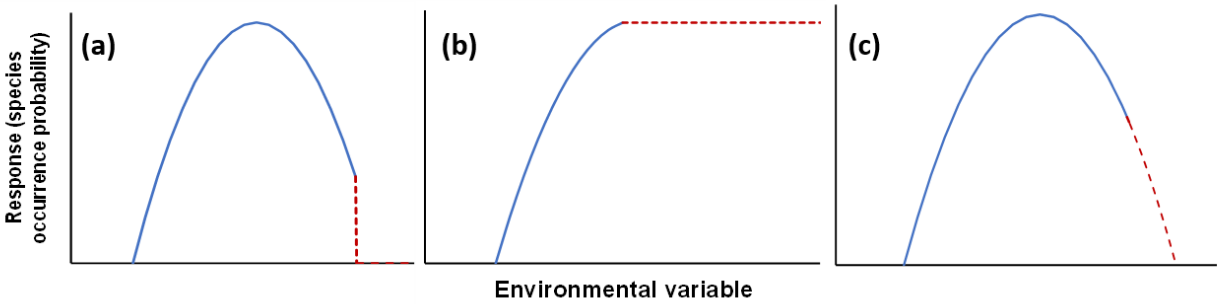


Figure 12.2. Three assumptions about the value of the response variable (e.g., probability of species occurrence) beyond the current range of an environmental variable. Blue, solid: species' response within the current range of the environmental variable. Red, dashed: species' assumed response beyond the upper limit of the current range of the environmental variable. See text for the descriptions of the assumptions (a) "set to zero", (b) "assume last value", and (c) "extrapolate function". These examples illustrate the case where the only values of the environmental variable beyond its current range are greater than the current range.

Model evaluation

The main risk in fitting complex models is overfitting, which occurs when the model predicts the training data (the data used in model-fitting) accurately, but not the species response under different conditions to those in the data. Most modelling approaches have procedures or parameters related to preventing the overfitting of complex models. For MaxEnt and related methods, report the regularization coefficients, β and the methods used to identify its optimal value. For other models, report the approach used (e.g., AIC and BIC diagnostics for model comparison and selection in parametric methods, or cross-validation procedures for optimizing tree size in tree-based methods).

In addition, the model should be evaluated in terms of its predictive power. The recommended approach is any type of k-fold cross-validation test that separates the data into independent testing and training parts (10 to 100 times, called "splits"), fits the model to the training data set, and reports the model's ability to predict the testing data. Assessors should report these results using, for instance, Test AUC. The recommended approach for separating the data into independent testing and training parts is blocking. This can be spatial blocking, environmental blocking, temporal blocking, or spatio-temporal blocking (Roberts *et al*. 2017). If the number of occurrences is very small, N–1 Jackknife method can be used (Pearson *et al*. 2007).

An important type of model evaluation is assessing model outputs with respect to the biology of the species. Assessors should interpret the results, including the current and future maps, variable importances, and univariate response plots in terms of the biology of the species, checking if they make ecological sense. It is critical that the predicted current distribution of the species matches the areas known or strongly suspected to be occupied by the species according to experts familiar with the species. Similarly, future projected range of the species, the variables found to be important determinants of the species niche, and the relationships of these variables to species occurrence (the response plots) should be consistent with what is known about the ecology of the species.

Incorporating uncertainties and assessing the robustness of model projections

Studies have shown that projections from alternative models can be so variable as to compromise assessment of whether species potential distributions should contract or expand for any given climate scenario (e.g., Araújo *et al*. 2005, Araújo *et al*. 2006, Pearson *et al*. 2006, Buisson *et al*. 2010). Assessments of the temporal trends in the sizes of species potential distributions should, therefore, include an assessment of the robustness of the projections by comparing results of a range of bioclimatic modelling techniques. We suggest that at least two, preferably three, modelling techniques should be compared and be as independent as possible with regards to how they link the response and the predictor variables (e.g., GAM and GLM are conceptually similar and tend to produce similar results). Various strategies may be employed in cases when models forecast inconsistent trends. One such strategy is to investigate the cause of the discrepancies. Typically, this would involve investigation of the species response curves obtained with each one of the methods, evaluating if there is any clear error or inconsistency with the biology of the species, and then selecting the projections by the method producing more reasonable results. This approach is useful for species with well-known ecologies where expert judgements can be made and contrasted with the model outputs. The downside of the approach is that it involves subjective judgement that may yield non-repeatable results. An alternative strategy is to run ensembles of forecasts using several established approaches and then combine the individual model projections through consensus methodologies (for a review see Araújo and New 2007). The disadvantage here is that potentially significant ecological knowledge is not being used. Note that alternative modelling approaches may require different response variables (e.g., MaxEnt requires background points whereas GLM methods require absence or pseudo-absence points). In such cases the same set of points cannot be used for all models.

Besides the modelling approach, there can be many other sources of uncertainty, including:

- Spatial and temporal uncertainty in occurrence records.
- Spatial and temporal uncertainty and resolution of climate and non-climate environmental data.
- Global Climate models (GCMs).
- Climate (emission) scenarios (RCPs/SSPs).
- Downscaling method for climate models.
- Degrees of model complexity and method for fitting.
- Methods for predicting beyond the current range of variables (or combinations of variables).
- Methods for variable selection.
- Methods for bias correction and identifying relevant biases.
- Regions from which to sample background points.
- Dispersal distances (restricting the projected range expansion; see below).
- Thresholding methods (see below).
- Generation length (determining the time points for which to calculate reduction).
- Uncertainty associated with extrapolation to novel conditions.
- Uncertainty associated with ensembles (if used).

Ideally, the results should be obtained using all combinations of these uncertainties by running models with at least two values (e.g., min and max, or two different models) for each of the above. When this is impractical, the same approach should be used for the sources that are suspected to result in the largest uncertainty in the results.

Using appropriate metrics of species range changes

Bioclimate models may be useful to assess trends in the availability of suitable climate conditions for species, but in order to infer population change (for criterion A3, for instance), the results need to exclude areas that are unlikely to be occupied by the species. This type of exclusion is important to avoid overestimating the future range expansion. In addition to the climatic and land-use variables included in the model (which related to physiological and habitat tolerances of the species), the range of the species may also be limited by biological processes such as dispersal of the focal species, dispersal of its mutualists, demographic stochasticity, and Allee effects, or by human activities such as direct exploitation (hunting, poaching). Recommended approaches include the following.

(i) Exclude areas that have a low probability of being suitable for the species. The results of the models can be used in two ways. One is based on the sum of probabilities or suitability indices from the model, and the other is based on measuring the potential area occupied by the species after transforming probabilities (or suitabilities) into estimates of presence and absence. To make such a transformation, it is necessary to use thresholds (see, for example, Liu *et al*. 2005), although a threshold can also be applied for the first method (before summing the probabilities). Use of the lowest presence threshold (e.g., Pearson *et al*. 2007) may be justified in cases with very few occurrence records, but balancing sensitivity and specificity is more appropriate when a larger number of presence/absence records are available. Sensitivity of results to the selection of alternative methods for defining thresholds should be examined. However, it should be noted that the measures of change in climate suitability that are relevant to red listing are relative measures (of proportional change in time) and these are, in principle, robust to alternative methods for defining thresholds. The absolute areas (of range or potential habitat) should not be used as part of assessments of species extinction risk under climate change because estimates of change from bioclimate models are very sensitive to the thresholds used. Note that thresholds may also be used when converting habitat suitability to population size (see section 12.1.9).

(ii) Exclude areas that are unlikely to be occupied by the species because of human land use, by using a land-use mask. Even when future land-use scenarios are available, the current land-use can be preferred because projected land-use maps usually have a low spatial resolution (coarse grain). As mentioned above, this is especially important if human land-use is more (or less) intensive in areas that will become more climatically suitable in the future than in the species' current range.

(iii) Exclude areas that are unlikely to be occupied by the species because of dispersal limitations. This can be done in two ways. One is excluding areas that are beyond natural and human-made dispersal barriers (e.g., areas that are separated from the current occupied area by a large river may be excluded). The other is excluding areas that are beyond the species' dispersal abilities, by allowing range expansion only up to a specific distance from the current occupied range of the species. The specific distance may represent the three-generation dispersal distance of the species, which can be calculated as dispersal distance × 3 × generation length / age of first reproduction. For example, if a species can disperse up to 4 km, has a generation length of 10 years, and the age of first reproduction of 5 years, it is possible that in 3 generations (30 years), the populations can disperse, reach maturity, and reproduce up to 6 times (30 yr/5 yr). In this case, the three-generation dispersal distance could be as high as 24 km. Uncertainty in this should be incorporated into the model runs (see above). Note that dispersal distance is frequently underestimated because the longest dispersal events are rarely observed, so inferred or modelled dispersal distances (e.g., based on observed range expansion rates or on mechanistic dispersal models) can be more appropriate than observed distances for this purpose. Assessors should make sure that the calculated distance is considered by species experts to be consistent with the biology of the species. Note that these areas could also be excluded from the model building (i.e., no background points; see above). Scenarios of "no dispersal" and "full dispersal" are not recommended unless they are well-justified (or give very similar results), because they tend to be overly pessimistic and overly optimistic, respectively, about the species' persistence.

(iv) Exclude areas that are unlikely to be occupied by the species because of species interactions. This can be done in different ways. One is to allow range expansion of the focal species only within the projected range of another species on which the focal species is an obligate dependent (e.g., an obligate mutualist or parasite). The projection of the other species' range for this purpose should be for 3 generations of the focal species. This is especially important if the mutualist species has more limited dispersal or a higher age at first reproduction. Note that this is only for the purpose of a more accurate prediction of the focal species' range expansion, not for modelling the species interaction itself. The second way to incorporate species interactions is to allow range expansion only within the ecoregions that the species occur currently or have occurred in the past. Although ecoregion boundaries will also shift in the future, this limitation may be useful as a very crude way of taking into account the complex dependencies of the focal species on the ecological community in which it is embedded. Note that ecoregion boundaries can also be used to restrict sampling of background points (see above).

Note that the above range exclusions can be used in combination.

In addition to potentially overestimating the species' future range for the reasons mentioned above, SDMs may also underestimate it. One reason underestimation may happen is genetic adaptation, but this is unlikely to have a substantial effect for most species within the time frames considered in IUCN Red List assessments. Another reason may be the plasticity of the species' response to climate, or tolerances of the species to conditions beyond those in their current range. If this is considered plausible, it may be incorporated into the model in many cases through the assumptions discussed above about the response of the species to conditions beyond those in its current range (see Figure 12.2).

See section 12.1.9 on how to use the climate model results to infer population reduction for assessing the species with criteria A3 and A4.

13. References

Aiello-Lammens, M.E., Boria, R.A., Radosavljevic, A., Vilela, B. and Anderson, R.P. 2015. spThin: an R package for spatial thinning of species occurrence records for use in ecological niche models. *Ecography* 38: 541-545.

Akçakaya, H.R. 2000. Population viability analyses with demographically and spatially structured models. *Ecological Bulletins* 48: 23–38.

Akçakaya, H.R. 2002. Estimating the variance of survival rates and fecundities. *Animal Conservation* 5: 333–336.

Akçakaya, H.R. and Raphael, M.G. 1998. Assessing human impact despite uncertainty: viability of the northern spotted owl metapopulation in the northwestern USA. *Biodiversity and Conservation* 7: 875–894.

Akçakaya H.R. and Sjögren-Gulve, P. 2000. Population viability analysis in conservation planning: an overview. *Ecological Bulletins* 48: 9–21.

Akçakaya, H.R., Ferson, S., Burgman, M.A., Keith, D.A., Mace, G.M. and Todd, C.A. 2000. Making consistent IUCN classifications under uncertainty. *Conservation Biology* 14: 1001–1013.

Akçakaya H.R, Burgman M.A., Kindvall O., Wood C.C., Sjogren-Gulve P., Hatfield J.S. and McCarthy M.A. 2004. '*Species Conservation and Management: case studies*.' Oxford University Press, Oxford.

Akçakaya, H.R., Butchart, S.H.M., Mace, G.M., Stuart, S.N. and Hilton-Taylor, C. 2006. Use and misuse of the IUCN Red List Criteria in projecting climate change impacts on biodiversity. *Global Change Biology* 12: 2037–2043.

Akçakaya, H.R., Keith, D.A., Burgman, M., Butchart, S.H.M., Hoffmann, M., Regan, H.M., Harrison, I., and Boakes, E. 2017. Inferring extinctions III: A cost-benefit framework for listing extinct species. *Biological Conservation* 214: 336–342.

Akçakaya, H.R., Hochkirch, A., Bried, J.T., van Grunsven, R.H.A., Simaika, J.P., De Knijf, G. and Henriques, S. 2021. Calculating population reductions of invertebrate species for IUCN Red List assessments. *Journal of Insect Conservation* 25: 377–382.

Anderson, B., Akçakaya, H.R., Araújo, M., Fordham, D., Martinez-Meyer, E., Thuiller, W. and Brook, B.W. 2009. Dynamics of range margins for metapopulations under climate change. *Proceedings of the Royal Society B* 276: 1415–1420.

Anderson, R.P. 2023. Integrating habitat-masked range maps with quantifications of prevalence to estimate area of occupancy in IUCN assessments. *Conservation Biology* 37: e14019.

Anderson, R.P. and Raza, A. 2010. The effect of the extent of the study region on GIS models of species geographic distributions and estimates of niche evolution: preliminary tests with montane rodents (genus *Nephelomys*) in Venezuela. *Journal of Biogeography* 37: 1378–1393.

Araújo, M.B. and Guisan, A. 2006. Five (or so) challenges for species distribution modelling. *Journal of Biogeography* 33: 1677–1688.

Araújo, M.B. and New, M. 2007. Ensemble forecasting of species distributions. *Trends in Ecology and Evolution* 22: 42–47.

Araújo, M.B., Pearson, R.G., Thuiller, W. and Erhard, M. 2005. Validation of species-climate impact models under climate change. *Global Change Biology* 11: 1504–1513.

Araújo, M.B., Whittaker, R.J., Ladle, R.J. and Erhard, M. 2005. Reducing uncertainty in extinction risk from climate change. *Global Ecology and Biogeography* 14: 529–538.

Bird, J.P., Martin, R., Akçakaya, H.R., Gilroy, J., Burfield, I.J., Garnett, S.T., Symes, A., Taylor, J., Şekercioğlu, Ç.H. and Butchart, S.H.M. 2020. Generation lengths of the world's birds and their implications for extinction risk. *Conservation Biology* 34: 1252–1261.

BirdLife International. 2004. Threatened birds of the world 2004. CD-ROM. BirdLife International, Cambridge, UK.

Boyce, M.S. 1992. Population viability analysis. *Annual Review of Ecology and Systematics* 23: 481–506.

Brook, B.W., Akçakaya, H.R., Keith, D.A., Mace, G.M., Pearson, R.G. and Araújo, M.B. 2009. Integrating bioclimate with population models to improve forecasts of species extinctions under climate change. *Biology Letters* 5: 723–725.

Brook, B.W., O'Grady, J.J., Chapman, A.P., Burgman, M.A., Akçakaya, H.R. and Frankham, R. 2000. Predictive accuracy of population viability analysis in conservation biology. *Nature* 404: 385–387.

Brooks, T.M., Pimm, S.L., Akçakaya, H.R., Buchanan, G.M., Butchart, S.H.M., Foden, W., Hilton-Taylor, C., Hoffmann, M., Jenkins, C.N., Joppa, L., Li, B.V., Menon, V., Ocampo-Peñuela, N. and Rondinini, C. 2019. Utility of Area of Habitat for the IUCN Red List. *Trends in Ecology and Evolution.* 34: 977–986. DOI: 10.1016/j.tree.2019.06.009.

Buisson, L., Thuiller, W., Casajus, N., Lek, S. and Grenouillet, G. 2010. Uncertainty in ensemble forecasting of species distribution. *Global Change Biology* 16: 1145-1157.

Burgman, M.A. and Fox, J.C. 2003. Bias in species range estimates from minimum convex polygons: implications for conservation and options for improved planning. *Animal Conservation* 6: 19–28.

Burgman, M.A., Keith, D.A. and Walshe, T.V. 1999. Uncertainty in comparative risk analysis of threatened Australian plant species. *Risk Analysis* 19: 585–598.

Burgman, M.A., Ferson, S. and Akçakaya, H.R. 1993. *Risk Assessment in Conservation Biology*. Chapman and Hall, London.

Bustamante, M.R. 2002. Monitoreos de poblaciones de anuros en los Andes de Ecuador, con énfasis en el estudio poblacional de las especies del Bosque protector Cashca Totoras (Provincia Bolívar). Tesis de Licenciatura, Pontificia Universidad católica del Ecuador.

Butchart, S.H.M. and Bird, J. 2009. Data Deficient birds on the IUCN Red List: what don't we know and why does it matter? *Biological Conservation* 143: 239–247.

Butchart, S.H.M., Stattersfield, A.J. and Brooks, T.M. 2006. Going or gone: defining ‘Possibly Extinct’ species to give a truer picture of recent extinctions. *Bulletin of the British Ornithologists Club* 126A: 7–24.

Butchart, S.H.M., Lowe, S., Martin, R.W., Symes, A., Westrip, J.R.S. and Wheatley, H. 2018. Which bird species have gone extinct? A novel quantitative classification approach. *Biological Conservation* 227: 9–18.

Cabral, J.S., Jeltsch, F, Thuiller, W., Higgins, S., Midgley, G.F., Rebelo, A.G., Rouget, M. and Schurr, F.M. 2013. Impacts of past habitat loss and future climate change on the range dynamics of South African Proteaceae. *Diversity and Distributions* 19: 363–376.

Cardoso, P., Borges, P.A., Triantis, K.A., Ferrández, M.A. and Martín, J.L. 2011. Adapting the IUCN red list criteria for invertebrates. *Biological Conservation* 144: 2432–2440.

Clarke, H., Lucas, C. and Smith, P. 2012. Changes in Australian fire weather between 1973 and 2010. *International Journal of Climatology* 33(4): 931–944, DOI: 10.1002/joc.3480.

Coleman, F.C., Koenig, C.C. and Collins, L.A. 1996. Reproductive styles of shallow-water groupers (Pisces: Serranidae) in the eastern Gulf of Mexico and the consequences of fishing spawning aggregations. *Environmental Biology of Fishes* 47: 129–141.

Collar, N.J. 1998. Extinction by assumption; or, the Romeo Error on Cebu. *Oryx* 32: 239–244.

Collen, B., Dulvy, N.K., Gaston, K.J., Gärdenfors, U., Keith, D.A., Punt, A.E., Regan, H.M., Böhm, M., Hedges, S., Seddon, M., Butchart, S.H.M., Hilton-Taylor, C., Hoffmann, M., Bachman, S.P. and Akçakaya, H.R. 2016. Clarifying misconceptions of extinction risk assessment with the IUCN Red List. *Biology Letters* 12: 20150843.

Dale, V.H., Joyce, L.A., McNulty, S., Neilson, R.P., Ayres, M.P., Flannigan, M.D., Hanson, P.J., Irland, L.C., Lugo, A.E., Peterson, C.J., Simberloff, D., Swanson, F.J., Stocks, B.J. and Wotton, B.M. 2001. Climate change and forest disturbances. *BioScience* 51(9): 723–734.

DeAngelis, D.L. and Diaz, S.G. 2019. Decision-making in agent-based modeling: a current review and future prospectus. *Frontiers in Ecology and Evolution* 6: DOI: 10.3389/fevo.2018.00237.

Dennis, B., Munholland, P.L. and Scott, J.M. 1991. Estimation of growth and extinction parameters for endangered species. *Ecological Monographs* 61: 115–143.

Deutsch, C.A., Tewksbury, J.J., Huey, R.B., Sheldon, K.S., Ghalambor, C.K., Haak, D.C. and Martin, P.R., 2008. Impacts of climate warming on terrestrial ectotherms across latitude. *PNAS* 105(18): 6668–6672.

Dormann, C.F., Elith, J., Bacher, S., Buchmann, C., Carl, G., Carré, G., Marquéz, J.R.G., Gruber, B., Lafourcade, B., Leitão, P.J., Münkemüller, T., McClean, C., Osborne, P.E., Reineking, B., Schröder, B., Skidmore, A.K., Zurell, D. and Lautenbach, S. 2013. Collinearity: A review of methods to deal with it and a simulation study evaluating their performance. *Ecography* 36: 27-46.

Dutson, G.C.L., Magsalay, P.M. and Timmins, R.J. 1993. The rediscovery of the Cebu Flowerpecker *Dicaeum quadricolor*, with notes on other forest birds on Cebu, Philippines. *Bird Conservation International* 3: 235–243.

Elith, J. and Graham, C.H. 2009. Do they / How do they / WHY do they differ? - on finding reasons for differing performances of species distribution models. *Ecography* 32: 66–77.

Elith, J., Graham, C. and the NCEAS species distribution modeling group. 2006. Novel methods improve prediction of species' distributions from occurrence data. *Ecography* 29: 129–151.

Elith, J., Leathwick, J.R. and Hastie, T. 2008. A working guide to boosted regression trees. *Journal of Animal Ecology* 77: 802–813.

Ferson, S., Root, W. and Kuhn, R. 1998. *RAMAS Risk Calc: Risk Assessment with Uncertain Numbers.* Applied Biomathematics, Setauket, New York.

Fielding, A.H. and Bell, J.F. 1997. A review of methods for the assessment of prediction errors in conservation presence/absence models. *Environmental Conservation* 24: 38–49.

Foden, W.B., Butchart, S.H.M., Stuart, S.N., Vié, J.-C., Akçakaya, H.R., Angulo, A., DeVantier, L.M., Gutsche, A., Turak, E., Cao, L., Donner, S.D., Katariya, V., Bernard, R., Holland, R.A., Hughes, A.F., O'Hanlon, S.E., Garnett, S.T., Şekercioğlu, Ç.H. and Mace, G.M. 2013. Identifying the world's most climate change vulnerable species: a systematic trait-based assessment of all birds, amphibians and corals. *PLoS ONE* 8(6): e65427. DOI:10.1371/journal.pone.0065427.

Franklin, J. 2010. *Mapping Species Distributions: Spatial Inference and Prediction.* Cambridge University Press, UK.

Gärdenfors, U. 2000. Population viability analysis in the classification of threatened species: problems and potentials. *Ecological Bulletins* 48: 181–190.

Gärdenfors, U., Hilton-Taylor, C., Mace, G.M. and Rodríguez, J.P. 2001. The application of IUCN Red List Criteria at regional levels. *Conservation Biology* 15: 1206–1212.

Gaston, K. J. 1996. The multiple forms of the interspecific abundance-distribution relationship. *Oikos* 76: 211–220.

Gilman, S.E., Urban, M.C., Tewksbury, J., Gilchrist, G.W. and Holt, R.D. 2010. A framework for community interactions under climate change. *Trends in Ecology and Evolution* 25(6): 325–331.

Gould, W.R. and Nichols, J.D. 1998. Estimation of temporal variability of survival in animal populations. *Ecology* 79: 2531–2538.

Guisan, A. and Zimmerman, N. 2000. Predictive habitat distribution models in ecology. *Ecological Modelling* 135: 147–186.

Guisan, A. and Thuiller, W. 2005. Predicting species distribution: Offering more than simple habitat models. *Ecology Letters* 8: 993–1009.

Hallingbäck, T., Hodgetts, N., Raeymaekers, G., Schumacker, R., Sérgio, C., Söderström, L., Stewart, N. and 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.

Hampe, A. 2004. Bioclimatic models: what they detect and what they hide. *Global Ecology and Biogeography* 11: 469–471.

He, F. and Gaston, K.J. 2000. Estimating species abundance from occurrence. *The American Naturalist* 156: 553–559.

Heikkinen, R.K., Luoto, M., Araújo, M.B., Virkkala, R., Thuiller, W. and Sykes, M.T. 2006. Methods and uncertainties in bioclimatic modelling under climate change. *Progress in Physical Geography* 30: 751–777.

Hobday, A.J., Tenger, M.J. and Hakker, P.L. 2001. Over-exploitation of a broadcast spawning marine invertebrate: decline of the white abalone. Reviews in Fish Biology and Fisheries 10: 493–514.

Hoffmann, A.A. and Sgrò, C.M. 2011. Climate change and evolutionary adaptation. *Nature* 470: 479–485.

IPCC. 2013. *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change.* Stocker, T.F., D. Qin, G.-K. Plattner, M. Tignor, S.K. Allen, J. Boschung, A. Nauels, Y. Xia, V. Bex and P.M. Midgley (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA. Downloaded from http://ipcc.ch/report/ar5/wg1/ (14 Jan 2016).

IUCN. 1994. *IUCN Red List Categories*. IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, U.K.

IUCN. 1998. *Guidelines for Re-introductions*. Prepared by the IUCN SSC Re-introduction Specialist Group. IUCN, Gland, Switzerland and Cambridge, U.K.

IUCN. 2001. *IUCN Red List Categories and Criteria: Version 3.1.* IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, U.K.

IUCN. 2003. *Guidelines for Application of IUCN Criteria at Regional Levels. Version 3.0.* IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, UK.

IUCN. 2012a. *Guidelines for Application of IUCN Red List Criteria at Regional and National Levels: Version 4.0.* IUCN, Gland, Switzerland and Cambridge, UK. Available at www.iucnredlist.org/technical-documents/categories-and-criteria

IUCN. 2012b. *IUCN Red List Categories and Criteria: Version 3.1. Second edition.* IUCN, Gland, Switzerland and Cambridge, UK. Available at www.iucnredlist.org/technical-documents/categories-and-criteria

IUCN. 2021. *IUCN Green Status of Species: A global standard for measuring species recovery and assessing conservation impact.* Version 2.0. Gland, Switzerland: IUCN. Available at https://www.iucnredlist.org/resources/green-status-assessment-materials

Juniper, T. 2003. *Spix's Macaw: the Race to Save the World's Rarest Bird*. Fourth Estate, London.

Kearney M. and Porter W. 2009. Mechanistic niche modelling: combining physiological and spatial data to predict species' ranges. *Ecology Letters* 12: 334–350.

Keith, D.A. 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, D., Akçakaya, H.R., Butchart, S.H.M., Collen, B., Dulvy, N.K., Holmes, E.E., Hutchings, J.A., Keinath, D., Schwartz, M.K., Shelton, A.O. and Waples, R.S. 2015. Temporal correlations in population trends: conservation implications from time-series analysis of diverse animal taxa. *Biological Conservation* 192: 247–257.

Keith, D.A., Auld, T.D., Ooi, M.K.J. and Mackenzie, B.D.E. 2000. Sensitivity analyses of decision rules in World Conservation Union (IUCN) Red List criteria using Australian plants. *Biological Conservation* 94: 311–319.

Keith, D., McCarthy, M.A., Regan, H., Regan, T., Bowles, C., Drill, C., Craig, C., Pellow, B., Burgman, M.A., Master, L.L., Ruckelshaus, M., Mackenzie, B., Andelman, S.J. and Wade, P.R. 2004. Protocols for listing threatened species can forecast extinction. *Ecology Letters* 7: 1101–1108.

Keith, D.A, Akçakaya, H.R., Thuiller, W., Midgley, G.F., Pearson, R.G., Phillips, S.J., Regan, H.M., Araújo, M.B. and Rebelo, A.G. 2008. Predicting extinction risks under climate change: coupling stochastic population models with dynamic bioclimatic habitat models. *Biology Letters* 4: 560–563.

Keith, D.A., Mahony, M., Hines, H., Elith, J., Regan, T.J., Baumgartner, J.B., Hunter, D., Heard, G.W., Mitchell, N.J., Parris, K.M., Penman, T., Scheele, B.E.N., Simpson, C.C., Tingley, R., Tracy, C.R., West, M. and Akçakaya, H.R. 2014. Detecting extinction risk from climate change by IUCN Red List Criteria. *Conservation Biology* 28: 810–819.

Keith, D.A., Butchart, S.H.M., Regan, H.M., Harrison, I., Akçakaya, H.R., Solow, A.R. and Burgman, M.A. 2017. Inferring extinctions I: A structured method using information on threats. *Biological Conservation* 214: 320–327.

Keith, D.A., Akçakaya, H.R. and Murray, N.J. 2018. Scaling range sizes to threats for robust predictions of risks to biodiversity. *Conservation Biology* 32: 322–332. DOI: 10.1111/cobi.12988.

Kindvall, O. 2000. Comparative precision of three spatially realistic simulation models of metapopulation dynamics. *Ecological Bulletins* 48: 101–110.

Kunin, W.E. 1998. Extrapolating species abundance across spatial scales. *Science* 281: 1513–1515.

Lacy, R.C. 2000. Considering threats to the viability of small populations with individual-based models. *Ecological Bulletins* 48: 39–51.

Lam, J.T.L. 2009. Shortened Nassau Grouper spawning ban. *Society for the Conservation of Reef Fish Aggregations Newsletter* 13: 4.

Lande, R., Engen, S. and Saether, B.-E. 2003. *Stochastic population dynamics in ecology and evolution*. Oxford University Press, Oxford, UK.

Liu, C., Berry P.M., Dawson T.P. and Pearson, R.G. 2005. Selecting thresholds of occurrence in the prediction of species distributions. *Ecography* 28: 385–393.

Mace, G.M. and Lande, R. 1991. Assessing extinction threats: toward a reevaluation of IUCN threatened species categories. *Conservation Biology* 5: 148–157.

Mace, G.M., Collar, N.J., Gaston, K.J., Hilton-Taylor, C., Akçakaya, H.R., Leader-Williams, N., Milner-Gulland, E.J. and Stuart, S.N. 2008. Quantification of extinction risk: IUCN's system for classifying threatened species. *Conservation Biology* 22: 1424–1442.

Mackenzie, D.I., Nichols, J.D., Royle, J., Pollock, K.H., Bailey, L.L. and Hines, J. 2017. *Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence*. Second Edition. Academic Press.

Magsalay, P., Brooks, T., Dutson, G. and Timmins, R. 1995. Extinction and conservation on Cebu. *Nature* 373: 294.

McCarthy, M.A., Possingham, H.P., Day, J.R. and Tyre, A.J. 2001. Testing the accuracy of population viability analysis. *Conservation Biology* 15: 1030–1038.

Memmott, J., Craze, P.G., Waser, N.M. and Price, M.V. 2007. Global warming and the disruption of plant–pollinator interactions. *Ecology Letters* 10: 710–717.

Merow, C., Smith, M.J., Edwards Jr, T.C., Guisan, A., McMahon, S.M., Normand, S., Thuiller, W., Wüest, R.O., Zimmermann, N.E. & Elith, J. 2014. What do we gain from simplicity versus complexity in species distribution models? *Ecography* 37: 1267-1281.

Miller R.M., Rodríguez, J.P., Aniskowicz-Fowler, T., Bambaradeniya, C., Boles, R., Eaton, M.A., Gärdenfors, U., Keller, V., Molur, S., Walker, S. and Pollock, C. 2007. National threatened species listing based on IUCN criteria and regional guidelines: Current status and future perspectives. *Conservation Biology* 21: 684–696.

Milly, P.C.D., Wetherald, R.T., Dunne, K.A. and Delworth, T.L. 2002. Increasing risk of great floods in a changing climate. *Nature* 415: 514–517.

Ocampo-Peñuela, N., Jenkins, C.N., Vijay, V., Li, B.V. and Pimm, S.L. 2016. Incorporating explicit geospatial data shows more species at risk of extinction than the current Red List. *Science Advances* 2: e1601367.

Orr, J.C., Fabry, V.J., Aumont, O., Bopp, L., Doney, S.C., Feely, R.A., Gnanadesikan, A., Gruber, N., Ishida, A., Joos, F., Key, R.M., Lindsay, K., Maier-Reimer, E., Matear, R., Monfray, P., Mouchet, A., Najjar, R.G., Plattner, G.K., Rodgers, K.B., Sabine, C.L., Sarmiento, J.L., Schlitzer, R., Slater, R.D., Totterdell, R.J., Weirig, M.F., Yamanaka, Y. and Yoo, A. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature* 437: 681–686.

Ostro, L.E.T., Young, T.P., Silver, S.C. and Koontz, F.W. 1999. A geographic information system method for estimating home range size. *Journal of Wildlife Management* 63: 748–755.

Pacifici, M., Santini, L., Di Marco, M., Baisero, D., Francucci, L., Grottolo Marasini, G., Visconti, P., and Rondinini, C. 2013. Generation length for mammals. *Nature Conservation* 5: 87–94.

Pearson, R.G. 2006. Climate change and the migration capacity of species. *Trends in Ecology and Evolution* 21: 111–113.

Pearson, R.G. 2007. Species Distribution Modeling for Conservation Educators and Practitioners. *Lessons in Conservation* 3: 54–89. Available at: https://www.amnh.org/content/download/141368/2285424/file/LinC3_SpeciesDistModeling.pdf.

Pearson, R.G. and Dawson, T.P. 2003. Predicting the impacts of climate change on the distribution of species: Are bioclimate envelope models useful? *Global Ecology and Biogeography* 12: 361–371.

Pearson, R.G., Raxworthy, C.J., Nakamura, M. and Peterson, A.T. 2007. Predicting species' distributions from small numbers of occurrence records: a test case using cryptic geckos in Madagascar. *Journal of Biogeography* 34: 102–117.

Pearson, R.G., Dawson, T.P., Berry, P.M. and Harrison, P.A. 2002. Species: A spatial evaluation of climate impact on the envelope of species. *Ecological Modelling* 154: 289–300.

Pearson, R.G., Thuiller, W., Araújo, M.B., Martinez, E., Brotons, L., McClean, C., Miles, L., Segurado, P., Dawson, T. and Lees, D. 2006. Model-based uncertainty in species' range prediction. *Journal of Biogeography* 33: 1704–1711.

Pearson, R.G., Stanton, J.C., Shoemaker, K.T., Aiello-Lammens, M.E., Ersts, P.J., Horning, N., Fordham, D.A., Raxworthy, C.J., Ryu, H., McNees, J. and Akçakaya, H.R. 2014. Life history and spatial traits predict extinction risk due to climate change. *Nature Climate Change* 4: 217–221.

Peterson, A.T., Soberón, J., Pearson, R.G., Anderson, R.P., Martínez-Meyer, E., Nakamura, M. and Araújo, M.B. 2011. *Ecological Niches and Geographical Distributions: A Modeling Perspective*. Princeton University Press, Princeton, New Jersey.

Phillips, S.J., Anderson, R.P. and Schapire, R.E. 2006. Maximum entropy modeling of species geographic distributions. *Ecological Modelling* 190: 231–259.

Phillips, S.J., Dudík, M., Elith, J., Graham, C.H., Lehmann, A., Leathwick, J. and Ferrier, S. 2009. Sample selection bias and presence-only distribution models: Implications for background and pseudo-absence data. *Ecological Applications* 19: 181-197.

Porszt, E.J., Peterman, R.M., Dulvy, N.K., Cooper, A.B. and Irvine, J.R. 2012. Reliability of indicators of decline in abundance. *Conservation Biology* 26: 894–904.

Pratt, T.K., Fancy, S.G. and Ralph, C.J. 2001. 'Akiapola'au (*Hemignathus munroi*) and Nukupu'u (*Hemignathus lucidus*). In: A. Poole and F. Gill (eds) *The Birds of North America*, no. 600. The Birds of North America, Philadelphia, PA.

Randin, C.F., Engler, R., Normand, S., Zappa, M., Zimmermann, N.E., Pearman, P.B., Vittoz, P., Thuiller, W. and Guisan, A. 2009: Climate change and plant distribution: local models predict high-elevation persistence. *Global Change Biology* 15: 1557–1569.

Redford, K.H., Amato, G., Baillie, J., Beldomenico, P., Bennett, E.L., Clum, N., Cook, R., Fonseca, G., Hedges, S., Launay, F., Lieberman, S., Mace, G.M., Murayama, A., Putnam, A., Robinson, J.G., Rosenbaum, H., Sanderson, E.W. Stuart, S.N., Thomas, P. and Thorbjarnarson, J. 2011. What does it mean to successfully conserve a (vertebrate) species? *BioScience* 61: 39–48.

Reynolds, M.H. and Snetsinger, J.J. 2001. The Hawaii rare bird search 1994–1996. *Studies in Avian Biology* 22: 133–143.

Rodríguez, J.P. 2002. Range contraction in declining North American bird populations. *Ecological Applications* 12: 238–248.

Sadovy, Y. and Cheung, W.L. 2003. Near extinction of a highly fecund fish: the one that nearly got away. *Fish and Fisheries* 4: 86–99.

Sadovy, Y. and Domeier, M. 2005. Are aggregation-fisheries sustainable? Reef fish fisheries as a case study. *Coral Reefs* 24: 254–262.

Sanderson, B.M., Knutti, R. and Caldwell, P. 2015. A representative democracy to reduce interdependency in a multimodel ensemble. *Journal of Climate* 28: 5171-5194.

Sethi, S.A., Branch, T.A. and Watson, R. 2010. Global fishery development patterns are driven by profit but not trophic level. *PNAS* 207: 12163–12167.

Shoo, L.P., Williams, S.E. and Hero, J.-M. 2005. Climate warming and the rainforest birds of the Australian wet tropics: using abundance data as a sensitive predictor of change in total population size. *Biological Conservation* 125: 335–343.

Sjögren-Gulve, P. and Hanski, I. 2000. Metapopulation viability analysis using occupancy model. *Ecological Bulletins* 48: 53–71.

Stanton, J.C. 2014. Present-day risk assessment would have predicted the extinction of the passenger pigeon (*Ectopistes migratorius*). *Biological. Conservation.* 180: 11–20.

Stanton, J.C., Pearson, R.G., Horning, N, Ersts, P. and Akçakaya, H.R. 2012. Combining static and dynamic variables in species distribution models under climate change. *Methods in Ecology and Evolution* 3: 349–357.

Stanton, J.C., Shoemaker, K.T., Pearson, R.G. and Akçakaya, H.R. 2015. Warning times for species extinctions due to climate change. *Global Change Biology* 20(3): 1066–1077.

Stoner, A.W., Davis, M.H. and Booker, C.J. 2012. Negative consequences of Allee effect are compounded by fishing pressure: comparison of Queen Conch reproduction in fishing grounds and a marine protected area. *Bulletin of Marine Science* 88(1): 89–104.

Thompson, C.J., Koshkina, V., Burgman, M.A., Butchart, S.H.M. and Stone, L. 2017. Inferring extinctions II: a practical, iterative model based on records and surveys. *Biological Conservation* 214: 328–335.

Thuiller, W., Brotons, L., Araújo, M.B. and Lavorel, S. 2004. Effects of restricting environmental range of data to project current and future species distributions. *Ecography* 27: 165–172.

Thuiller, W., Münkemüller, T., Schiffers, K.H., Georges, D., Dullinger, S., Eckhart, V.M., Edwards Jr, T.C., Gravel, D., Kunstler, G., Merow, C., Moore, K., Piedallu, C., Vissault, S., Zhao, M., Tuleya, R.E., Bender, M., Vecchi, G.A., Villarini, G., Chavas, D. 2015. Global Projections of Intense Tropical Cyclone Activity for the Late Twenty-First Century from Dynamical Downscaling of CMIP5/RCP4.5 Scenarios. *Journal of Climate* 28: 7203–7224.

Trivedi, M.R., Berry, P.M., Morecroft, M.D. and Dawson, T.P. 2008. Spatial scale affects bioclimate model projections of climate change impacts on mountain plants. *Global Change Biology* 14: 1089–1103.

White, G.C., Franklin, A.B. and Shenk, T.M. 2002. Estimating parameters of PVA models from data on marked animals. In: S.R. Beissingerand and D.R. Mccullough (eds) *Population Viability Analysis*, pages 169–190. University of Chicago Press, Chicago.

Wiig, Ø., Amstrup, S., Atwood, T., Laidre, K., Lunn, N., Obbard, M., Regehr, E. and Thiemann, G. 2015. *Ursus maritimus*. *The IUCN Red List of Threatened Species* 2015: e.T22823A14871490. http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en

Wilhelm, S.I., Rail, J.-F., Regular, P.M., Gjerdrum, C. and Robertson, G.J. 2016. Large-scale changes in abundance of breeding Herring Gulls (*Larus argentatus*) and Great Black-Backed Gulls (*Larus marinus*) relative to reduced fishing activities in southeastern Canada. *Waterbirds* 39: 136–142, 137.

Zimmermann, N.E., Zurell, D. and Schurr, F.M. 2014. Does probability of occurrence relate to population dynamics? *Ecography* 37(12): 1155–1166.

Zurell, D., Elith, J. and Schröder, B. 2012. Predicting to new environments: Tools for visualizing model behaviour and impacts on mapped distributions. *Diversity and Distributions* 18: 628-634.

14. Appendix: Summary of Changes to the Guidelines

Changes in version 16 (March 2024)

Section 12.1: Major restructuring; substantial new text and a new figure. Note that many of the subsection numbers within section 12.1 have been changed.

Changes in version 15.1 (July 2022)

Section 3.2.3: Minor correction to the text "A precautionary attitude (i.e., low risk tolerance) will classify a taxon as threatened unless it is highly likely that it is not threatened…" (the word "likely" replaces "unlikely")

Sections 5 and 5.4: Text changed to clarify that for criterion A "reversible" refers to the reduction and "understood" and "ceased" refer to the causes of reduction.

Changes in version 15 (January 2022)

Section 3.1: Clarification of data quality categories.

Section 4.5.4: Emphasizing that it is incorrect to calculate a simple (unweighted) average of the 3-generation reduction amounts of the different subpopulations.

New Section 4.5.6 on differentiating fluctuations from reduction.

Section 4.8: Clarification of habitat vs. population fragmentation.

Section 4.11: Further clarification of the definition of Location.

Section 5: Discussion of the reasons for scaling reductions with generation time and for calculating reduction over 3 rather than fewer generations.

Section 10.1: Using uncertainty guidelines for NT.

Section 10.1: Discontinuing NT for conservation-dependent species (also deletion in Section 5.4).

Sections 11.1 and 11.3: Emphasizing the importance of, and further guidance on, the concept of "exhaustive surveys."

Changes in version 14 (July 2019)

Section 4.11: Clarification of "rapidly" in the definition of location.

Section 11: New guidelines for listing taxa as EX or CR(PE) (or as EW or CR(PEW)).

Section 11.1: Use of EW for plant or fungal taxa represented by viable propagules in storage facilities.

Changes in version 13 (March 2017)

Section 2.3: Additional explanation of the basis for using the highest category of threat.

New Section 2.3.1 on the quantitative thresholds.

Section 4.3.1. Clarification of "reproduction" and biased sex ratios.

Section 4.4. Averaging generation length over all individuals; new paragraph on definition of "age".

Section 4.6. Documenting the location of declines in relation to the species' range.

Sections 4.10, 4.10.1, 4.10.3 Extensive edits to clarify issues of scale in estimating AOO.

Section 4.10.7. Clarification of scaling the estimated area of occupied habitat derived from habitat maps for calculating AOO and EOO.

New Section 4.10.8 about the effect of sampling effort and detectability on estimates of AOO.

New Section 4.10.9 on the complementarity of AOO, EOO and number of locations.

Section 4.11. Clarification that assessments should consider all areas whether they are under threat or not.

Section 12.1.12. Clarification of the use of elevation in bioclimate models.

Changes in version 12 (February 2016)

Section 2.1.2: Text on applying the criteria in very small geographic areas.

Section 2.2: Use of the term "red-listed".

Section 2.2.1: Clarifying the 5-year rule for transfer between categories.

Section 4.2: Clarifying subdivision.

Section 4.3.1: Text on suppressed individuals.

Section 4.3.2: Edits related to clonal colonial organisms.

Section 4.4: Additional explanation of the "pre-disturbance" generation time.

Section 4.5: Major restructuring and new text on calculating reductions. Also, the spreadsheet file CriterionA_Workbook.xls is updated with additional tabs demonstrating basic calculations.

Section 4.5.3: This new section includes most of former section 5.8; the rest of former section 5.8 is merged with 4.5.1.

Section 4.8: Clarification of habitat vs. population fragmentation.

Section 4.9: Additional explanation of the risk-spreading function of EOO.

Section 5: Additional explanation of the rationale of criterion A.

Section 5.1: New section on the basis of reductions, including a new table.

Section 5.2: This is the former section 5.1 (use of time caps).

Section 5.3: This is the former section 5.2 (how to apply A4).

Section 5.4: This is the former section 5.3 (the ski-jump effect), with a more descriptive title, and expanded text (point (3) at the end).

Section 5.5: This is the former section 5.4 (severely depleted populations), with a more descriptive title, and additional text and examples.

Section 5.6: This is the former section 5.5 (fisheries), now divided into two subsections, with additional text discussing issues related to fisheries management.

Section 5.7: This is the former section 5.6 (was titled “Trees”).

Section 5.8: This is the former section 5.7 (loss of habitat and reduction).

Former section 5.8 is merged with parts of section 4.5 (see above).

Section 10.1: Definition of a "targeted taxon-specific or habitat-specific conservation or management programme".

Section 10.4: New section on when it is not appropriate to use DD.

Section 12.1: Major restructuring; substantial new text and a new Figure. Note that many of the subsection numbers within section 12.1 have been changed.

This appendix is expanded to cover all previous versions.

Changes in version 11 (February 2014)

Section 2.1.3: Substantial changes related to introduced taxa and subpopulations.

Section 2.1.4: New section on managed subpopulations.

Section 3.2.3: New guidance on setting the dispute tolerance and the risk tolerance values.

Section 4.4: New paragraph on using pre-disturbance generation length.

Section 4.9: Additional explanation on using minimum convex polygon for EOO.

Section 4.10.7: Expanded discussion on using habitat maps and models for EOO and AOO.

Section 11.1: New paragraph on using EW when none of the subpopulations are wild.

Changes in version 10.1 (September 2013)

Section 11.2.1: New paragraph added.

Minor corrections in sections 4.3, 4.5, and 13.

Changes in version 10 (February 2013)

Section 2: Table 2.1 and Figure 2.1 updated; minor changes to the last paragraph of section 2.1.2; clarification of LC and NT categories and minor corrections in sections 2.2 and 2.3.

Section 4.1: Clarification of the definitions of population and population size.

Section 4.2: Clarifying the relation between a species' mobility and the delineation of its subpopulations.

Section 4.6: The relation between continuing decline and "current population trend."

Section 4.11: The number of locations when there are two or more serious plausible threats.

Section 7: New paragraphs (third and fourth) clarifying the subcriteria i and ii of C2a.

Section 8: Minor change to the 2nd paragraph, clarifying "very short time period" in D2.

Changes in version 9.0 (September 2011)

Section 4.4: The guidelines for calculating generation length are revised substantially.

Section 4.5.1: Added text: "If populations fluctuate widely, or oscillate with periods longer than generation time, fitting a time series longer than three generations may give a more representative estimate of the long-term population reduction. However, regardless of the length of the time series fitted, the reduction should be calculated for the most recent three generations. The model to be fitted should be based on the pattern of decline, which may be inferred from the type of threat."

Section 4.6: Two new paragraphs (3rd and last), and addition to the 5th paragraph ("Note that …").

Section 5.5: Sentence modified: "If declines continued, there would be reason for concern; in this case a new assessment, against all 5 criteria, may indicate that the taxon is still threatened."

Section 10.1: A new example added to the list of examples where an NT listing would be justified.

Section 10.3: Substantial revision to the 2nd DD tag, which is now named "Taxonomic uncertainty explains lack of information."

This appendix added.

Changes in version 8.1 (August 2010)

Minor corrections, incl. to Table 2.1

Changes in version 8.0 (March 2010)

Section 2.3: Minor change to refer to the new section 12

Section 4.10.5: Several minor changes, mostly to equations to make them clearer.

Figure 4.4: New figure

Section 5: New paragraphs (third and fourth) to clarify subcriteria a and b.

Section 5: New sentence: "If any of the three conditions (reversible and understood and ceased) are not met in a substantial portion of the taxon's population (10% or more), then A2 should be used instead of A1."

Section 8: Changes in the first and third paragraph to clarify, and to give an example for "a very short time" (within one or two generations).

Section 12: New section on Threatening Processes, including guidelines for applying the criteria to species impacted by global climate change.

Changes in version 7.0 (August 2008)

Section 2.1.1: Expanded guidance on taxonomic scales, including newly described and undescribed species, and subpopulations.

Section 2.2.1: Detailed definition of the reasons for transfer between categories.

Section 4: Additional guidance on calculating the number of mature individuals, generation time, future reduction, EOO, and number of locations.

Section 10.3. Data deficient flags.

Section 11. New section on the extinct categories and the PE tag.

Changes in version 6.2 (Dec 2006)

Section 2.3: Changes to paragraph on comparison of criteria A-D vs E.

Section 8: Minor changes to section on taxa known only from the type locality.

Changes in version 6.1 (Aug 2006)

Minor changes, including version number on page 1.

Changes in version 6.0 (July 2006)

Section 4.3.2: Mature individual for colonial or modular organisms

Section 4.9: Clarification on EOO, including risk-spreading; discouraging exclusion of discontinuities or disjunctions except in extreme circumstances, but encouraging it for calculating change in EOO; EOO of migratory species.

Section 4.10: Further explanation of why a specific scale is necessary for AOO; new section on *AOO based on habitat maps and models*.

Section 4.11: Guidance on number of locations with different threats in different areas

Section 5: How to apply criterion A4; discussion of population data contradicting habitat data; description of the workbook file (CriteriaA workbook.xls) accompanying the guidelines.

Section 6: Guidelines for applying Criterion B (numbering for subcriterion a)

Section 8. New guidelines and an example for applying Criterion D2

Section 10: Examples for when to use and when not to use NT and DD.

Changes in version 5.0 (April 2005)

Expanded sections on extreme fluctuations and severely fragmented; NT based on conservation dependence

Changes in version 4.0 (March 2004)

New section on *Transfer between categories*.

Clarifications on continuing decline vs. reduction; criterion A basis; A1 vs. A2; A4.

Changes in version 3.0 (May 2003)

Additions to clarify issues related to taxa below the rank of variety, introduced taxa, generation length for clonal plants, specifying criteria for NT; new examples and references, and numerous minor edits.

Changes in version 2.0 (Jan 2003)

First version that covered all criteria and definitions (48 pages).

Changes in version 1.1 (Dec 2001)

Minor additions such as clarifying that "non-overlapping" is not "isolated" (10 pages).

Version 1.0 (June 2001)

This first version was titled "*Guidelines for Assessing Taxa with Widely Distributed or Multiple Populations Against Criterion A*" and became section 5.8 in version 2.